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Par

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EVALUATION OF THE SHRIMP FISHERY IN THE PERNAMBUCO, NORTHEAST OF BRAZIL: AN ECOSYSTEM APPROACH

(*L'évaluation de la pêche de la crevette en Pernambuco, au nord-est du Brésil : une approche écosystémique*)

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"*Educação não transforma o mundo.*
Educação muda as pessoas.
Pessoas mudam o mundo – Paulo Freire"

CONTENTS

LIST OF FIGURES

Sections

RÉSUMÉ EN FRANÇAIS

INTRODUCTION

STRUCTURE AND OBJECTIVES

CHAPTER 1

CHAPTER 2

CHAPTER 3

CHAPTER 4

FINAL CONSIDERATIONS FOR MANAGEMENT SUPPORT

PERSPECTIVES AND WEAKNESSES TO IMPROVE

LIST OF TABLES

Sections

CHAPTER 1

CHAPTER 2

CHAPTER 3

CHAPTER 4

Abstract

The shrimp fishery is responsible for one of the main anthropogenic impacts on the seabed and associated communities. In Northeast Brazil, this fishery is of small scale, characterized mainly by weak or completely absence of management; and by a high socio-economic importance for many people that depend on this activity as source of income and food. The overall aim of this thesis is to assess the current and potential future impact of fishing and environmental changes from the multiple methods adapted to data-limit framework under the scope of Ecosystem Approach to Fishery (EAF) in Sirinhaem, Pernambuco, using, as study case, a small-scale shrimp trawling in Northeastern Brazil. Firstly, an integrative view of the fishery was carried out encompassing the characteristics of environment and fishing aspects, and the dynamics of the target and bycatch species (Chapter 1). The importance of crustaceans, especially the target species (shrimps) in the support to coastal food-web was accessed using two complementary tools (stomach content and stable isotope) (Chapter 2). A temporally dynamic model (Ecosim) was built to evaluate the potential isolated and combined effects of different fishing effort control policies and environmental changes on marine resources and ecosystem (Chapter 3). Finally, a semi-quantitative risk analysis – PSA (Productivity and Susceptibility Analysis) adapted to regional conditions, was developed in order to evaluate, for the first time, the vulnerability and the potential risk (low, moderate and high) of the target and non-target species exploited by the trawl fishing in the north-eastern Brazil. Shrimp fishing occurs in shallow waters at depth varying from 10 to 20 m associated to mud zones. The abundance and catches of target and non-target species are positively correlated to the rainfall season. Penaeidae shrimps are the main targets, particularly the seabob shrimp (*Xiphopenaeus kroyeri*) the most abundant, and the pink shrimp (*Penaeus subtilis*) and white shrimp (*Penaeus schmitti*) with the high market-values. These species, together with other invertebrates (e.g., worms and crabs), are extremely important preys for the fish fauna, highlighting their importance in the food web. Potential decreasing the abundance of these preys, including shrimps, due to cumulative effects of trawling in the area, may lead to intense changes in the trophic structure of the ecosystem affecting the food web and the sustainability of the fishery. Considering the target species, although the traditional stock assessment carried out in the region do not indicate overexploitation, the pink shrimp (*P. subtilis*) is o more affected by the increasing of effort than *P. schmitti* and *X. kroyeri.* However, considering the particularities of our case study and without accounting for the effect of environmental changes, not adopting effort control measures for the current trawling conditions do not appear to cause major losses for target species in terms of biomass and catches. Amongst the fish bycatch, the Scianidae and Pristigasteridae families were the most important species in terms of abundance and biomass, with most of them being consumed by the local community and classified as the moderate risk, given its high resilience (e.g., *Pellona harroweri*, *Isopisthus parvipinnis*, *Chirocentrodon bleekerianus*). However, Elasmobranchs and catfish, often discarded or consumed; hakes and croakers' fishes, usually commercialized; were assigned as high vulnerable, mainly given the low productivity (medium to long life-span and low spawning potential reproduction, for elasmobranchs and catfish) and/or the high capture rates of young individuals and overlap with the fishing areas (mainly for the hakes and croakers). Considering the integrated results here observed, we evaluated the possible regulation which would be adapted to our study case. Given its reduced extension of the fishing grounds, spatial management approaches (e.g., Marine Protect Area – MPA or no-fishing zones) maybe not very effective as a possible regulation in the region. In addition, large effort reductions or the definition of size and gear limitations did not appear to be necessary measures, considering that, according to the traditional stock assessment, the target species are being exploited at biologically accepted levels. However, the controlled decrease of the trawling effort up to 10% were promising, with better fishing management performance than the closed season, which did not present significant improvements in terms of ecosystem functioning. Considering that several bycatch fish species are also potentially vulnerable to bottom trawling, given its biology, ecology and importance for other fleets, associated to the lack of studies, they should be assigned as priority for management and data collection. The use of Bycatch Reduction Devices (e.g., fisheye, grid and square mesh) to exclude bycatch may be one alternative however, given crucial role of the bycatch to food security on small-scale fisheries, as in our case, its viability needs to be better evaluated in terms of the socio-economic aspects. Finally, regardless of what fishery regulation may be applied in the management of small-scale shrimp fisheries in Sirinhaém, Northeastern Brazil, we found clear evidence that environmental changes (e.g., rainfall, primary productivity), consequence of the climate changes, cause significant adverse impacts in the ecosystem. These effects should be considered in any eventual regulatory measure, since the cumulative effect environment changes and fishery, considerably threat the ecosystem, and consequently, the sustainability of the activity.

Keywords: Ecosystem Approach to Fisheries, Tropical fisheries, Small-scale fisheries, Trawling, EwE, Isotope Stable Analysis, PSA, Bycatch, Brazil

Resumo

A pesca do camarão é responsável por um dos principais impactos antropogênicos sobre o fundo marinho e comunidades associadas. No Nordeste do Brasil, esta pescaria é de pequena escala, caracterizada principalmente pela fraca ou total ausência de manejo; e por uma alta importância sócio-econômica para muitas pessoas que dependem desta atividade como fonte de renda e alimento. O objetivo geral desta tese é avaliar o impacto atual e potencial futuro da pesca e das mudanças ambientais a partir dos múltiplos métodos adaptados à estrutura limite de dados sob o escopo da Abordagem Ecossistêmica para Pesca (AEP) em Sirinhaem, Pernambuco, como estudo de caso para pesca de arrasto de camarão em pequena escala no nordeste do Brasil. Primeiramente, uma visão integradora da pesca foi realizada englobando as características do meio ambiente, aspectos da pesca, e a dinâmica das espécies-alvo e das capturas acessórias (Capítulo 1). A importância dos crustáceos, especialmente das espécies-alvo (camarões) no suporte da rede costeira de alimentação, foi acessada utilizando duas ferramentas complementares (conteúdo estomacal e isótopo estável) (Capítulo 2). Um modelo dinâmico-temporal (Ecosim) foi construído para avaliar os potenciais efeitos isolados e combinados de diferentes políticas de controle do esforço de pesca e mudanças ambientais nos recursos marinhos e no ecossistema (Capítulo 3). Finalmente, uma análise de risco semi-quantitativa - PSA (Productivity and Susceptibility Analysis) adaptada às condições regionais, foi desenvolvida a fim de avaliar, pela primeira vez, a vulnerabilidade e o risco potencial (baixo, moderado e alto) das espécies alvo e não alvo exploradas pela pesca de arrasto no nordeste do Brasil. A pesca do camarão ocorre em águas rasas em profundidades que variam de 10 a 20 m associados a zonas de lama. A abundância e as capturas de espécies alvo e não alvo estão correlacionadas positivamente com o período das chuvas. Os camarões Penaeidae são os principais alvos, particularmente o camarão sete-barbas (*Xiphopenaeus kroyeri*) o mais abundante, o camarão rosa (*Penaeus subtilis*) e o camarão branco (*Penaeus schmitti*) com os altos valores de mercado. Estas espécies, juntamente com outros invertebrados (por exemplo, poliquetas e caranguejos), são presas extremamente importantes para a fauna de peixes, destacando sua importância na teia alimentar. Uma diminuição da abundância dessas presas, incluindo camarões, devido aos efeitos cumulativos do arrasto na área, pode levar a mudanças intensas na estrutura trófica do ecossistema, afetando a teia alimentar e a sustentabilidade da pesca. Considerando as espécies-alvo, embora a avaliação tradicional dos estoques realizada na região não indique superexploração, o camarão rosa (*P. subtilis*) também é mais afetado pelo aumento do esforço do que *P. schmitti* e *X. kroyeri*. Entretanto, considerando as particularidades de nosso estudo de caso e sem levar em conta o efeito das mudanças ambientais, não adotar medidas de controle de esforço para as atuais condições de arrasto não parece causar grandes perdas para as espécies-alvo em termos de biomassa e capturas. Entre as capturas acessórias de peixes, as famílias Scianidae e Pristigasteridae foram as espécies mais importantes em termos de abundância e biomassa, sendo a maioria delas consumidas pela comunidade local e classificadas como de risco moderado, dada sua alta resiliência (ex., *Pellona harroweri*, *Isopisthus parvipinnis*, *Chirocentrodon bleekerianus*). Entretanto, elasmobrânquios, bagres freqüentemente descartados ou consumidos; pescadas e corvinas, geralmente comercializados; foram considerados altamente vulneráveis, principalmente dada a baixa produtividade (média a longa vida útil e baixo potencial de reprodução, para elasmobrânquios e bagres) e/ou as altas taxas de captura de indivíduos jovens e se sobrepõem às áreas de pesca (principalmente para as pescadas e corvinas). Considerando os resultados integrados aqui observados, avaliamos possíveis regulamentações que poderiam ser adaptadas ao nosso caso de estudo. Dada sua extensão, as abordagens de gerenciamento espacial (ex., Área de Proteção Marinha – APAs ou zonas de exclusão de pesca) talvez não sejam muito eficazes como uma possível regulamentação na região. Além disso, grandes reduções do esforço ou a definição de tamanho e limitações das artes não pareciam ser medidas necessárias, considerando que, de acordo com a avaliação tradicional dos estoques, as espécies alvo estão sendo exploradas em níveis biologicamente aceitos. Entretanto, a diminuição controlada do esforço de arrasto próxima a 10% foi promissora, com melhor desempenho para manejo da pesca do que o período de defeso, que não apresentou melhorias significativas em termos de funcionamento do ecossistema. Considerando que várias espécies de peixes da fauna acompanhante também são potencialmente vulneráveis ao arrasto de fundo, dada sua biologia, ecologia e importância para outras frotas, associadas à falta de estudos, elas devem ser prioridade para o gerenciamento e coleta de dados. O uso de dispositivos de redução de capturas acessórias (ex., olho de peixe, grade e malha quadrada) para excluir as capturas acessórias pode ser uma alternativa, porém dado o papel crucial das capturas acessórias para a segurança alimentar na pesca em pequena escala, como no nosso caso, sua viabilidade precisa ser melhor avaliada em termos dos aspectos sócio-econômicos. Finalmente, independentemente de qual regulamentação da pesca possa ser aplicada no manejo da pesca de camarão em pequena escala em Sirinhaém, nordeste do Brasil, encontramos evidências claras de que mudanças ambientais (ex., chuvas, produtividade primária), conseqüência das mudanças climáticas, causam impactos adversos significativos no ecossistema. Estes efeitos devem ser considerados em qualquer eventual medida regulatória, uma vez que o efeito cumulativo das mudanças ambientais e da pesca, ameaça consideravelmente o ecossistema e, conseqüentemente, a sustentabilidade da atividade.

Palavras-chave: Abordagem Ecossistêmica da Pesca, Pesca Tropical, Pesca de Pequena Escala, Arrasto, EwE, Análise Estável Isotopo, PSA, Capturas acessórias, Brasil

RÉSUMÉ EN FRANÇAIS

Le chalutage de fond, l'une des techniques de pêche les plus utilisées dans le monde impacte négativement les habitats marins en raison de ses niveaux élevés de prises accessoires, affectant (i) la disponibilité des proies pour les poissons démersaux, ce qui pourrait détériorer la condition physique des poissons (Johnson *et al.*, 2015), (ii) la structure trophique (Ramalho *et al.*, 2018) et (iii) le rendement des captures dans les zones les plus affectées par le chalutage (Collie *et al.*, 2017). Il modifie également drastiquement le substrat et les communautés benthiques (Halpern *et al.*, 2008; Ortega *et al.*, 2018a), impactant la faune des fonds marins (Hiddink *et al.*, 2017).

Les pêcheries de crevettes au chalut, notamment tropicales, sont la plus grande source de rejets mondiaux, représentant 27,3% (1,86 millions de tonnes) du total estimé des rejets entre 1990 et 2001 (Kelleher, 2005). Il n'existe pas d'estimations actualisées des rejets de la pêche à la crevette dans le monde et le niveau actuel est inconnu. Les prises accessoires peuvent être définies comme les captures d'espèces non ciblées pouvant avoir une valeur commerciale, être consommées par l'équipage et les communautés locales (pêches artisanales), utilisées comme appâts (pêches industrielles), ou rejetées au port ou en mer (Davies *et al.*, 2009; Gilman *et al.*, 2014).

Au Brésil, les crevettes sont exploitées par une pêcherie multi-espèces le long de toute la côte, particulièrement dans les zones peu profondes avec des chalutiers de fond motorisés (Costa *et al.*, 2007), où les Penaeidae représentent la cible principale (Lopes, 2008). Trois systèmes de pêche, qui diffèrent dans leur taille, technologie et volume de capture se rencontrent sur la zone côtière du Brésil (Figure 1): (i) la pêche industrielle, présente dans la région Nord (embouchure du fleuve Amazone), le Sud-Est et le Sud du Brésil; (ii) la pêche semi-industrielle avec une technologie et une puissance de pêche intermédiaires, et (iii) la pêche artisanale, opérant le long de toute la côte, impliquant un plus grand nombre de pêcheurs et se caractérisant par un faible niveau de technologie, de capture et de profit (Dias-Neto, 2011). Les pêcheries industrielles de crevettes génèrent des taux élevés de prises accessoires qui sont effectivement rejetées en mer : 5,5 à 10,5 kg de rejets pour 1 kg de crevettes pêchées dans le Sud du Brésil (Vianna and Almeida, 2005), et 2,2 à 11 kg pour 1 kg de crevettes débarquées dans le Nord (Paiva *et al.*, 2009). Le taux de prises accessoires de la pêche artisanale à la crevette dans le Nord-Est est estimé á 1 à 5 kg de poissons pour 1 kg de crevettes capturées, la majorité étant consommée ou commercialisée localement (Silva-Júnior et al., 2019).

Les principales politiques de gestion de la pêche à la crevette se concentrent uniquement sur les espèces cibles et sont limitées par des permis de pêche, la fermeture saisonnière des activités de pêche, la réglementation de la taille des mailles des filets de pêche, le contrôle des navires (taille) et la taxe sur le pétrole (Santos, 2010; Dias-Neto, 2011). Ces initiatives de gestion ne considèrent pas la capture d'espèces non ciblées, contrairement au Code de conduite pour une pêche responsable (FAO, 1995). En

effet, ce dernier recommande que toutes les captures, et pas seulement celles des espèces ciblées, soient gérées de manière écologiquement durable, sur les principes de la co-gestion participative et de la gestion des pêches basée sur l'écosystème (EBFM) ou l'approche écosystémique des pêches (AEP). À cette fin, ces mesures de gestion doivent être précédées ou accompagnées d'approches qui intègrent le maximum d'informations possibles pour identifier les espèces risquant d´être les plus affectées par la pêche en tenant compte des effets des variations environnementales et des réglementations politiques.

L'objectif principal de cette thèse est d'évaluer le contexte actuel et le potentiel futur impact de la pêche et des changements environnementaux sur l'écosystème côtier de Sirinhaém en tant qu'étude de cas pour le chalutage de crevettes à petite échelle dans le nord-est du Brésil, ainsi que de contribuer à la réflexion sur la mise en place d'éventuelles mesures de gestion. Premièrement, une vision intégrative des multiples dimensions abiotiques et biotiques liées à la pêche artisanale de la crevette sur la côte sud de Pernambuco a été abordée dans le chapitre 1. Cette synthèse a été suivie par le deuxième chapitre qui se concentre sur la détermination de l'importance des espèces cibles (crevettes) en tant que proies pour les espèces non-cibles (poissons capturés accidentellement), et discute l'effet possible du chalutage de fond sur les interactions trophiques, qui peuvent affecter la communauté marine locale et la durabilité de la pêche. Le chapitre 3 s'attache à promouvoir un diagnostic des effets des mesures réglementaires encore inconnues dans la région, afin que les gestionnaires répondent aux objectifs de conservation des écosystèmes et de développement durable de la pêche. Bien que nous ayons identifié les espèces cibles de l'écosystème affectées par les changements environnementaux et par la pêche, ce résultat était limité au niveau du groupe pour les prises accessoires. Le chapitre 4 a été spécifiquement dédié à l'évalution de la vulnérabilité et du risque potentiel des espèces cibles et non cibles exploitées par la pêche artisanale de la crevette.

CHAPTER 1. SMALL SCALE SHRIMP FISHERY IN NORTHEAST BRAZIL: AN OVERVIEW

Une étude intégrée de la pêcherie a été réalisée dans ce premier chapitre, englobant les caractéristiques de l'environnement et les aspects de la pêche, ainsi que la dynamique des espèces cibles et des prises accessoires, afin de promouvoir le durabilité à la gestion de l'écosystème. La zone de pêche au chalut à Sirinhaém, au nord-est du Brésil, est limitée aux fonds vaseux proches de la côte (10-20 m de profondeur) et les abondances et les périodes de reproduction des espèces, ainsi que la dynamique de la pêche, sont principalement contrôlés par les facteurs environnementaux (par exemple, les précipitations, la chlorophylle) (Figure 1).

Dans la pêche artisanale à la crevette au chalut pratiquée dans le nord-est du Brésil, plus précisément dans le Sirinhaém, la quantité de prises accessoires (non ciblées) capturées (en poids) est inférieure à celle des crevettes (ciblées). Parmi les crevettes, seabob (*Xiphopenaeus kroyeri*) est la plus abondante, puis la crevette rose (*Penaeus subtilis*) et la blanche (*Penaeus schmitti*), dont la valeur commerciale est plus élevée. Bien qu'elles soient largement capturées, ces espèces cibles sont des stratèges *r* avec une petite taille, une croissance rapide, une maturité précoce, un fort potentiel de reproduction et elles sont résilientes. Par conséquent, selon l'évaluation traditionnelle des stocks, elles sont exploitées à des niveaux biologiquement acceptables (Silva *et al.*, 2015, 2018; Lopes *et al.*, 2017).

Parmi les prises accessoires de poissons (93 espèces), les familles Scianidae et Pristigasteridae étaient les plus importantes en termes d'abondance et de biomasse, la majorité de ces espèces étant consommée ou commercialisée localement, comme source complémentaire de nourriture et de revenus. Cependant, ces espèces non ciblées sont souvent ignorées dans les mesures de gestion, étant donné leur grande importance socio-économique pour les communautés locales de la région. Elles doivent être mieux évaluées dans le cadre de l'approche écosystémique de la pêche (AEP) en tenant compte de leurs interactions dans le réseau trophique, ce qui est essentiel pour évaluer la conservation des espèces et la durabilité de la pêche.

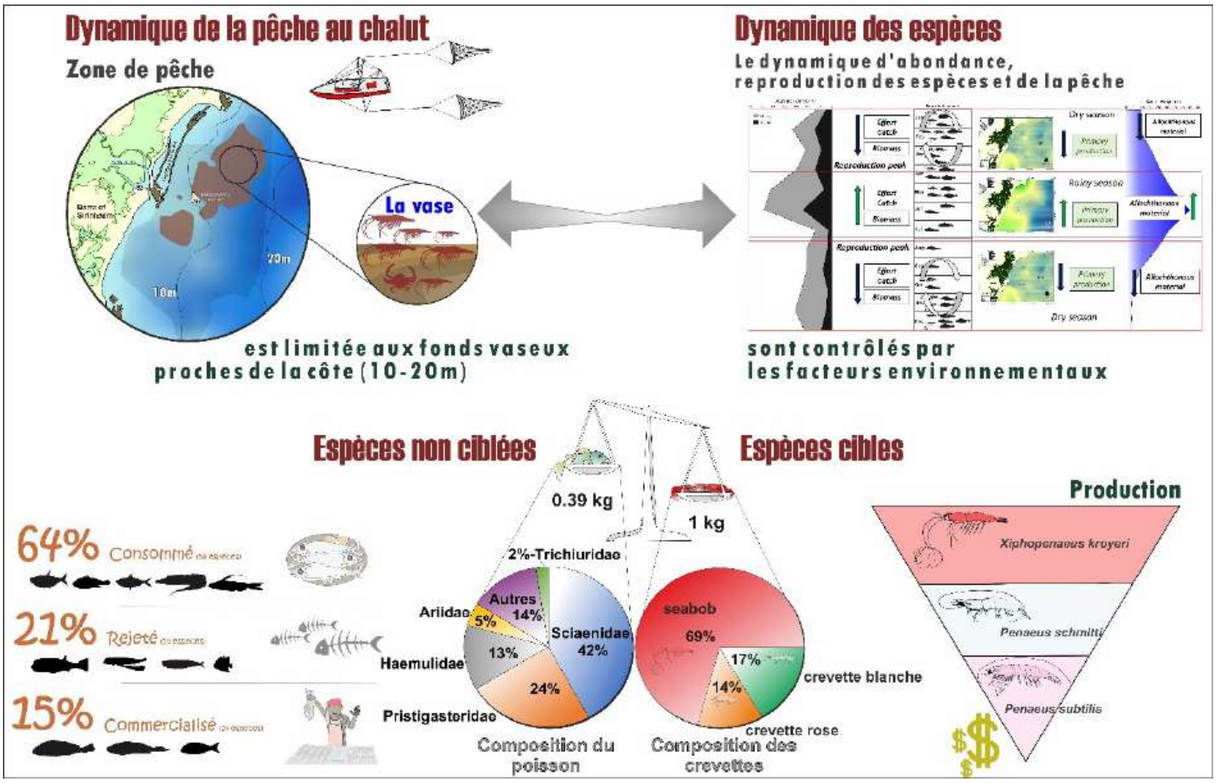


Figure 1. Principaux résultats du chapitre 1 de la thèse.

CHAPTER 2. TROPHIC STRUCTURE OF NEKTOBENTHIC COMMUNITY EXPLOITED BY A MULTISPECIFIC BOTTOM TRAWLING FISHERY IN NORTHEASTERN BRAZIL

Dans ce chapitre, avec deux outils complémentaires l'analyse des contenus stomacaux et l'analyse des isotopes stables, nous avons décrit la contribution des sources benthiques et l'importance des crustacés, en particulier des crevettes, dans le transfert de l'énergie de la base du réseau trophique vers les niveaux trophiques supérieurs côtier à Sirinhaém, nord-est du Brèsil. La présence de fonds vaseux dans ces zones côtières, qui favorisent généralement de grandes occurrences d'invertébrés benthiques, tels que les vers et les crustacés, explique cette énorme importance pour l'alimentation de la faune de poissons côtiers. En raison de l'absence de réglementation des activités de chalutage de fond dans la zone, les effets cumulatifs du chalutage sur les paramètres de la population (par exemple, la taille et la consommation alimentaire), en diminuant potentiellement l'abondance des proies benthiques, peuvent entraîner des changements dans la structure trophique de l'écosystème, ce qui peut provoquer un effet de cascade trophique (top-down ou bottom-up) et potentiellement affecter le réseau trophique et la durabilité de la pêche (Figure 2).

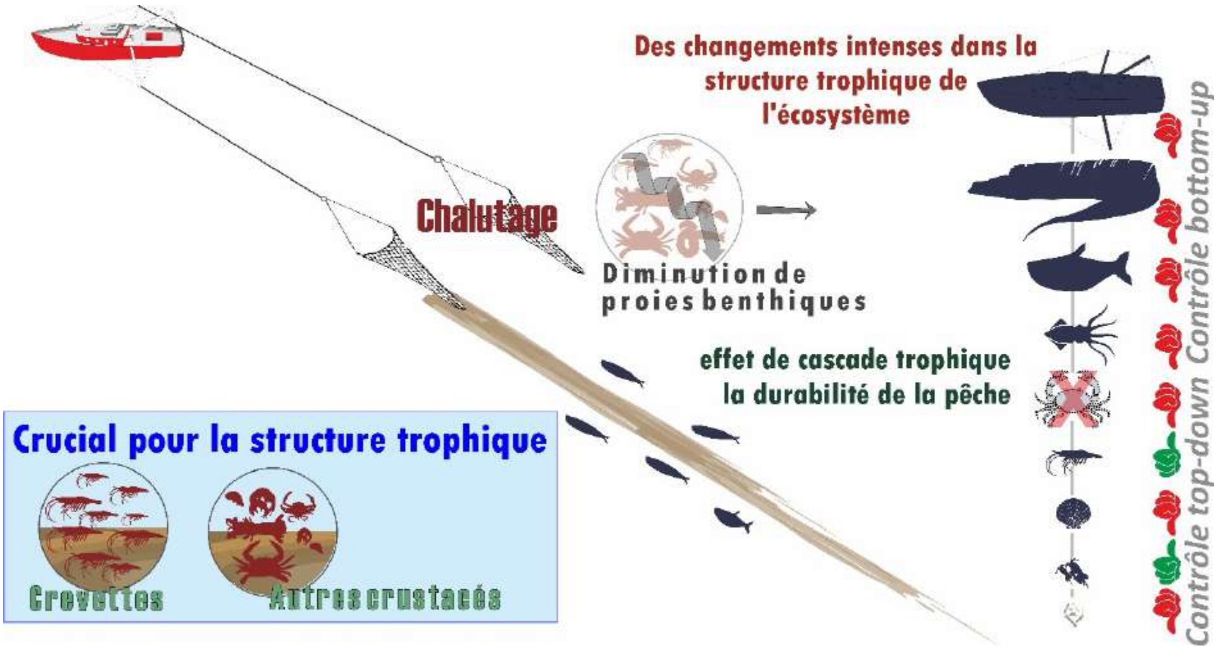


Figure 2. Principaux résultats du chapitre 2 de la thèse.

CHAPTER 3. HOW THE FISHING EFFORT CONTROL AND ENVIRONMENTAL CHANGES AFFECT THE SUSTAINABILITY OF A TROPICAL SHRIMP SMALL SCALE FISHERY

Bien que les chapitres précédents présentent un bilan de l'écosystème en définissant où, comment et quelles espèces sont capturées par le chalut de fond, ils ne quantifient pas les effets possibles de cette pêche et des facteurs environnementaux au niveau individuel ou de l'écosystème, notamment dans le cadre actuel de non-réglementation du chalutage. Dans ce chapitre, il s'agissait, à notre connaissance, de la première tentative d'évaluation de l'impact potentiel des pêcheries de crevettes au Brésil en utilisant une approche écosystémique avec un modèle Ecopath and Ecosim (EwE). Les tendances des indicateurs de l'écosystème (par exemple, les indices basés sur la biomasse, le niveau trophique et la taille) ont montré le rôle ascendant joué par la variabilité environnementale sur le fonctionnement et la structure de l'écosystème. Dans ce chapitre, la modélisation trophique montre que l'abondance des espèces est fortement associée aux facteurs environnementaux, comme souligné dans le chapitre 1, et, Nous avons démontré que la plus forte concentration de chlorophylle pendant la saison des pluies dans les eaux peu profondes près de l'embouchure de la rivière, où les pêcheries opèrent, peut avoir un impact sur l'abondance des crevettes et par conséquent sur la productivité des pêcheries. Cet effet environnemental est plus déterminant sur l'équilibre de l'écosystème et de la pêche que les mesures de gestion telles que la fermeture de la saison de pêche et les variations de l'effort de pêche de ±10% (Figure 3). Cependant, il est évident que dans un futur proche (2030), avec l'augmentation incontrôlée du chalutage combinée aux changements environnementaux globaux, des impacts négatifs significatifs affecteront le fonctionnement de l'écosystème. Néanmoins, une diminution contrôlée des activités des chalutiers de fond pourrait contribuer à réduire, même à de faibles niveaux, ces effets très négatifs et à maintenir un niveau similaire de débarquements, sans compromettre la structure de l'écosystème.

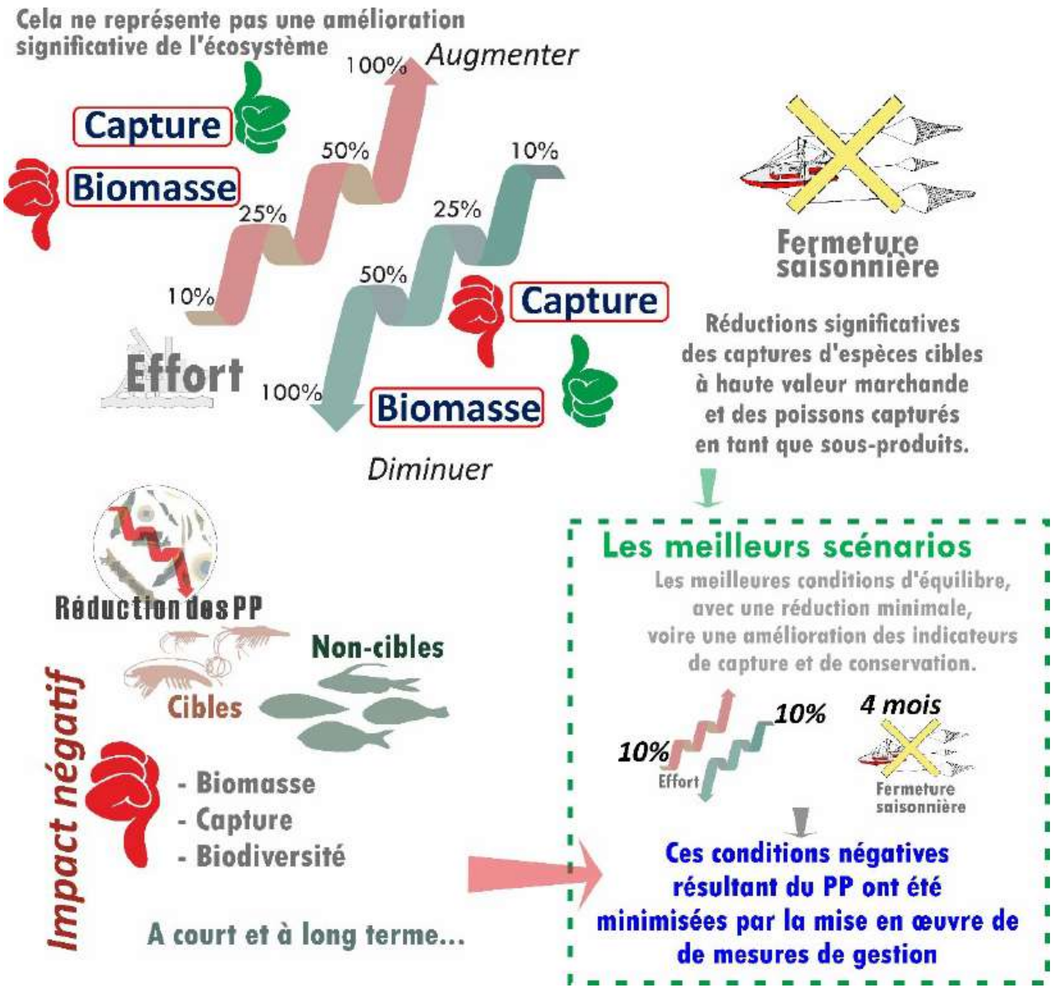


Figure 3. Principaux résultats du chapitre 3 de la thèse.

CHAPTER 4. VULNERABILITY OF MARINE RESOURCES AFFECT BY TROPICAL SHRIMP SMALL SCALE FISHERY IN A TROPICAL AREA

La quantité limitée d'informations disponibles, principalement sur les prises accessoires de poissons, a limité nos conclusions afin d'identifier, au niveau spécifique, les espèces les plus vulnérables au chalutage, et qui méritent une attention particulière de la part des gestionnaires. Compte tenu de l'importance mondiale des pêches artisanales, et de leurs prises accessoires en particulier, qui sont généralement négligées par les évaluations et par les décideurs, nous évaluons la vulnérabilité et le risque potentiel à un niveau spécifique des espèces cibles et non cibles exploitées par la pêche à la crevette. Pour cela, dans le chapitre 4, nous appliquons une évaluation semi-quantitative des risques écologiques, la PSA (Productivity and Susceptibility Analysis), qui appartient à la famille des modèles limités en données. Son calcul se base sur la productivité biologique et la capturabilité par l'engin de pêche. De plus, nous apportons une approche adaptée aux conditions régionales, en incorporant des incertitudes pour permettre une meilleure confiance des résultats. Les risques pour deux des principales espèces cibles (*X. kroyeri* et *P. subtilis*) bien que considérés comme élevés par l'une des méthodes utilisées pour l'estimation de la vulnérabilité, l'évaluation traditionnelle des stocks développée dans la région indique que ces espèces sont capturées dans un niveau d'exploration acceptable. Les élasmobranches, les poissons-chats souvent rejetés ou consommés localement (par exemple, *Pseudobatos percellens*, *Rhizoprionodon porosus* et *Bagre marinus*), les merlus et les Scianidae habituellement commercialisés (par exemple, *Micropogonias furnieri*, *Macrodon ancylodon* et *Cynoscion virescens*) ont été considérés comme des espèces de prises accessoires de haute vulnérabilité et devraient être prioritaires, dans pour une évaluation urgente et/ou la collecte de données (Figure 4).

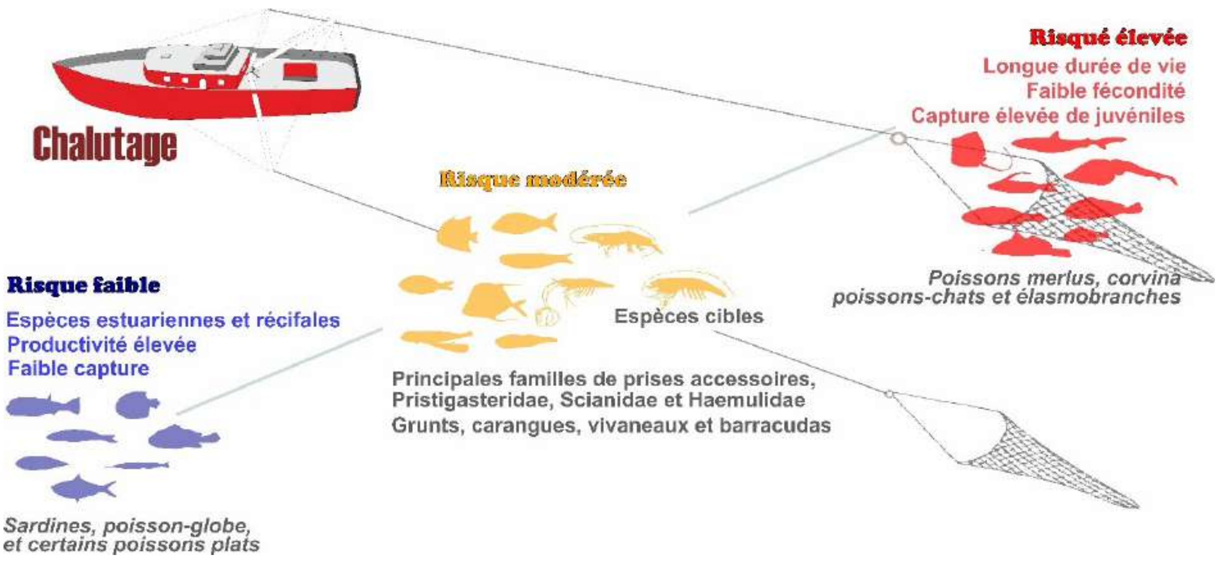


Figure 4. Principaux résultats du chapitre 4 de la thèse.

Les espèces les plus abondantes des prises accessoires (par exemple, *Pellona harroweri*, *Isopisthus parvipinnis*, *Chirocentrodon bleekerianus*) ont été classées à risque modéré et, compte tenu du rôle bénéfique des prises accessoires pour les communautés locales, comme indiqué au chapitre 1, ainsi que des effets négatifs potentiels d'un point de vue nutritionnel, économique et social dans un scénario de diminution des prises accessoires, comme indiqué au chapitre 3, elles méritent également une priorité de recherche.

CONCLUSION

Les résultats conjoints de tous les chapitres, nous permettent de conclure qu'actuellement, les espèces cibles ne constituent pas la principale menace du chalutage de crevettes à petite échelle dans la région. En outre, nous identifions plusieurs espèces non ciblées qui ne sont souvent pas prises en compte dans les mesures de gestion. Etant donné leur grande importance socio-économique de la région, elles doivent être mieux évaluées dans le cadre de l'AEP en tenant compte de l'effet sur l'ensemble de la dynamique trophique et la durabilité des prises accessoires, essentielles pour la sécurité alimentaire.

Sur la base de nos résultats, nous avons évalué les principales mesures de gestion appliquées dans les pêcheries de crevettes au Brésil. Bien que la réduction contrôlée de l'effort de pêche actuel de près de 10% soit prometteuse, la forte diminution de l'effort de pêche ou la définition de limites de taille et d'engins ne semblent pas être une mesure nécessaire, étant donné que, selon l'évaluation traditionnelle des stocks, les espèces cibles sont exploitées à des niveaux biologiquement acceptables. En ce qui concerne la période de fermeture de la pêche, nous n'avons pas observé d'amélioration importante de l'écosystème et de la pêche étant donné le schéma saisonnier de reproduction des espèces. La faible abondance des crevettes est liée à la saison sèche qui correspond au pic de reproduction de ces espèces, ce qui a pour effet de réduire fortement les activités de chalutage ou de les rendre économiquement non rentables en raison de la baisse de la production qui couvre à peine les coûts opérationnels de la pêche. Du point de vue de la gestion de l'espace, nous avons identifié que les principales zones de pêche étaient petites et limitées à des lits vaseux proches de la côte. Ainsi, étant donné son étendue, les approches de gestion spatiale (par exemple, les zones marines protégées ou les zones d'interdiction de pêche) ne sont peut-être pas très efficaces dans une éventuelle gestion des pêches dans la région.

Enfin, indépendamment des mesures qui peuvent être appliquées dans la gestion des pêcheries de crevettes à petite échelle à Sirinhaém, au nord-est du Brésil, nous avons trouvé des preuves claires que les changements environnementaux (par exemple, les précipitations, la productivité primaire) résultant des changements climatiques causent des impacts négatifs significatifs sur l'écosystème. Ainsi, les changement environnementaux devraient être pris en compte dans toute mesure réglementaire éventuelle, puisque l'effet cumulé de ces changements et de la pêche, menace considérablement la durabilité de l'écosystème et donc de la pêche.

INTRODUCTION

Demand for food and exploitation: cause and effect

All humans should have the right of an equal opportunity to satisfy their basic needs through the use of collectively owned natural resources from the natural law principle of the collective ownership of the earth by humankind (Risse, 2012). The UN Resolution 1803 (1962), focusing on natural resource management, declared, for example: (i) the right of peoples and nations to permanent sovereignty over their natural wealth and resources; (ii) the exploration of such resources, should be in conformity with the rules and conditions which the peoples and nations freely consider to be necessary or desirable with regard to the authorization, restriction or prohibition of such activities; (iii) the free and beneficial exercise of the sovereignty of peoples and nations over their natural resources must be furthered liy the mutual respect of States based on their sovereign equality; and (iv) violation of tlie rights of peoples and nations to sovereignty over their natural wealth and resources is contrary to the spirit and principles of the Charter of the United Nations and hinders the development of international cooperation and the maintenance of peace. Therefore, right to free access to natural resources is guaranteed from a moral and legal point of view. However, given the individual human nature of consumption and profit, free access without proper control can cause a race for the resources, which many times, may result on irreversible effects for them and the ecosystem. Hardin (1968) in his essay "The Tragedy of the Commons" affirmed that "The population problem has no technical solution; it requires a fundamental extension in morality", also arguing that, in the long term, maximizing individual behavior over finite natural resources would result in total collapse, and without external intervention or control there would be no solution. Overall, the world's fisheries, specially in the developing countries, with a fragile governance, still fall into this theory.

The fishing, together with the agriculture, is one of the oldest activities ever described in humankind (Sahrhage and Lundbeck, 1992). Estimates indicate that the first evidences of fishing in the world are reported at more than 500 000 years ago, with fragments from the cichlid Tilapia, and catfish that were found with remains of *Homo habilis* and the later *Homo erectus* at Olduvai in eastern Africa (Gartside and Kirkegaard, 2009). In a totally changed world, with a growing demand for food and with an activity that in no way resembles its origins, fishery activity are always seeking a balance between exploitation and conservation.

Currently, marine resources are one of the main food sources in the planet, significantly contributing to food security and well-being of human society (Oyinlola *et al.*, 2018). Fish and fish products are among the main traded commodities in the world, with nearly 40% of the total production, reflecting the sector's growing degree of integration in the global economy (Bellmann *et al.*, 2016). Accelerated human population growth implies an increase of the global food demand, consequently intensifying the search for more effective methods of production, often unsustainable. Over time, the increasing presence of the ice, diesel-powered vessels, synthetic fiber, GPS (Global Position System), sonar and radar incorporated into the fishery process, greatly contributed to the increase of the fishing power and hence the effectiveness of this activity. The fishing gears evolved to cover large areas of the bottom, driving small to large fish's shoals into the nets (Watson and Tidd, 2018).

Trawling fishery and its effects

Bottom trawling corresponds to nearly 25% of global catches (Watson and Tidd, 2018), with a continuous increase since 1950 (Watson *et al.*, 2006). Bottom trawling targets mainly fish, crustaceans, and bivalves living in, on, or above the seabed (Bensch *et al.*, 2009). It has also large adverse implications to marine habitats given its high levels of non-targeted catches (Figure 1), affecting (i) the prey availability for demersal fishes, potentially leading to reduced food intake and body condition of fish (Johnson *et al.*, 2015), (ii) the trophic structure (Ramalho *et al.*, 2018) and (iii) the yield of the captures in chronically trawled areas (Collie *et al.*, 2017). Bottom trawling also strongly modifies the substrate and benthic communities (Halpern *et al.*, 2008; Ortega *et al.*, 2018a), negatively affecting the seabed biota (Hiddink *et al.*, 2017).

Bycatch may be defined as the retained catch of non-targeted but commercially valuable species, or species consumed by crew and local communities (small-scale fisheries), used for bait (industrial fisheries), or rejected at port or at sea (Davies *et al.*, 2009; Gilman *et al.*, 2014). The development of global fisheries has also resulted by an increase in bycatch. Estimates derived through catch reconstructions from 1950 to 2010 indicated that up to 2000, levels of discard ranged between 10% and 20% of the total reconstructed catches, with a peak of 19 million tons in 1989 (Pauly and Zeller, 2016). Bottom trawls, one of the most common fishing gear worldwide, produce the highest level of bycatch and discards when compared to other fishing gears (Zeller *et al.*, 2017). Studies conducted by FAO (Food and Agriculture Organization of the United Nations) in the early 1990s at 2000s recorded discards of nearly 7.2 million tons produced by the shrimp and demersal finfish trawl fisheries in the world (Kelleher, 2005).

Catches of shrimps, lobsters and crabs catches, reached a new record high in 2018, with more than 5 million tons landed, of which 34% (2.1 million tons) were shrimps alone (FAO, 2020a). Shrimp trawl fisheries, specially the tropical ones, is the greatest source of global discards, accounting for 27.3% (1.86 million tons) of estimated total discards between 1990 and 2001 (Kelleher, 2005). No updated estimates of the levels of discards in the global shrimp fishery is available and the current scenario is basically unknown.

Most of the shrimp are caught by large industrial trawling fishing operations, but some small-scale shrimp fisheries (Figure 5), including non-motorized boats (Gillett, 2008), mainly operating in estuaries and coastal waters, play a great role for traditional communities (Gillett, 2008), contribute little to global discards (Zeller *et al.*, 2017). Small-scale fishery provides, to millions of persons, an important source of income, employment and food, being considered one of the main economic activities in coastal communities worldwide (Chollett *et al.*, 2014). Currently, in many countries, this sector faces social difficulties (Figure 5), such as the lack of alternative occupations for fishermen (Cinner *et al.*, 2009), inadequate technical and financial support and weak governance (de Oliveira Leis *et al.*, 2019). In addition, it confronts with environmental problems (Figure 5), such as pollution (Marín and Berkes, 2010), habitat degradation (Rogers *et al.*, 2018) and the collapse of fish stocks (Plank *et al.*, 2017). In developing countries (e.g., some Latin American nations, Brazil included), the ineffective implementation of public policies on small-scale fishery may have serious economic consequences for the sector and, consequently, to be a barrier to sustainable management (Mattos and Wojciechowski, 2019; Jimenez et al., 2020).

Brazilian shrimp fishing

In Brazil, shrimps are exploited in multispecies fisheries along the entire coastline, mainly in shallow areas with motorized bottom trawl nets (Costa *et al.*, 2007), Penaeidae being the main target (Lopes, 2008). Three fishery systems, which differ in size, technology and volume, occured along the Brazilian coast (Figure 1): (i) the industrial fleet operating mainly in the North region (mouth Amazon River), Southeast and South Brazil, from Rio de Janeiro to Rio Grande do Sul; (ii) a semi-industrial fleet with an intermediate technology and fishing power, and (iii) artisanal, operating along the entire coast, characterized by the high number of people involved; low level of technology, capture and profit (Dias-Neto, 2011). The industrial shrimp fisheries in the Southern Brazil have high bycatch rates, being effectively rejected and discarded at sea: 5.5 to 10.5 kg of discards to 1 kg of landed shrimp (Vianna and Almeida, 2005), while in the North, 2.2 to 11 kg of discards to 1 kg of landed shrimp have been recorded (Paiva *et al.*, 2009). The bycatch rates of the small-scale shrimp fishery in Northeast are estimated as 1 to 5 fish per 1 kg of shrimp caught, the majority being consumed or commercialized locally (Silva Júnior *et al.*, 2019).

The shrimp fisheries in Northeast is basically composed of artisanal fleet reaching a total of 16,146 tons in 2008 according to last Brazilian official fishery reliable statistics (IBAMA, 2008), representing 9.4% of the total caught in the country. It is estimated that this activity have, alone, more than 100,000 of persons envolved, 1,700 motorized and 20,000 non-motorized operating boats (Santos, 2010). Pernambuco has the fifth larger capture of shrimps in the Northeast, being the only state with no fishery policy available for this modality. Sirinhaém (case of study) has the largest and most productive motorized fishing fleet among the coastal cities of Pernambuco, corresponding to 50% of the shrimp production (Tischer and Santos, 2003) and represents a crucial source of income for the local population (Lira *et al.*, 2010).

In Brazil, the main regulations available to the shrimp fishery involve limitations of fishing licenses, closed season and mesh size regulation(Santos, 2010; Dias-Neto, 2011). Species of the bycatch are barely considered into the regulations. Minimize the global discard rate and maintain to current capture sustainably is a great challenge, mainly for developing countries due to growing demands for food security and human nutritional health (Golden *et al.*, 2016). Initiatives in course, such as international project *Sustainable Management of Bycatch in Latin America and Caribbean Trawl Fisheries* (REBYC-II LAC - http://www.fao.org/in-action/rebyc-2/overview/en/) of FAO with 4 pilot sites along of Brazilian coast (e.g. Pará, Pernambuco, Paraná/Santa Catarina and Rio Grande do Sul), are fundamentals in the process of encouraging effective management of bycatch through improved information, participatory approaches and appropriate incentives.

Hence, initiatives to minimize the catch of non-target species must be preceded or accompanied by approaches that integrate the maximum amount of information possible in order to identify the species likely to be most affected by fishing, taking into account the effects of environmental variations and policy measures. These information are essential to achieve sustainable development and ecosystem-based management, providing decision-support for proper management (Bellido *et al.*, 2011; James *et al.*, 2018).

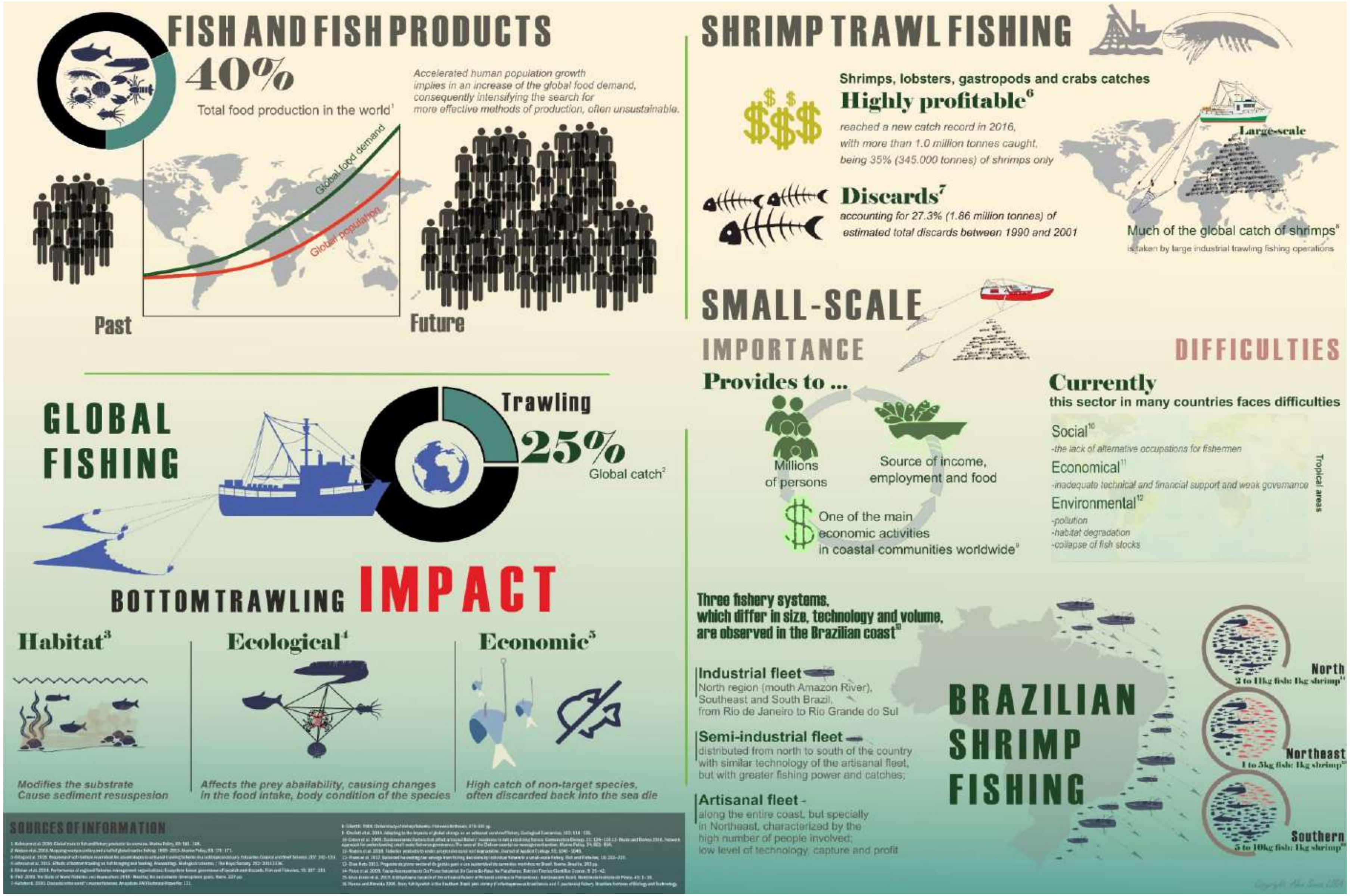


Figure 1. Summary of the impacts, importance, difficulties and problems of the shrimp bottom trawl fishery, focusing on artisanal fisheries in Brazil.

Ecosystem Approaches

The Code of Conduct for Responsible Fisheries (FAO, 1995) recommends that the entire catch, not only the targeted species, should be managed in an ecologically sustainable manner, based on principles of adaptive co-management and Ecosystem-Based Fishery Management (EBFM) or Ecosystem Approach to Fishery (EAF). Globally, the ecosystemic approach have been successfully applied in the United States- Townsend et al. (2019); Baltic sea- Möllmann et al. (2014); Australia- Smith et al. (2007); Canada- O'Boyle and Jamieson (2006); New Zealand- Reid and Rout (2020); Mexico- Arnott et al. (2012); South African- Shannon et al. (2004); Southern Brazil- Scherer and Asmus (2016). These approaches are an effective framework for ecosystem management that considers "the knowledge and uncertainties about biotic, abiotic, and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries" (Garcia *et al.*, 2003). However, implementation of EBFMs requires additional information on the dynamic of ecosystem, fishery, economy, ecology and biology of the target and no-target species (Brodziak and Link, 2002; Pikitch *et al.*, 2004; Babcock *et al.*, 2005; Kroetz *et al.*, 2019; Lidström and Johnson, 2020). Although the EAF or EBFM are extremely effective, they are rarely applied in developing countries, where the information about artisanal fishing is scarce or poorly informative, hampering the proper management of these fisheries.

Multiple studies or methods may be considered in the context of EAF and EBFM to provide a straightforward set of decision parameters to small-scale fisheries managers to fulfil both fisheries and conservation management. Taking into account the functional role and relationships between species is crucial in EAF and EBFM, especially for communities affected by non-selective fisheries such as bottom trawling, where cumulative effects can lead to intense changes in the trophic structure of the ecosystem, affecting the food web (cascade effect) and thus the sustainability of fisheries. Several methods are used to study the trophic structure of ecosystems, such as stomach content analysis or natural trophic markers (carbon and nitrogen stable isotopes), in order to characterize trophic interactions, complexity and connectivity between ecosystems (e.g., rivers, estuaries, reefs and deep oceans) (Ferreira *et al.*, 2004; Noble *et al.*, 2007; Choy *et al.*, 2017; Barik *et al.*, 2019; Hayden *et al.*, 2019). Moreover, a better understanding of trophic interactions can be obtained through models that integrate multiple aspects of the ecosystem, such as Ecopath with Ecosim (EwE) (Wolff *et al.*, 2000; Christensen and Walters, 2004). These ecological models have been widely applied to characterize trophic interactions and changes at the scale of biological communities (Lira *et al.*, 2018; Zhang *et al.*, 2019) as well as to assess the effect of management policies on the environment and ecological compensation (Hattab *et al.*, 2013; Halouani *et al.*, 2016a; Vasslides *et al.*, 2017).

In addition, other models are used when catches or biological data are incomplete, aggregated across species or are insufficient to perform a quantitative stock assessment (Lucena-Frédou *et al.*, 2017), as for many tropical fisheries. For example, the semi-quantitative risk analysis – PSA (Productivity and Susceptibility Analysis), based on the relationship between the biological productivity (Stobutzki *et al.*, 2001; Hobday *et al.*, 2007) and the susceptibility to fishing (Patrick *et al.*, 2010; Lucena-Frédou *et al.*, 2017) allows estimating the level of vulnerability of both target species and bycatch. This approach can be extremely useful in identifying species that should be prioritized for research, management and conservation.

For the Brazilian coastal fisheries, particularly the small-scale, given the lack of continuous monitoring of landings and effort, associated to the absence of studies that aggregately evaluate environmental, fishing, biological and ecological aspects of the activity, the initiatives of management based in ecosystem approach are scarce. This is clearly observed in the Brazilian shrimp fisheries since, given the lack of assessment, current fishery regulation are restricted, when available, to target species, without accounting the non-target species or the ecosystem as a whole (Gillett, 2008; Santos, 2010; Dias-Neto, 2011).

Case study - small-scale shrimp fishing in Sirinahem, Northeast Brazil

In the Southwestern Atlantic, Barra of Sirinhaém (BSIR), located on the southern coast of Pernambuco, in Northeast Brazil, is mainly influenced by nutrient supply of the Sirinhaém river and the multiple tributaries (Arrumador, Trapiche, Aquirá) (Figure 2).

Figure 2. The coast of Sirinhaém with Landsant-8 and Sentinel-2 satellite images and the island of Santo Aleixo, south of Pernambuco, Northeast Brazil.

The region is characterized by a tropical climate, with precipitation ranging from 20 to 450 mm·month^{-1} and rainy season between May and October. The mean surface water temperature is 29°C, pH and salinity vary between 8.0 and 8.7 and 23 and 37, respectively (Mello, 2009; APAC, 2015). Fishing, sugar cane industry and other farming industries are considered the main activities in the area (CPRH, 2011). The fishing zones are inside or close to the Marine Protected Areas around Santo Aleixo Island (MPAS of Guadalupe and Costa dos Corais) (Figure 3). Fleet operates from 1.5 to 3.0 miles off the coast, mainly between 10 and 20 m depth. Hauls last from 4 to 8 hours and boat velocity vary between 2 and 4 knots. Boats often have 8-10 m of length, horizontal opening net of 6.1 m, mesh sizes of body and cod end of 30 mm and 25 mm, respectively.

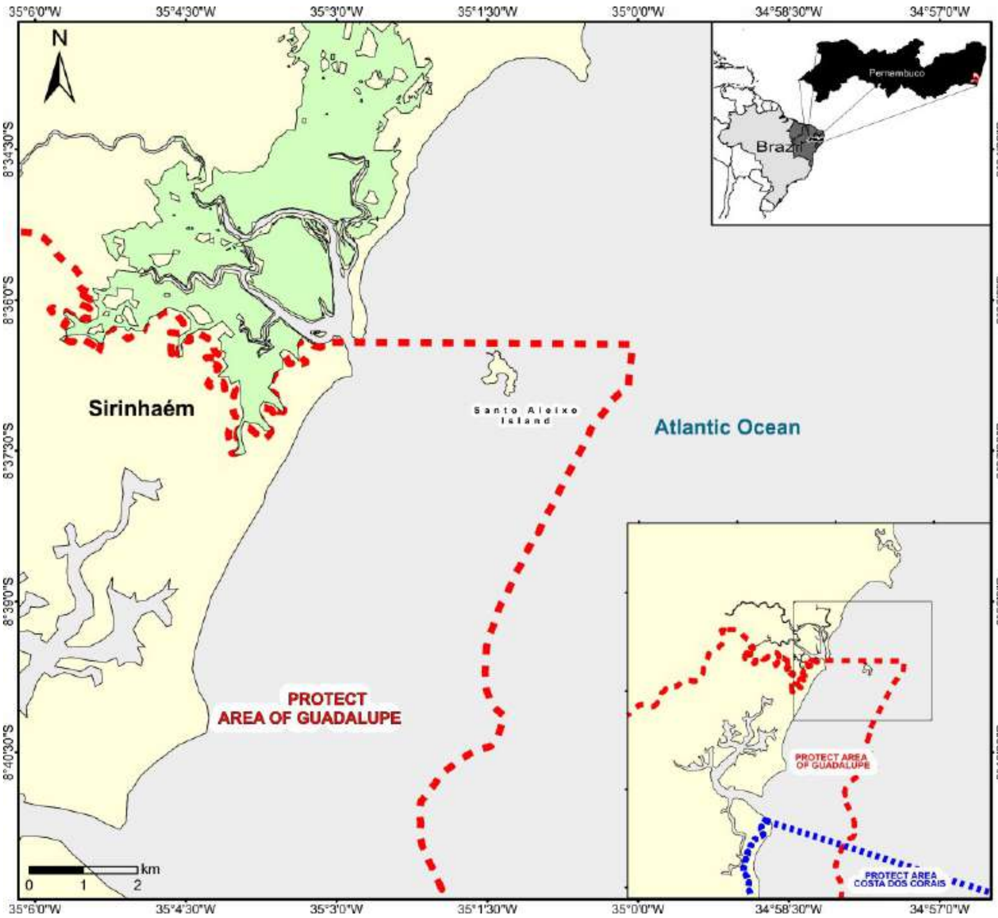


Figure 3. Study area, description of fishing methods and composition of the bottom trawl catch in Barra de Sirinhaém (BSIR), south of Pernambuco, Northeast Brazil.

STRUCTURE AND OBJECTIVES

The main aim of this thesis is to assess the current framework and potential future impact of fishing and environmental changes on the Sirinhaém coastal ecosystem, as a study case for a small-scale shrimp trawling in Northeastern Brazil. We propose to study the abiotic and biotic dimensions of the environment, and to analyze the effects of fishing on several levels of biological organization, from target to non-target, from individual to population and ecosystem (Figure 1). This will allow to answer the following questions:

- What are the dynamics of the small-scale shrimp fishery in Sirinhaem, Northeast Brazil?
- Do environmental changes have a crucial role in the dynamics of the fishery?
- Which species are most affected by trawling, to what extent are they threatened and why?
- What are the priority species for data collection and regulation in this fishery? Target catch only or bycatch (and which) also?
- Are fishery management measures really needed in the region? How effective would they be?
- What are the lessons that could be learned and replicated for other tropical multispecies fisheries with limited data status? Can we replicate the methods proposed here?

To achieve this, the thesis is organized into four Chapters considering an ecosystem approach (Figure 1).

The first Chapter (Chapter 1) consists of an integrative review of the multiple abiotic and biotic dimensions related to the artisanal shrimp fishery in the southern coast of Pernambuco, in order to help decision makers to implement measures for species, gears or areas, contributing to the management of fisheries in the region, according to EAF principles.

Within this Chapter, we propose to describe the points listed below:

- Characterization of the fishing area (bathymetry and seabed structure, salinity gradient, temperature and precipitation);

- Historical catch volume;

- Summary of information on the target shrimp species;

- Characterization and destination of bycatch.

This Chapter entitled "The Ecosystem Approach to Fisheries in Action: a study case of the shrimps small-scale fishery in tropical Brazil" will be submitted to Review in Fish Biology and Fisheries. The information obtained in this study was essential for the construction of all the following Chapters.

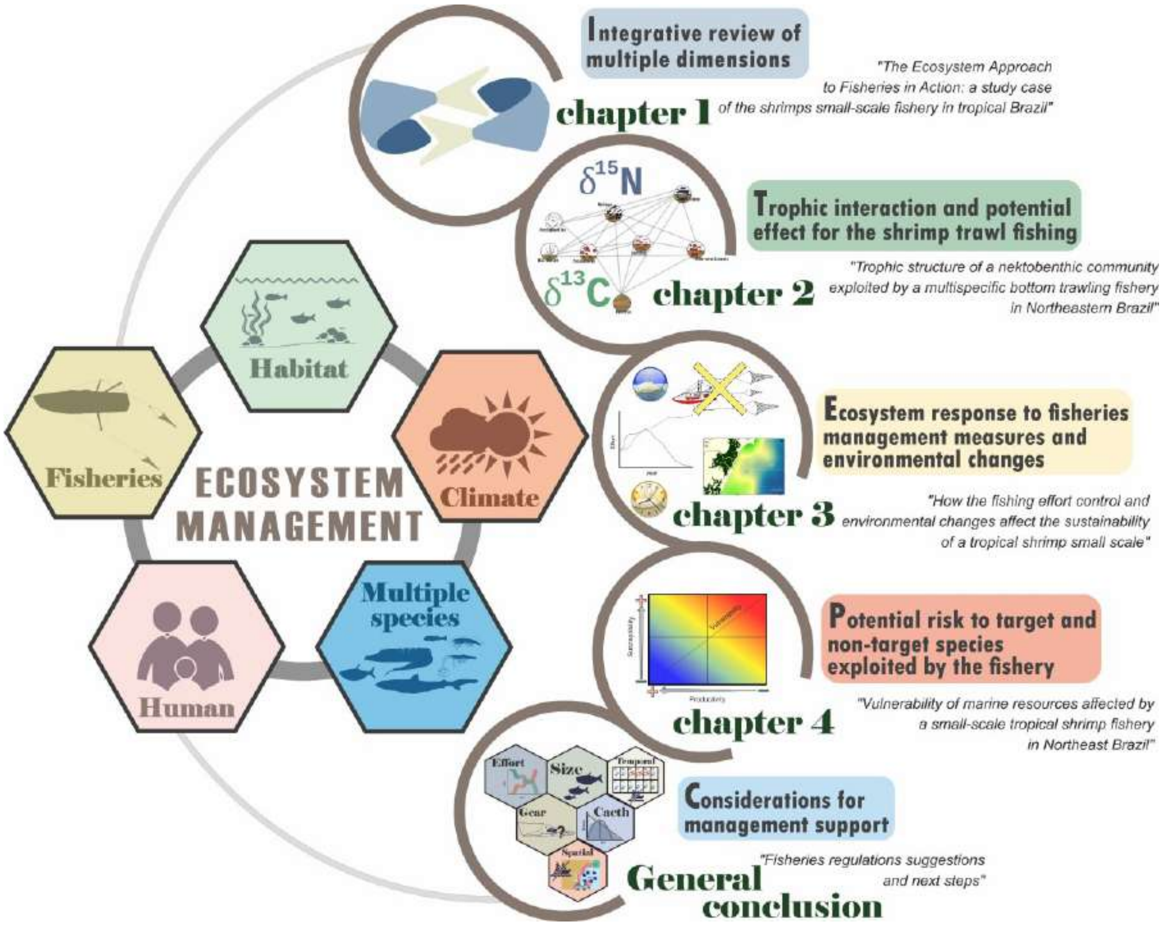


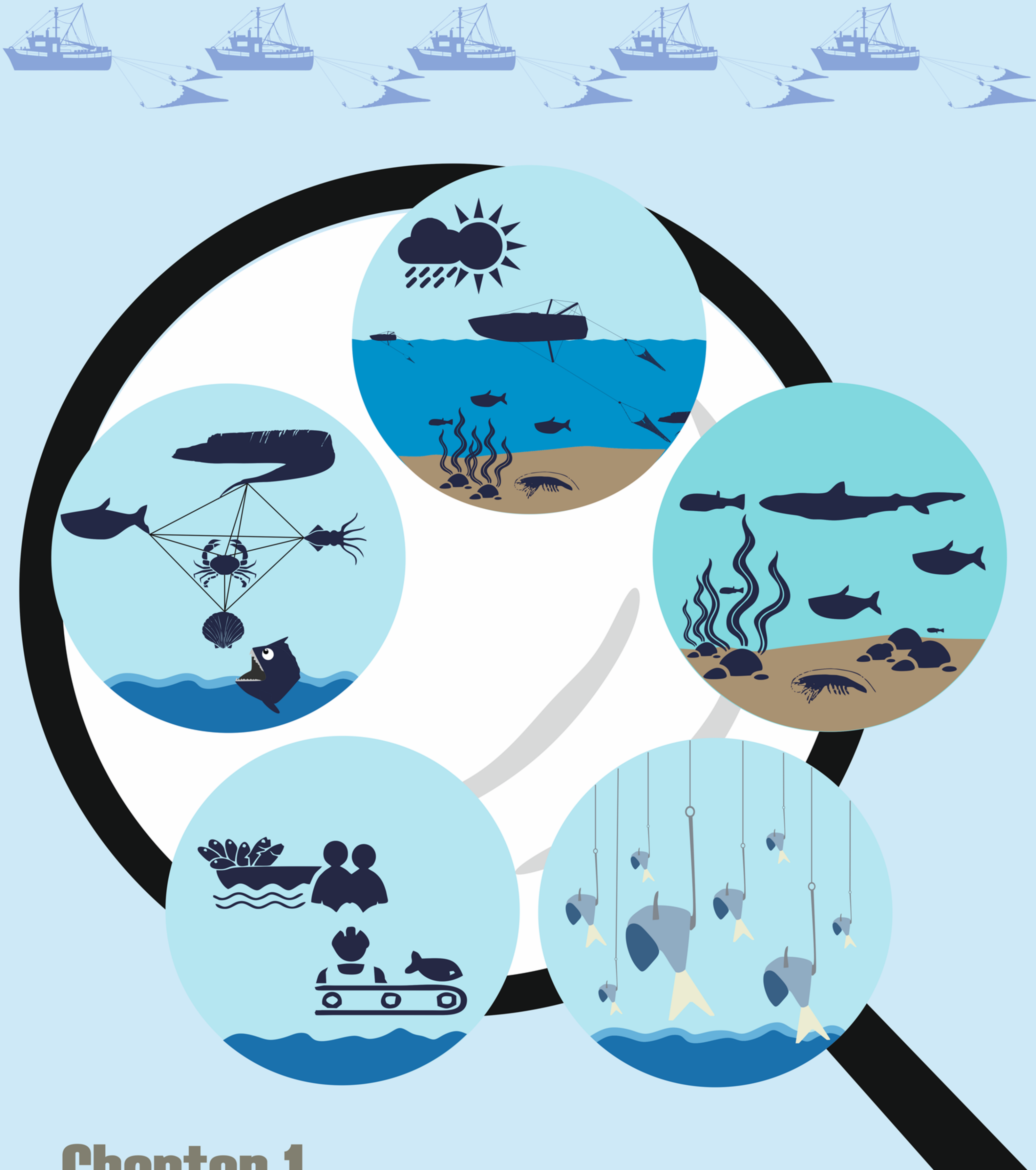
Figure 1. Summary of the Chapters and respective objectives of the thesis.

The second Chapter (Chapter 2) focuses on determining the importance of target species (shrimp) as prey for non-target species (bycatch) and discusses the possible effect of bottom trawling on trophic interactions, which may affect the local marine community and the sustainability of the fishery. For this, we studied the trophic structure of the nektobenthic community through stable isotope analyses (SIA) of carbon and nitrogen as well as stomach contents (SCA). This study was the subject of a paper entitled "*Trophic structure of a nektobenthic community exploited by a multispecific bottom trawling fishery in Northeastern Brazil* - https://doi.org/10.1371/journal.pone.0246491" published in the journal PlosOne. This information derived from the SCA and SIA analysis will be used as input and calibration data for the Ecopath with Ecosim (EwE) model developed in the next Chapter.

The third Chapter (Chapter 3) focuses on the development of the mass balance trophic model Ecopath with Ecosim to obtain a first representation of the trophic functioning of the ecosystem, by simulating the effect of environmental changes (reduction of primary productivity) and the potential effect of different management measures (closed season and effort level control). This Chapter has been published as an article entitled "*How the fishing effort control and environmental changes affect the* *sustainability of a tropical shrimp small scale fishery* - https://doi.org/10.1016/j.fishres.2020.105824" in the Fishery Research.

In the fourth and last Chapter (Chapter 4), a semi-quantitative risk analysis - PSA (Productivity and Susceptibility Analysis) adapted to the regional conditions, was applied, allowing for the estimation of a vulnerability rank and the identification of the potential risk of the shrimp fishery on the target and non-target species exploited on the Sirinhaém coast. In addition, uncertainty were incorporated into the model in order to assess the effect of subjectivity on the estimates. From our case study, we believe that this approach could be applied to the assessment of other tropical fisheries, where uncertainties and limited information hinder management and conservation actions by decision makers. This Chapter entitled "*Vulnerability of marine resources affect by tropical shrimp small scale fishery in a tropical area*" was submitted to ICES Journal of Marine Science.

Finally, we provided a general discussion with an integrated overview of the small-scale tropical trawling fishery in Sirinhaém, within the EAF or EBFM, from which recommendations and suggestions for future studies for ecosystem management arose. In addition, the challenges and future shortcomings for the fisheries policy in the region are presented in the overall conclusion.



Chapter 1

The Ecosystem Approach to Fisheries in Action: a study case of the shrimps small-scale fishery in tropical Brazil

CHAPTER 1. The Ecosystem Approach to Fisheries in Action: a study case of the shrimps small-scale fishery in tropical Brazil

Introduction

Trawling fishery and its effects

Marine resources are one of the main food sources in the planet, significantly contributing to food security and well-being of human society (Oyinlola *et al.*, 2018). Fish and fish products are among the main traded commodities in the world, with nearly 40% of the total production, reflecting the sector's growing degree of integration in the global economy (Bellmann *et al.*, 2016).

Accelerated human population growth implies an increase of the global food demand, consequently intensifying the search for more effective methods of production, often unsustainable. Over time, the increasing presence of the ice, diesel-powered vessels, synthetic fiber, GPS (Global Position System), sonar and radar incorporated into the fishery process, greatly contributed to the increase of the fishing power and hence the effectiveness of this activity. The fishing gears evolved to cover large areas of the bottom, driving small to large fish's shoals into the nets (Watson and Tidd, 2018).

Recently, studies encompassing the reconstruction of the global fishing (Zeller *et al.*, 2017; Cashion *et al.*, 2018), also including the Illegal, Unreported and Unregulated Fisheries (IUU) and discards, indicated that the purse seining and trawling fisheries are responsible for more than half of the global catches. Bottom trawling corresponds to nearly 25% of global catches (Watson and Tidd, 2018), with a continuous increase since 1950 (Watson *et al.*, 2006). Bottom trawling targets mainly fish, crustaceans, and bivalves living in, on, or above the seabed (Bensch *et al.*, 2009). It has also large adverse implications to marine habitats given its high levels of non-targeted catch, affecting (i) the prey availability for demersal fishes, potentially leading to reduced food intake and body condition of fish (Johnson *et al.*, 2015), (ii) the trophic structure (Ramalho *et al.*, 2018) and (iii) the yield of the captures in chronically trawled areas (Collie *et al.*, 2017). It also strongly modifies the substrate and benthic communities (Halpern *et al.*, 2008; Ortega *et al.*, 2018a), negatively affecting the seabed biota (Hiddink *et al.*, 2017).

Bycatch may be defined as the retained catch of non-targeted but commercially valuable species, or species consumed by crew and local communities (small-scale fisheries), used for bait (industrial fisheries), or rejected at port or at sea (Davies *et al.*, 2009; Gilman *et al.*, 2014). The increase of the global fisheries along time has also resulted in a raising, at the same rate, of the bycatch, including those discarded (Pauly and Zeller, 2016). Bottom trawls, one of the most common fishing gear worldwide, produce the highest level of bycatch and discards when compared to other fishing gears (Zeller *et al.*, 2017). Estimates derived through catch reconstructions from 1950 to 2010 indicated that, up to 2000, levels of discard ranged between 10% and 20% of the total catches (Pauly and Zeller, 2016). Studies conducted by FAO (Food and Agriculture Organization of the United Nations) in the early 1990s to 2000s recorded discards of nearly 7.2 million tons produced by the shrimp and demersal finfish trawl fisheries in the world (Kelleher, 2005). Shrimp trawl fisheries, specially the tropical ones, is the greatest source of global discards, accounting for 27.3% (1.86 million tons) of estimated total discards between 1990 and 2001 (Kelleher, 2005). No updated estimates of the levels of discards in the global shrimp fishery is available and the current scenario is basically unknown.

Catches of shrimps, lobsters and crabs catches, reached a new record high in 2018, with more than 5,000 million tons landed, of which 35% (2,115 million tons) were shrimps alone (FAO, 2020a). Most of the shrimp are caught by large industrial trawling fishing operations, but some small-scale shrimp fisheries fishing, including non-motorized boats (Gillett, 2008), mainly operating in estuaries and coastal waters, play a great role for traditional communities (Gillett, 2008), contribute little to global discards (Zeller *et al.*, 2017). Small-scale fishery provides, to millions of persons, an important source of income, employment and food, being considered one of the main economic activities in coastal communities worldwide (Chollett *et al.*, 2014). Currently, in many countries, this sector faces social difficulties, such as the lack of alternative occupations for fishermen (Cinner *et al.*, 2009), inadequate technical and financial support and weak governance (de Oliveira Leis *et al.*, 2019). In addition, it confronts with environmental problems, such as pollution (Marín and Berkes, 2010), habitat degradation (Rogers *et al.*, 2018) and the collapse of fish stocks (Plank *et al.*, 2017). In developing countries (e.g. some Latin American nations, Brazil included), the ineffective implementation of public policies on small-scale fishery may have serious economic consequences for the sector and, as a result, this lack of sustainability and institutional weakness can obstruct the implementation of public policies to enforce more sustainable management measures (Mattos and Wojciechowski, 2019; Jimenez *et al.*, 2020).

Brazilian shrimp fishing

In Brazil, shrimps are exploited in multispecies fisheries along the entire coastline, mainly in shallow areas with motorized bottom trawl nets (Costa *et al.*, 2007), Penaeidae being the main target (Lopes, 2008). Three fishery systems, which differ in size, technology and volume, are observed along the Brazilian coast: (i) the industrial fleet operating mainly in the North region (mouth Amazon River), Southeast and South Brazil, from Rio de Janeiro to Rio Grande do Sul; (ii) a semi-industrial fleet distributed from north to south of the country with similar technology of the artisanal fleet but with greater fishing power and catches; and (iii) artisanal fleet that operates along the entire coast, but specially in Northeast, characterized by the high number of people involved; low level of technology, capture and profit (Dias-Neto, 2011). The industrial shrimp fisheries in the Southern Brazil have high bycatch rates, bycatches being effectively rejected and discarded at sea: 5.5 to 10.5 kg of discards to 1 kg of landed shrimp (Vianna and Almeida, 2005), while in the North, 2.2 to 11 kg of discards to 1 kg of landed shrimp have been recorded (Paiva *et al.*, 2009). The bycatch rates of the small-scale shrimp fishery in Northeast are estimated as 1 to 5 fish per 1 kg of shrimp caught, bycatch being to majority consumed or commercialized locally (Silva Júnior *et al.*, 2019).

The shrimp fisheries in Northeast is basically formed by artisanal fleet and, in 2008, according to last Brazilian official fishery reliable statistics (IBAMA, 2008), this sector represented 9.4% of the total caught in the country. It is estimated that this activity alone employs more than 100,000 persons, 1,700 motorized and 20,000 non-motorized boats in Northeast, and particularly in Pernambuco state, where it is considered very important on the socio-economic level (Santos, 2010). However, Pernambuco is the only state where the artisanal shrimp fishery has currently no regulation. Sirinhaém (case of study) has the largest and most productive motorized fishing fleet among the coastal cities of Pernambuco, corresponding to 50% of the shrimp production (Tischer and Santos, 2003), being extremely important as source of income for local population (Lira *et al.*, 2010).

In Brazil, in general shrimp fishery regulation involves limited fishing licenses, closed season and mesh regulation, while for the industrial fleet, the Turtle Excluder Device (TED) is the only compulsory measure considered for the bycatch (Santos, 2010; Dias-Neto, 2011). Minimizing global discard and maintaining sustainable captures are great challenges, mainly for developing countries due to the growing demands for food security and human nutritional health (Golden *et al.*, 2016). Strategies based on principles of adaptive co-management and Ecosystem Approach to Fisheries (EAF) (Guanais *et al.*, 2015) have proved to be very promising in recent years (Serafini *et al.*, 2017). The EAF is an effective framework for ecosystem management that considers "the knowledge and uncertainties about biotic, abiotic, and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries" (Garcia *et al.*, 2003). EAF-based policies hold great promise for understanding and potentially mitigating the impacts of trawling. They have being applied in different countries (Jennings and Rice, 2011), fisheries (Gianelli *et al.*, 2018), resources (Cuervo-Sánchez *et al.*, 2018) and environments (Rosa *et al.*, 2014). The Code of Conduct for Responsible Fisheries (FAO, 1995) recommends that the entire catch, not only the targeted species, should be managed in an ecologically sustainable manner. Moreover, understanding how the biotic component relates to abiotic conditions (e.g., physical, economic, or social) in terms of dynamics and functioning is crucial to achieving the EAF goal.

Although information concerning the shrimp fishing in Northeast of the Brazil, specifically in Sirinhaém is available, it remains very stratified (Tischer and Santos, 2003; Lopes *et al.*, 2014, 2017; Silva *et al.*, 2015, 2016, 2018; Silva Júnior *et al.*, 2015; Peixoto *et al.*, 2018), and do not considers the interactions between the basic elements of the fishery (fish and fishers), habitats and environmental conditions as recommend by the Ecosystem Approach to Fisheries (Garcia et al., 2003). This lack of integrated information is considered as one of the main obstacles to the use of management strategies for the maintenance of sustainable fishing, species conservation and the environmental preservation (Medeiros *et al.*, 2013; Prestrelo and Vianna, 2016).

In present study, we propose an overview of the small-scale shrimp fisheries in Sirinhaém, as a case study for Northeast of Brazil, describing: (i) the abiotic characteristics of the shrimp fishing sites; (ii) the temporal evolution of the catches for the main target species (shrimps); (iii) the main aspects of the population dynamics and stock assessment of the target shrimps and (iv) the quali-quantification description and destination of the fish bycatch. Unlike most studies that consider only part of the catch or ecosystem (Bruno *et al.*, 2013; Niella *et al.*, 2017; da Costa *et al.*, 2018; Delgado *et al.*, 2018; Dolder *et al.*, 2018), this review aims at compile, in an integrative way, the multiples dimensions of the shrimps small-scale fishery, which may contribute to decision makers, to the establishment of regulations (related to gear, species and space), contributing to the fishery management in the region, according to the principles of the EAF.

Material and methods

Study area

Sirinhaém, located in Southern coast of Pernambuco, Northeast of Brazil (Figure 1), is influenced mainly by nutrient supply of the Sirinhaém river. The climate is tropical, with a rainy season occurring between May and October. The rainfall ranges from 20 to 450 mm·yr^{-1}, the mean water temperature is 29°C, the pH and salinity range between 8 and 8.7 and 23–37, respectively (Mello, 2009; APAC, 2015). Fishing, sugar cane industry and other farming industries are considered to main productivity activities in the region (CPRH, 2011). The coast is located within (Marine Protected Area of Guadalupe) or near (Marine Protected Area Costa dos Corais) important Marine Protected Areas, around of Santo Aleixo Island (Figure 1).

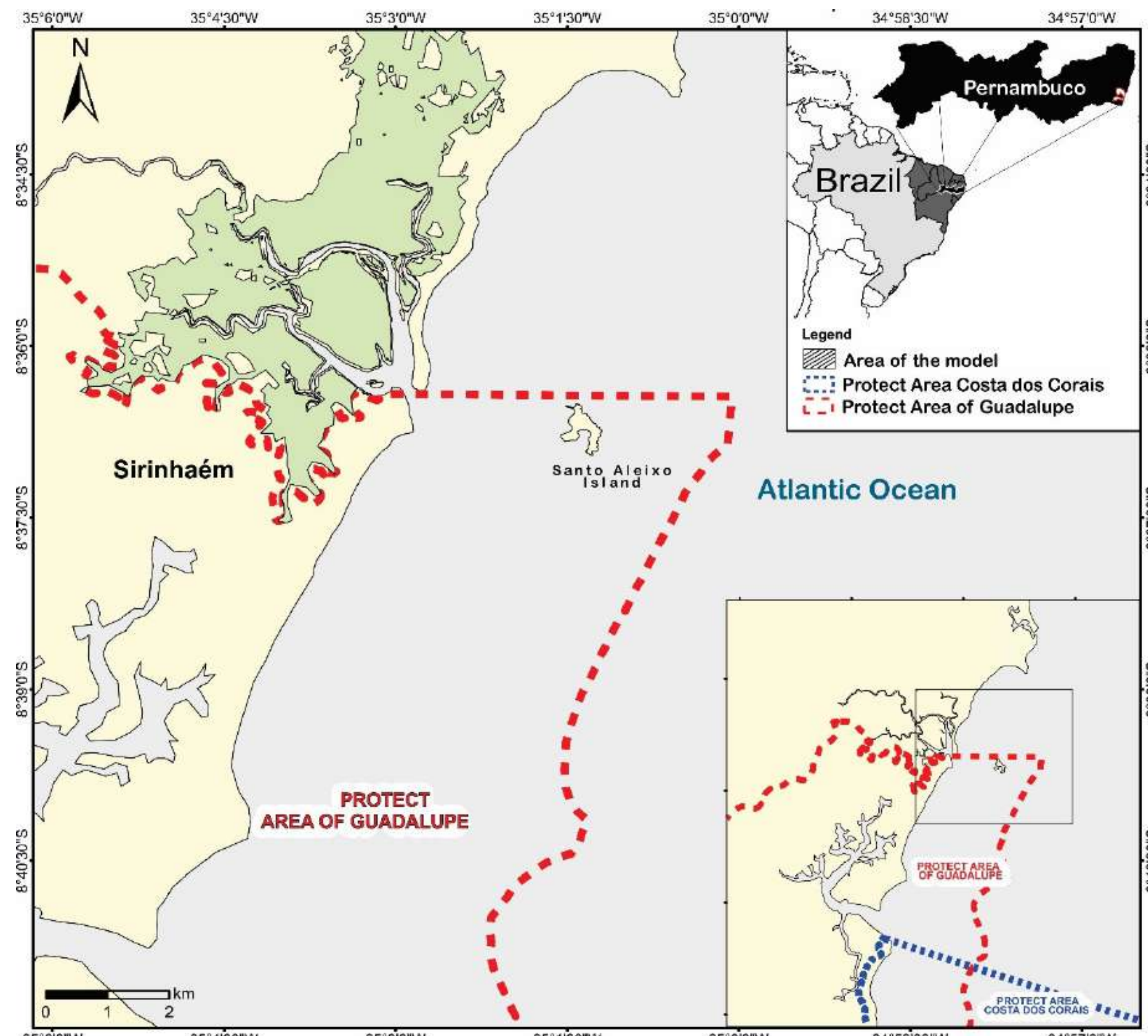


Figure 1. Study area in the coast of state of Pernambuco, Northeastern Brazil.

Data sources

In order to characterize the shrimp trawling fishery in Sirinhaém, information about the fishery (landings, catch composition, fishery zones, bycatch destination), and the abiotic (Morpho-Sedimentary Facies, the bathymetry, Chlorophyll a concentration, the pluviometry) and biotic (life history traits of the target and non-target species) compartments were collected. This data collection was based on (a) primary and (b) secondary sources, obtained from literature, reports and official governmental data (Figure 2). The sources of each parameters are described below.

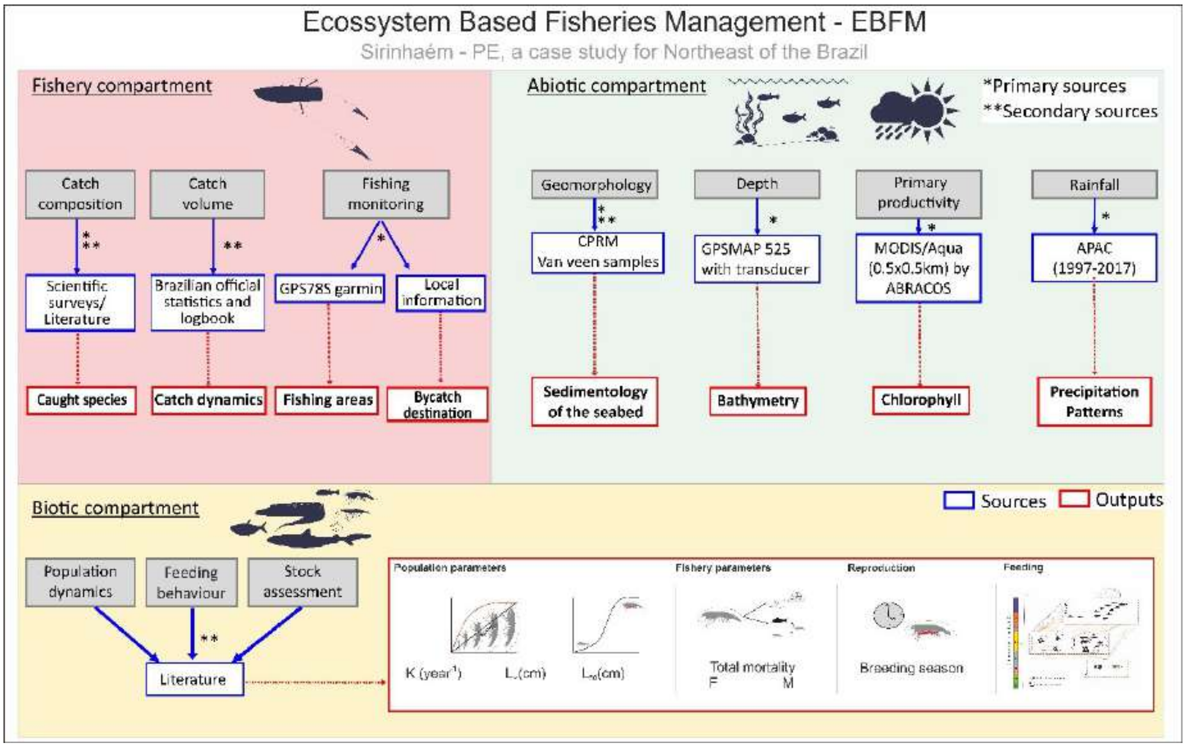


Figure 2. Compiled or estimated information according by Ecossystem Based Fisheries Management (EBFM), in terms of environmental, fishing, biological and ecological features on Sirinhaém-PE coast, a case study for small-scale shrimp fishing in the Northeastern Brazil.

We used as primary sources, bathymetry profiles defined by depth tracks carried out monthly in the coast zone of Sirinhaém from May 2017 to January 2018, with a sampling interval of 1 log per second recorded using a GPSMAP 525 Garmin with transducer set in an artisanal shrimp fishing local boat. The fishery areas were obtained monthly by the fisheries monitoring carried out between May 2017 and January 2018, where the different fishery zones were indicated by fishermen and recorded using a GPS78S Garmin. Sediment sampling from drag Seabed carried out in December 2018 and July 2019 was carried out to define the seabed substract. To describe the fish bycatch composition, scientific samples were conducted monthly, between August 2011 and July 2012; and quarterly, from October 2012 to June 2014; May 2017 to January 2018, using two bottom otter trawls (10 m wide and 6.1 m deep), mesh sizes of body and cod end of 30 mm and 25 mm, respectively. For each sample, three trawling tows of two hours each were carried out. Once collected, the specimens were immediately put-on ice onboard, then transported to the laboratory and stored in a freezer (-18º C) until the analysis. Information about the bycatch destination were obtained from May 2017 to January 2018 by fisheries monitoring where the final destination was indicated by fishermen.

Considering the secondary sources, several data were used. Complementary sediment information were acquired by the Brazilian National Oceanographic Database (BNDO), based in transverse profile along of the coastline obtained with van Veen Grab (Assis, 2007). Average Chlorophyll concentrations were obtained by satellite images in two different periods, between October and November 2015 and April and May 2017. Data were acquired from Moderate Resolution Imaging Spectroradiometer on the Aqua satellite (MODIS/Aqua); (grid resolution: 0.5 × 0.5km) by ABRACOS Project (Bertrand, 2015, 2017). Pluviometry data were obtained monthly, from 1993 to 2017, from Agência Pernambucana de Águas e Climas- APAC. The fishery official statistics were based on the official fishery statistics bulletins published between 1988 and 2007 by IBGE “Instituto Brasileiro de Geografia e Estatística” (1988-1989) and IBAMA “Instituto Brasileiro do Meio Ambiente” (1990-2007). This source does not discriminate the species, thus, complementary, logbooks, discriminated by shrimp species, were obtained with vessel owners and intermediaries of the shrimp fishery from 2009 to 2014. Finally, we compiled information obtained from published literature about catch composition, status of the stock and biological traits, including life history, reproduction and feeding habitats of the shrimps and bycatch species caught in the study area. See Table S1 and S2 for more details.

Data Analysis

Fishery and abiotic compartment

Bathymetry and Chlorophyll *a* - The geostatistical interpolation method (Universal Kriging – UK) was used to estimates the depth values and Chlorophyll *a* at unsampled points (Curtarelli et al., 2015). Kriging interpolation is a geostatistical method widely used in environmental sciences, mainly to describe spatial patterns and interpolate the values of the primary variable at unsampled locations (Cigagna *et al.*, 2015; Amiri *et al.*, 2017; Du *et al.*, 2018). Specifically, UK have certain advantages over other interpolation methods as (i) it does not require knowledge nor stationarity of the mean over the region of interest; (ii) it allows to account for local variation of the mean and (iii) estimates better follow the variation of the data, changing proportionally with the local data averages (Goovaerts, 1997; Li and Heap, 2014).The semivariogram was adjusted interactively up to better fitting (Curtarelli *et al.*, 2015). The assessment of the model was defined based on the cross-validation results (Li and Heap, 2014), following the determination coefficient (R^2) and the root mean square error (RMSE) and average standard error (ASE) (Goovaerts, 1997).

Fishery area – Points of the different fishery zones classified by fishers was represented trough Voronoi diagram. Voronoi polygons are generated by partitioning the sampling points into convex polygons holding only one of the original data points. Thus, any observation within of boundaries of these polygon is considered more closer to of this point than any other sample point (Longley *et al.*, 2010). To evaluate the spatial distribution of the fishery, the fishing spots were represented by grid plot (resolution: 0.5 x 0.5 km).

Sedimentology of the seabed –seabed substrate was classified, according to Folk (1954), in the four general textural classes: mud, sand, sandy gravel, and gravel. The sediment data was interpolated using natural neighbor method to generate a thematic map to represents the textural seabed map according Lucatelli et al. (2020).

Precipitation pattern - Decomposition procedure was used in the rainfall time series using a classical additive model (Hyndman and Athanasopoulos, 2017) to extract the season component.

$$\text{Eq.1 Additive model: } Y_t = \text{Trend} + \text{Seasonal} + \text{Random}$$

The additive model describes the trend and seasonal factors in a series, indicating the months with great probability of rainfall peak (Rainy season) and minimum (Dry season) to each year (Issahaku *et al.*, 2016). This model is useful when the seasonal variation is relatively constant over time (Hyndman and Athanasopoulos, 2017).

Catch seasonality - To investigate the monthly catch pattern, the shrimp fishery data were grouped by months and related with rainfall through Pearson's correlation coefficient (r).

Bycatch destination - We evaluated the proportion of use or trade destination into three categories: Discarded (Di), Consumed (Co) or Commercialized (Cm).

Biotic compartment

Fish bycatch composition - In laboratory, the species caught were identified based on the specific taxonomic keys (Carpenter, 2002a, 2002b, 2002c) and then counted, measured (Total and standard length) and weighed. Species composition was described by taxonomic hierarchy based on Nelson et al. (2016) and the frequency estimated by constancy of occurrence index (IC) (Dajoz, 1983), which classified the species as constant (present in more than 50% of samples), accessory (present in 25%–50% of samples), and occasional (< 25%).We also classified the species according to the IUCN Red List categories at the regional level (ICMbio, 2018), which comprises 10 levels: Extinct (EX), Regionally Extinct (RE), Extinct in the Wild (EW), Critically Endangered (CR), Endangered (EN), Vulnerable (VU), Near Threatened (NT), Least Concern (LC), Data Deficient (DD) and Not Applicated (NA). The classification criteria, application guidelines, and IUCN Red List methodology on how to apply the Criteria are publically available (IUCN, 2000, 2012).

Biological traits - To evaluate the population patterns, we considered different biological traits: asymptotic total length ($L_{\infty;}$ cm), length at first maturity ($L_{50;}$ cm) and growth coefficient (*k*; year^{-1}). To describe the feeding patterns of the fishes caught as bycatch by shrimp trawling, the stomach content items were gathered in 9 prey groups (detritus, phytoplankton, zooplankton, worm, crab, mollusk, other crustaceans, shrimp and fish) and it was graphically displayed through heatmap (consumer x prey) along with an Agglomerative Hierarchical Cluster (AHC) using prey weight proportion (%W) for each consumer according to Lira et al. (2021b). All parameters are detailed in supplementary material Table S1 and S2.

All statistics and geostatistical analyses were performed using the R environment (Core Team, 2020) and ESRI® ArcGIS™ software, respectively.

Results

Characterization of the abiotic condition of the shrimp fishing sites

Barra de Sirinhaém (BSIR) coastal zone is characterized by shallow water, ranging from 1 to 20 m depth (Figure 3). A flat bottom area is located in the first 10 meters of depth, followed by a small depression in the outer region of the Santo Aleixo Island, where the artisanal trawl fishery is carried out, mainly at depths ranging from 10 to 20 m, associated to muddy sediment (Figure 3).

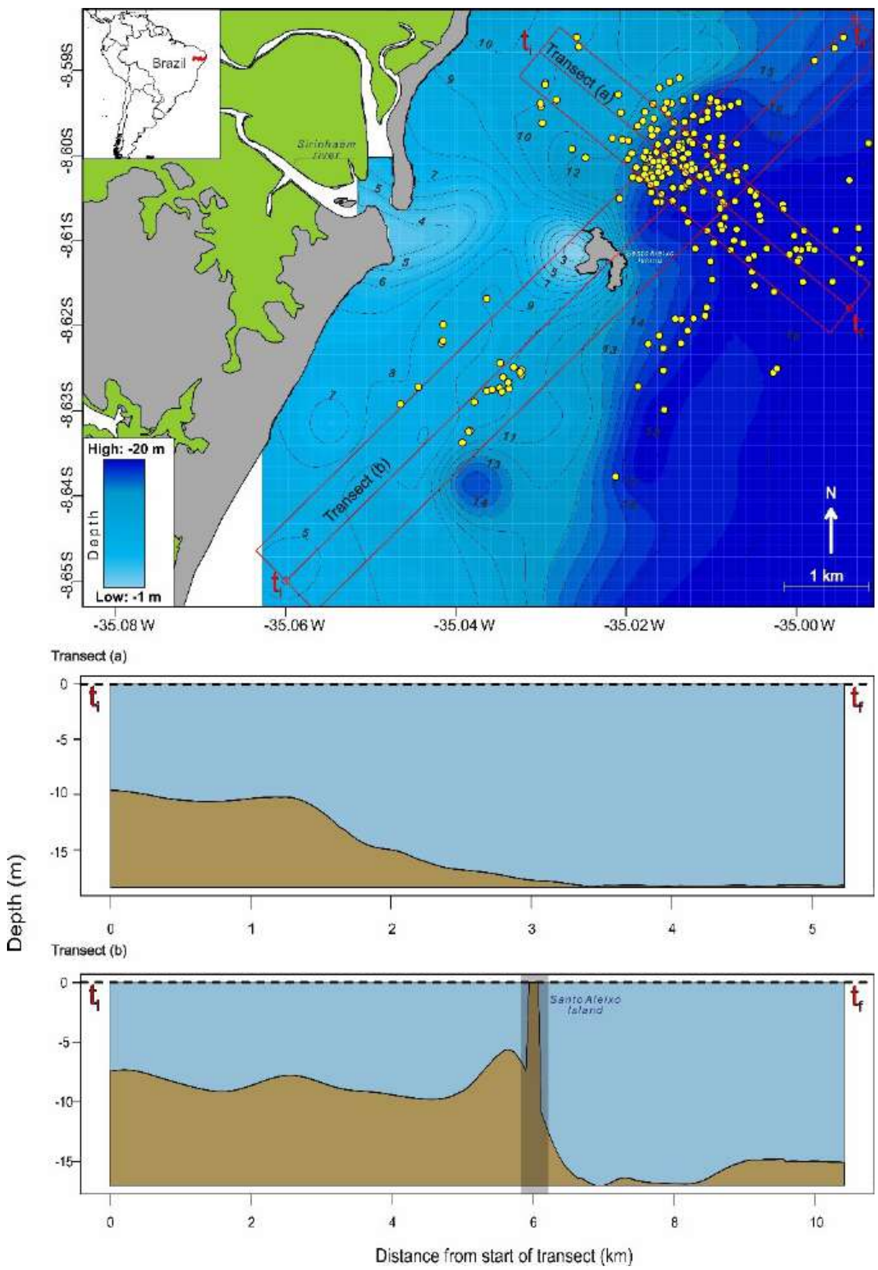


Figure 3. Bathymetry and profile of depth in Sirinhaém-PE coast, Northeastern Brazil.

According to the fishers, six fishing areas (Figure 4) were identified, with the fishing zones Baixo, Meio and Lama de Fora concentrating most of the fishing activity.

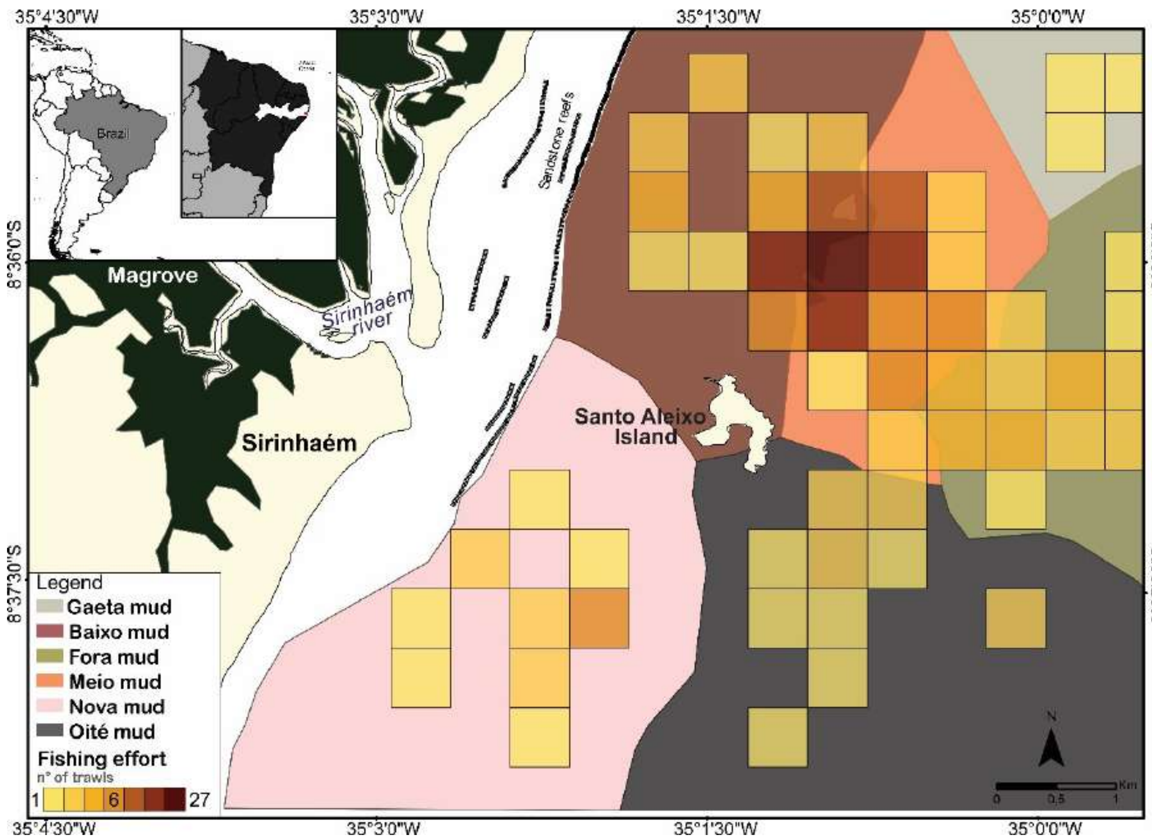


Figure 4. Mapping of the fishery zones indicated by the fishermen's in Sirinhaém-PE coast, Northeastern Brazil. Grid resolution: 0.5 x 0.5 km.

The spatial distribution of fishing effort was associated to the seabed composition. The continental shelf is mostly composed of sand and mud (Figure 5). Mud concentrated sediment is located in the southeast to the Northeast around the Santo Aleixo Island (between 10 and 20 m) with about 23 km^2. The central region of this large area is composed of extremely fine muds with a concentration of silt higher 70%, while the lowest mud proportion is located in the northwest and southeast of the area, evidencing a more sandy mud. In addition to sand, the coastal zone around the mud area contains large bank of gravel and sandy gravel (Figure 5).

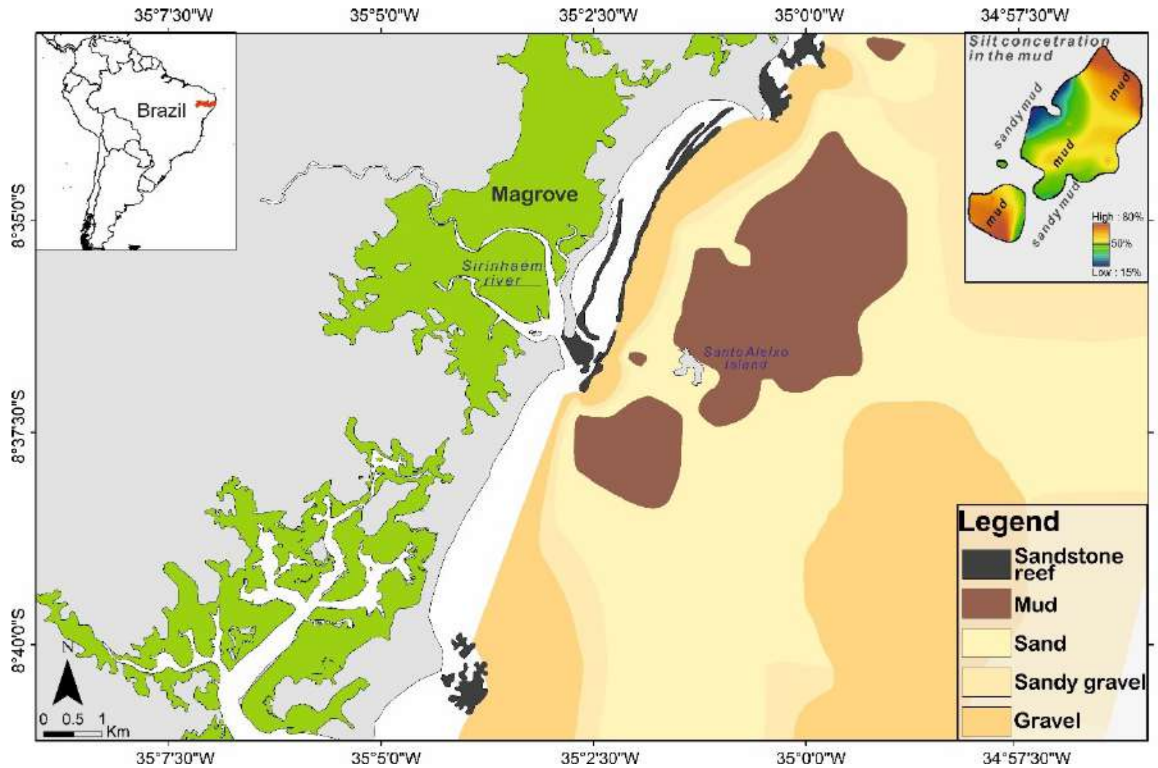


Figure 5. Seabed surface sediments (a) and carbonate concentration (b) Source: Assis et al. (2015) in Sirinhaém-PE coast, Northeastern Brazil. Points represent scientific samples.

The rainfall fluctuation over time (1993-2017) (APAC-Agência Pernambucana de Aguas e Clima) was roughly constant, with some years of high precipitation, such as those reported in 2000, 2010 and 2017. The period from April to August is characterized as the rainy season and the one from September to March as the dry season (Figure 6a).

A large seasonal variation of chlorophyll (Chl *a*) concentration occurred. From October to November, during the dry season, the Chl *a* concentration varied from 0.084 to 0.108 $\mu g.l^{-1}$, while during April- May (peak of the rainy season), it oscillated from 0.621 to 2.113 $\mu g.l^{-1}$ (Figure 6b). The stratification in chlorophyll concentration was clearer during the rainy season, with high concentration in shallow waters near the mouths of rivers, and lower on depths greater than 18 m (Figure 6b).

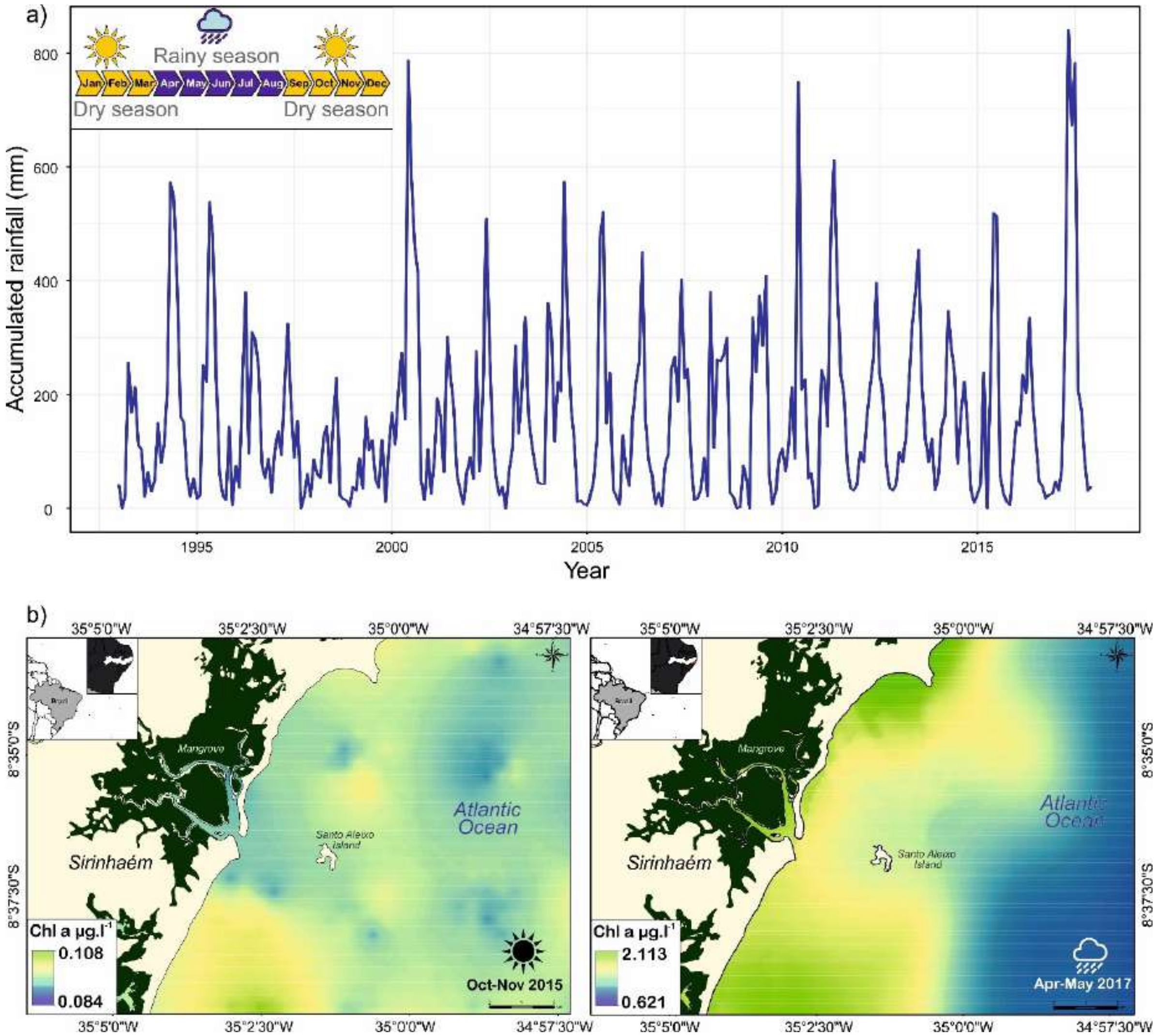


Figure 6. (a) Monthly rainfall time series between 1993 to 2017 (Source: APAC) and (b) Chlorophyll a concentrations average (µg.l^{-1}) maps derived by Aqua satellite images (MODIS) to distinct periods (dry – Oct/Nov and rainy – Apr/May) in Sirinhaém-PE coast, Northeastern Brazil. The colors in (a) represent the months peak and minimum probability of precipitation obtained by classical additive model.

Target species (shrimps)

Temporal catch volume

Based on the official statistics, the state of Pernambuco, between 1988 and 2007, the shrimp catches trend increase (Figure 7a). Minimum catches values were recorded in 1992 (91.4 t), mainly due to a failure on data collection. Higher catches were recorded in 2005 (583.1 t) and 2006 (553.1 t). During this period, Barra de Sirinhaém was responsible, on average, for 25% of the state's production, peaking in 1988, when it accounted for approximately 45% of all shrimp production in Pernambuco.

During this period (1988-2007), shrimp catches in Sirinhaém were an average (± SD), 63.2 ± 13.9 t/ year. Minimum catches were recorded in 1991 (41.9 t) and 1999 (42.8 t) and maximum values in 1996 (86.0 t) and 2004 (91.0 t) (Figure 7a).

Considering data obtained by logbooks, shrimps of the family Penaeidae are the main species exploited by artisanal trawl fishery, including: the pink shrimp (*Penaeus subtilis* and *P. brasiliensis*), the white shrimp (*Penaeus schmitti*), and the seabob shrimp (*Xiphopenaeus kroyeri*) (Figure 7b). Some shrimp species with low catches, such as *Nematopalaemon schmitti* and *Exhippolysmata oplophoroides*, also occurs, but are not reported, while others are not separated by species (e.g. *P. brasiliensis* and *P. subtilis*, grouped and commercialized as pink shrimp).

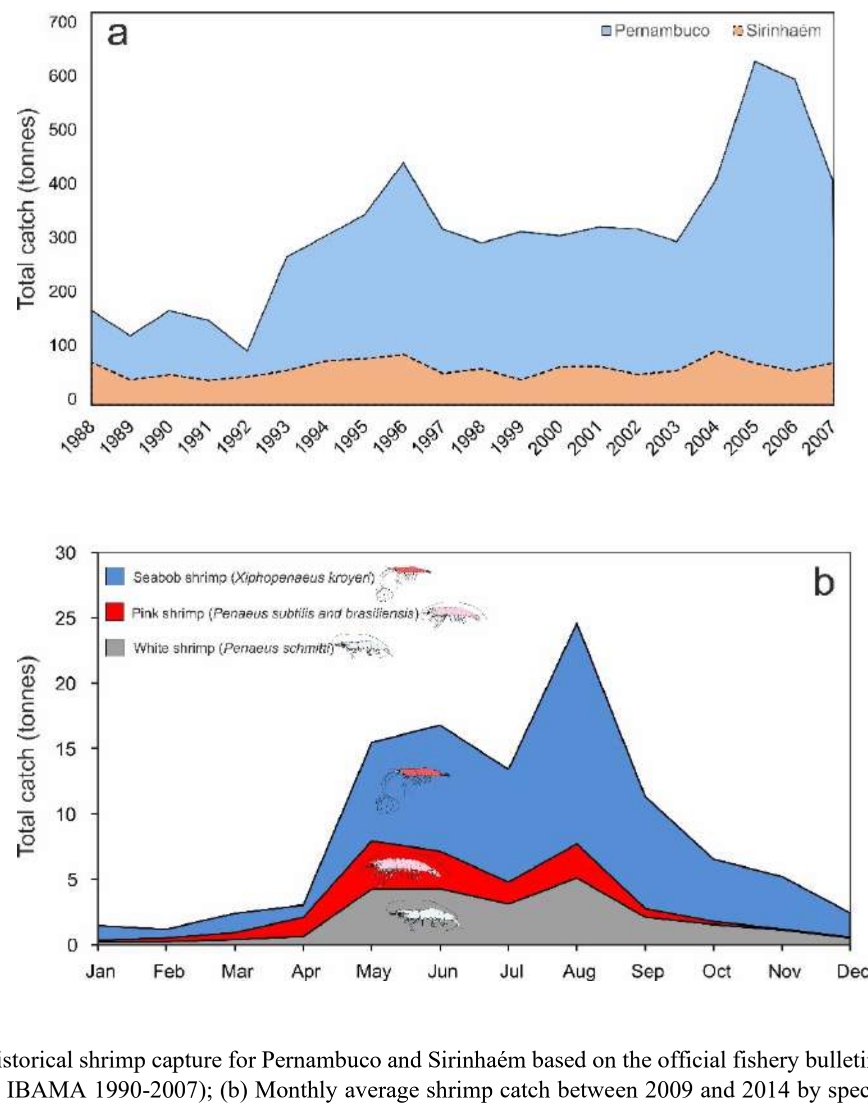


Figure 7. (a) Historical shrimp capture for Pernambuco and Sirinhaém based on the official fishery bulletins (source: IBGE 1988-1989 and IBAMA 1990-2007); (b) Monthly average shrimp catch between 2009 and 2014 by species (source: non-official statistics logbook).

Monthly catch trends had a similar pattern for all main target species. Catches peak occurred between May and August, while from September to April they were reduced to less than 5 t per month (Figure 7). *X. kroyeri* had the highest catches for all months, followed by the white shrimp (*P. schmitti*) and by the pink shrimp (*P. subtilis* and *P. brasiliensis*).

The months with highest catches are also those with highest rainfall. A positive and statistically significant relationship occur between catch and rainfall for all species (Figure 8): *X. kroyeri* (Figure 8a) (r = 0.49; *p-value* = 0.11; R^2 = 0.24), *P. schmitti* (Figure 8b) (r = 0.66; *p-value* = 0.01; R^2 = 0.45) and *P. subtilis* (Figure 8c) (r = 0.87; *p-value* < 0.001; R^2 = 0.76).

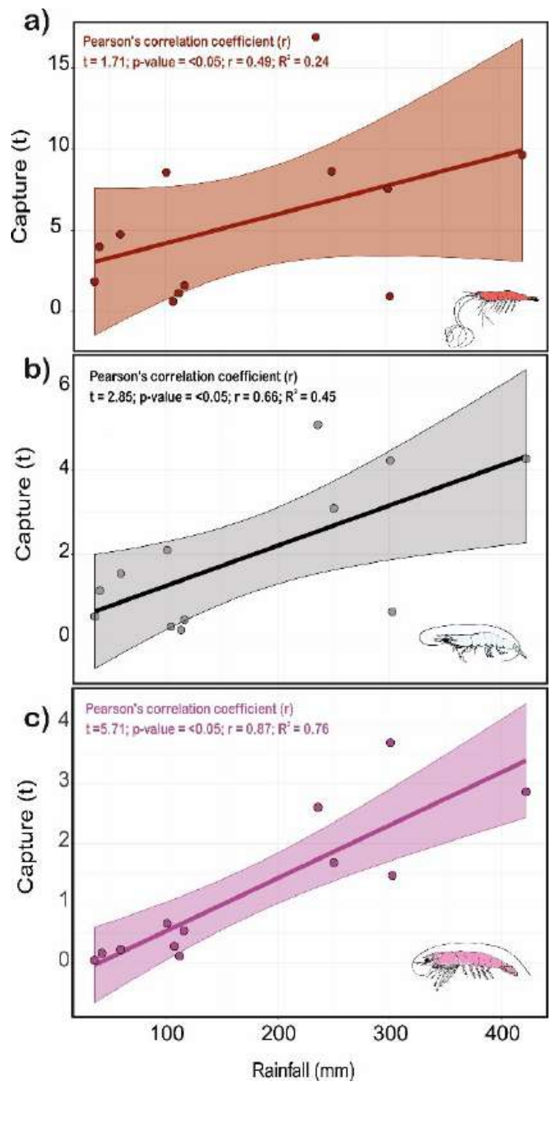


Figure 8. Correlation between the monthly rainfall average (mm) (1993-2017) and the capture of (a) *Xiphopenaeus kroyeri*, (b) *Penaeus schmitti* and (c) *Penaeus subtilis/brasiliensis* from 2009 to 2014 (source: Logbook data) in Sirinhaém-PE coast, Northeastern Brazil.

Population dynamics and stock assessment

Among the species exploited by artisanal trawl fishery in the study region, *X. kroyeri* have the higher growth coefficient (k = 2.8 year^{-1}), earlier maturity (L_{50} = 8.9 cm Total Length - TL) and lowest asymptotic total length (L_{∞}= 14 cm TL) when compared to the other species. Except for *P. schmitti* that reproduce year-round, the other species have reproductive peaks associated mainly to months of low rainfall. *X. kroyeri* reproduces mainly in November, December and February and *P. subtilis* from October to March (Table 1). For both species (*X. kroyeri*, *P. subitilis* and *P. schmitti*), the length at first capture is below the length of first sexual maturity (e.g., *P. subtilis*; Lc= 9.46 cm and L_{50}= 11.9 cm), however the traditional stock assessment carried out do not indicate overexploitation (see Table 1 for more details).

Table 1. Summary of the biological and fishery parameters of the shrimp target species in the coast of Pernambuco, north-eastern Brazil. Where L_{50}: length at first maturity (cm); L_{∞}: asymptotic total length (cm); k: growth coefficient (year^{-1}); F: fishery mortality (year^{-1}); M: natural mortality (year^{-1}); E: exploitation rate; Lc: length at first catch (cm), Long: longevity (year^{-1}); maximum recruitment yield (E_{MRY}) and Season: periods of significant reproductive intensity (month).

		Xiphopenaeus kroyeri (seabob-shrimp)[1,2]	*Penaeus subitilis* (pink-shrimp)[3,4]	*Penaeus schmitti* (white-shrimp)[5,6]
Population parameters				
	L_{50}	8.9	11.9	14.2
	L_{∞}	14.0	21.6	19.3
	k	2.80	1.19	1.47
	M	3.60	2.15	1.37
	Long	1.34	2.38	1.91
Fishery parameters				
	F	6.80	4.71	2.89
	E	0.65	0.68	0.68
	Lc	8.42	9.46	11.55
	E_{MSY}	0.79	0.67	0.76
Reproduction				
	Season	Nov-Dec and Feb	Oct-Mar	Jan-Dec

1,2- Lopes et al. (2017, 2014) 3,4- Silva et al. (2016, 2015) 5,6- Peixoto et al. (2018) and Silva et al. (2018)

Diet composition

On average, the stomachs for the three target species (*X. kroyeri* - Xip.kro, *P. schmitti* - Pen.sch and *P. subtilis* - Pen.sub) were over 30% full, indicating low percentage of empty (Lira *et al.*, 2021a). Sixteen food and non-food items were identified, where over 50% occurrence (FO%) of stomach contents of three species were based on Cirripedia, polychaetes and decapoda, indicating very similar diets (Figure 9). Another significant part consists of sediment, organic matter (O.M.) and algae totaling averaging 30 to 40% of occurrence of the diet. A final group of 10 items among gastropods, nematodes, ostracods and correspond approximately from 5 to 10% of the stomach contents of the species (Figure 9).

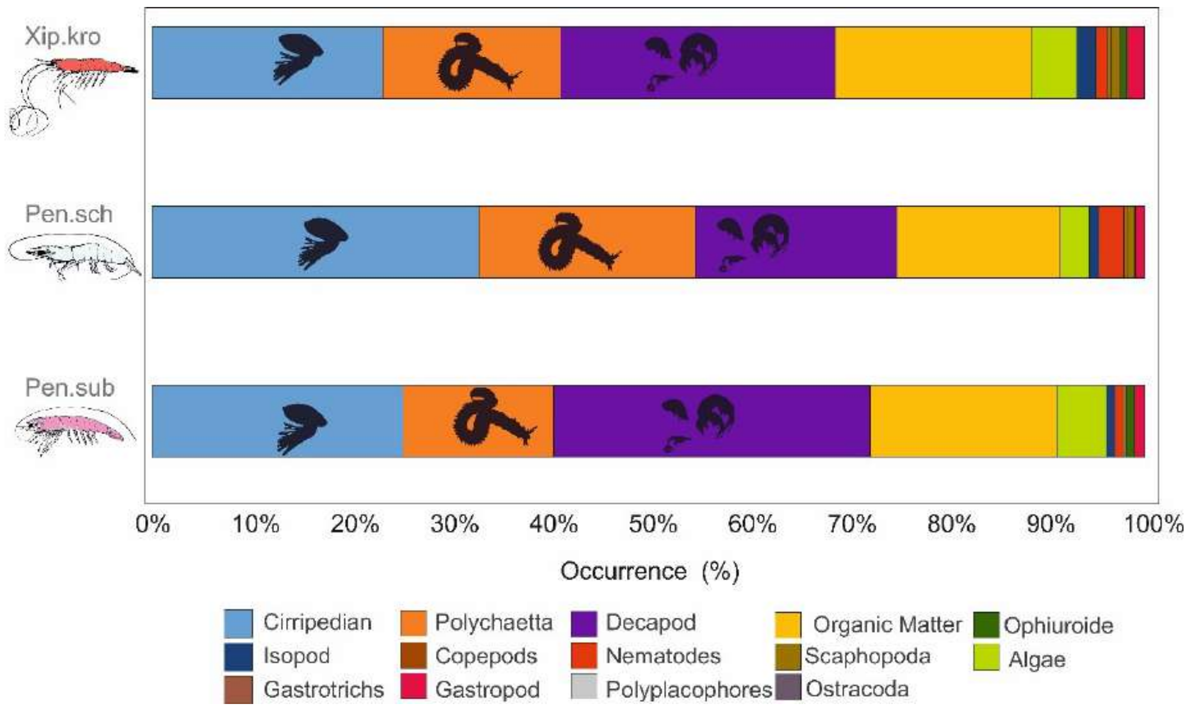


Figure 9. Percentage occurrence of stomach contents of shrimp species (X. kroyeri - Xip.kro, P. schmitti - Pen.sch and P. subtilis - Pen.sub) caught in Barra de Sirinhaém, Northeastern Brazil, according Lira et al. (2021b).

Fish bycatch

Catch composition

The amount of fish bycatch of the small-scale shrimp trawling fishery carried out in Northeast Brazil, specifically in the Sirinhaém is much lower than reported from other regions in Brazil and around the world (0.39 kg fish bycatch caught for each 1 kg of shrimp) (Table 2).

The ichthyofauna incidentally caught by the small-scale shrimp fishery over the past two decades off Sirinhaém-PE showed a total of 39, in the mid-2001, to 85 species, during 2011-2014 (Table 2). A total of 24,217 individuals of 93 species, 21 orders and 35 families were caught during the overall period. Two families were more representative, Pristigasteridae (3 species) and Scianidae (19 species), accounting, on average, for 70% of the total catch, in 2001 and 2018. Most species were classified as occasional (2001-2002, 15 species; 2011-2014, 45 species) or accessory (2001-2002, 8 species; 2011-2014, 12 species; 2017-2018; 21 species). Species classified as constant represented 41%, 34% and 55% of sampled species for 2001-2002, 2011-2014 and 2017-2018, respectively (Table 2).

Chirocentrodon bleekerianus, Odontognathus mucronatus and *Pellona harroweri* were the species of highest abundance over time, followed by species of the genre Stellifer. In the past two decades, some new species were reported in the fishery: Elasmobranchii- *Rhizoprionodon porosus, Pseudobatos percellens, Urotrygon microphthalmum*; and some Perciformes- *Diapterus auratus* and *D. rhombeus*. (Table 2).

During the studied period (2001 to 2018), according to the Brazilian assessment based on the IUCN Red List classification, none of the species caught was classified as Threatened (Vulnerable- VU; Endangered- EN; or Critically Endangered- CR), four were categorized as Near Threatened (NT) (*Hyporhamphus unifasciatus*, *Cynoscion acoupa, Lutjanus analis* and *L. synagris*) and 13 as Data Deficient (DD). Most species (77) were categorized as Least Concern (LC), and six as Not Evaluated (NE) (Table 1). All species NT were rare (Table 2).

Bycatch destination

Fifty-nine species (65%) of the bycatch are regularly consumed by the local community, while 14 species (15% of all species) are commercialized (e.g., hake and croakers *Cynoscion virescens*, *Isopisthus parvipinnis*, and *Micropogonias furnieri*), contributing as additional source of income (Table 2). However, 19 species (21%), are exclusively discarded, mainly small size sardines (e.g., *Chirocentrodon bleekerianus* and *Odontognathus mucronatus*), catfish (e.g., *Aspistor luniscutis, Aspistor quadriscutis*) and puffer fish (e.g., *Lagocephalus laevigatus*, *Sphoeroides greeleyi*, *Sphoeroides testudineus*) (Table 2).

Table 2. List of species, number of individuals (white number), occurrence frequency based in Dajoz (1983), Reginal IUCN classification (ICMbio, 2018) (Near Threatened (NT), Least Concern (LC), Data Deficient (DD)) and use/trade destination (Di: Discarded; Co: Consumed; Cm: Commercialized), for bycatch sampled in Sirinhaém, Pernambuco, Northeastern Brazil, from a small-scale shrimp trawl fishery. Constancy index (IC): C- constant; A- accessory; O- occasional and NO- did not occur. Sources: 2001-2002, from Tischer and Santos (2003), 2011-2014, from Silva Júnior et al. (2019), and 2017-1018, from REBYC-LAC II .

Order	Family	Specie	Use/Trade	Constancy index (IC): C >%50	A 25-50%	O <25%	NO
				IUCN	2001 2002	2011 2014	2017 2018
Carcharhiniformes	Carcharhinidae	*Rhizoprionodon porosus (Poey 1861)*	*Di*	*DD*			1
Rajiformes	Rhinobatidae	*Pseudobatos percellens (Walbaum 1792)*	Co	*DD*		1	
Myliobatiformes	Dasyatidae	*Hypanus guttatus (Bloch & Schneider 1801)*	Co	*LC*		3	
	Urotrygonidae	*Urotrygon microphthalmum Delsman 1941*	Di	*DD*		1	
Albuliformes	Albulidae	*Albula nemoptera (Fowler 1911)*	Co	*LC*		2	
Anguilliformes	Ophichthidae	*Myrichthys ocellatus (Lesueur 1825)*	Di	*LC*			3
Clupeiformes	Pristigasteridae	*Chirocentrodon bleekerianus (Poey 1867)*	Di	*LC*		2363	128
		Odontognathus mucronatus Lacepède 1800	Di	*LC*	61	1806	84
		Pellona harroweri (Fowler 1917)	Co	*LC*	203	6137	376
	Engraulidae	*Anchoa filifera (Fowler 1915)*	Co	*LC*	101		
		Anchoa januaria (Steindachner 1879)	Co	*LC*			2
		Anchoa spinifer (Valenciennes 1848)	Co	*LC*		58	1
		Anchoa tricolor (Spix & Agassiz 1829)	Co	*LC*		4	
		Anchovia clupeoides (Swainson 1839)	-	*LC*	25		
		Anchoviella lepidentostole (Fowler 1911)	Di	*LC*	1	4	
		Cetengraulis edentulus (Cuvier 1829)	Co	*LC*	55	238	10
		Lycengraulis grossidens (Spix & Agassiz 1829)	Co	*LC*	19	267	7
	Clupeidae	*Harengula clupeola (Cuvier 1829)*	Co	*LC*	12	46	3
		Opisthonema oglinum (Lesueur 1818)	Co	*LC*	3	17	1
		Rhinosardinia bahiensis (Steindachner 1879)	Co	*LC*		**2**	
Siluriformes	Ariidae	*Aspistor luniscutis (Valenciennes 1840)*	Di	*LC*	2	22	
		Aspistor quadriscutis (Valenciennes 1840)	Di	*LC*		1	9
		Bagre bagre (Linnaeus 1766)	Co	*NT*		**30**	
		Bagre marinus (Mitchill 1815)	Co	*DD*	4	170	2
		Cathorops spixii (Agassiz 1829)	Co	*LC*		1	3
		Sciades herzbergii (Bloch 1794)	Co	*LC*		1	
Ophidiiformes	Ophidiidae	*Lepophidium brevibarbe (Cuvier 1829)*	Di	*DD*		1	
Atheriniformes	Atherinopsidae	*Atherinella brasiliensis (Quoy & Gaimard 1825)*	Co	*LC*		1	
Beloniformes	Hemiramphidae	*Hyporhamphus unifasciatus (Ranzani 1841)*	Cm	*NT*		5	

Carangiformes	Echeneidae	*Echeneis naucrates Linnaeus 1758*	Di	*LC*		1	
	Carangidae	*Carangoides bartholomaei Cuvier 1833*	Co	*LC*		3	
		Caranx hippos (Linnaeus 1766)	Co	*LC*		2	
		Chloroscombrus chrysurus Jordan & Gilbert 1883	Co	*LC*	12	36	
		Selene brownii (Cuvier 1816)	Cm	*LC*		132	8
		Selene setapinnis (Mitchill 1815)	Cm	*LC*	7	9	
		Selene vomer (Linnaeus 1758)	Cm	*LC*	2	31	
Istiophoriformes	Sphyraenidae	*Sphyraena guachancho Cuvier 1829*	*Co*	*LC*	**4**	**79**	**5**
Pleuronectiformes	Paralichthyidae	*Citharichthys macrops Goode 1880*	Co	*LC*		1	
		Citharichthys spilopterus Günther 1862	Co	*LC*		31	
		Cyclopsetta chittendeni Bean 1895	Co	*LC*		1	
		Etropus crossotus Jordan & Gilbert 1882	Co	*LC*	10	17	5
		Paralichthys brasiliensis (Ranzani 1842)	Co	*LC*		26	
	Achiridae	*Achirus declivis Chabanaud 1940*	Co	*LC*	12	69	2
		Achirus lineatus (Linnaeus 1758)	Co	*LC*		11	
		*Trinectes inscriptus (Gosse 1851)**	-	-	5		
		Trinectes paulistanus (Miranda Ribeiro 1915)	*Co*	*LC*		155	
Syngnathiformes	Dactylopteridae	*Dactylopterus volitans (Linnaeus 1758)*	Co	*LC*			1
	Cynoglossidae	*Symphurus plagusia (Quoy & Gaimard 1824)*	Co	*LC*	57	33	
		Symphurus tessellatus (Quoy & Gaimard 1824)	Co	*LC*		102	3
Scombriformes	Trichiuridae	*Trichiurus lepturus Linnaeus 1758*	Co	*LC*	15	219	10
	Stromateidae	*Peprilus paru (Linnaeus 1758)*	Di	*LC*		14	1
Perciformes	Gerreidae	*Diapterus auratus Ranzani 1842*	Co	*LC*		25	4
		Diapterus rhombeus (Cuvier 1829)	Co	*LC*		43	5
		Eucinostomus argenteus Baird & Girard 1855	Co	*LC*	1	60	
		Eucinostomus gula (Quoy & Gaimard 1824)	Co	*LC*	129	211	4
		Eugerres brasilianus (Cuvier 1830)	Co	*LC*	8		
	Mullidae	*Upeneus parvus Poey 1852*	Co	*LC*		1	
	Pempheridae	*Pempheris schomburgkii Müller & Troschel 1848*	Co	*LC*		2	
	Serranidae	*Diplectrum formosum (Linnaeus 1766)*	Co	*LC*		1	
	Haemulidae	*Anisotremus moricandi (Ranzani 1842)*	Co	*LC*		4	
		Conodon nobilis (Linnaeus 1758)	Co	*LC*	12	251	94
		Genyatremus luteus (Bloch 1790)	Co	*LC*		9	1
		Haemulon aurolineatum Cuvier 1830	Co	*LC*		10	
		Haemulon plumierii (Lacepède 1801)	Co	*DD*		1	
		Haemulon steindachneri (Jordan & Gilbert 1882)	Co	*LC*		6	
		Haemulopsis corvinaeformis (Steindachner 1868)	Cm	*LC*	139	1113	63
	Lutjanidae	*Lutjanus analis (Cuvier 1828)*	Cm	*NT*		1	

		Lutjanus synagris (Linnaeus 1758)	Cm	*NT*	2	18	
	Polynemidae	*Polydactylus octonemus (Girard 1858)**	-	-	53		
		Polydactylus virginicus (Linnaeus 1758)	Co	*LC*		270	24
Scorpaeniformes	Triglidae	*Prionotus punctatus (Bloch 1793)*	Di	*LC*		3	1
Moroniformes	Ephippidae	*Chaetodipterus faber (Broussonet 1782)*	Di	*LC*	1	13	4
Acanthuriformes	Sciaenidae	*Bairdiella ronchus (Cuvier 1830)*	Co	*LC*	32	61	
		Cynoscion acoupa (Lacepède 1801)	Cm	*NT*	6		
		Cynoscion leiarchus (Cuvier 1830)	Cm	*LC*	49		
		Cynoscion virescens (Cuvier 1830)	Cm	*LC*	10	58	11
		Isopisthus parvipinnis (Cuvier 1830)	Cm	*LC*	47	804	19
		Larimus breviceps Cuvier 1830	Cm	*LC*	138	982	41
		Macrodon ancylodon (Bloch & Schneider 1801)	Co	*LC*	7	156	4
		Menticirrhus americanus (Linnaeus 1758)	Cm	*DD*		218	65
		Menticirrhus littoralis (Holbrook 1847)	Co	*DD*		4	
		Micropogonias furnieri (Desmarest 1823)	Cm	*LC*		56	1
		Nebris microps Cuvier 1830	Co	*LC*		73	2
		Ophioscion punctatissimus Meek & Hildebrand 1925	Co	*DD*	7	293	4
		Ophioscion sp.[1]*	-	-		294	59
		Paralonchurus brasiliensis (Steindachner 1875)	Co	*LC*	86	517	89
		Stellifer brasiliensis (Schultz 1945)	Co	*LC*	469	286	31
		Stellifer microps (Steindachner 1864)	Co	*LC*		1634	154
		Stellifer rastrifer (Jordan 1889)	Co	*LC*		702	140
		Stellifer stellifer (Bloch 1790)	Co	*LC*		530	54
		Umbrina coroides Cuvier 1830	Co	*LC*		1	
Lophiiformes	Ogcocephalidae	*Ogcocephalus vespertilio (Linnaeus 1758)*	Di	*LC*		1	
Tetraodontiformes	Ostraciidae	*Acanthostracion polygonius Poey 1876*	Di	*LC*		1	
	Tetraodontidae	*Lagocephalus laevigatus (Linnaeus 1766)*	Di	*LC*		2	2
		Sphoeroides greeleyi Gilbert 1900	Di	*LC*		1	
		Sphoeroides testudineus (Linnaeus 1758)	Di	*DD*		1	

*Species do not occur in the study area, probably taxonomic error; **Species without taxonomic confirmation.

Biological traits

Biological traits of 39 species of the ninety-three (93) species were obtained by literature review. The asymptotic length (L_∞) and the length at first maturity (L_{50}) ranged from 8.5 and 5.1 cm (*Rhinosardinia bahiensis*) to 203.4 and 67.2 cm (*Hypanus guttata*), respectively, while the growth coefficient (*k*) ranges from 0.05 year^{-1} (*Lutjanus analis*) to 1.65 year^{-1} (*Anchoa tricolor*) (Table S1 and Figure 10). The most abundant species have rapid growth and relatively low asymptotic size (e.g., *Pellona harroweri* and *Chirocentrodon bleekerianus*).

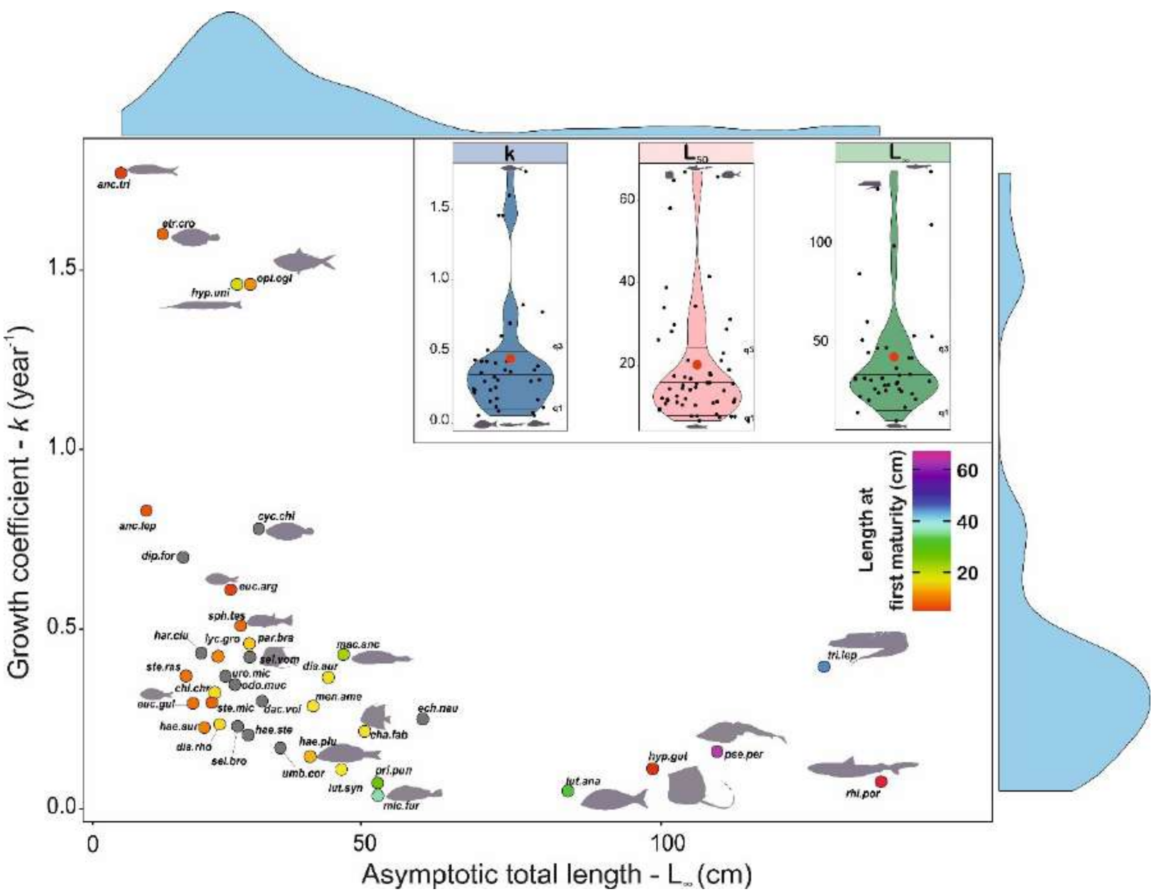


Figure 10. Asymptotic length (L_∞ - cm), length at first maturity (L_{50} - cm) and growth coefficient (k – year^{-1}) for 39 fish species caught as bycatch in Sirinhaém, Pernambuco Northeast Brazil. Each point represents a set of L_∞ and k parameters for each species (see Table S1for species name). The rainbow color ramp dots represent the L_{50} value and grey dots indicate absence of this value. Top-right, in the violin plot the red and grey points are the general mean and all values for K, L_∞ and L_{50}).

Feeding habit

The majority of the fifty-four (54) fish species evaluated has their diets associated to benthic preys, in particular shrimps (Figure 11 and Table S2). The greatest number of species (17 spp) were reported to have zoobentivorous feeding strategies (e.g., *Larimus breviceps* – Lar.bre, *Paralonchurus brasiliensis* – Par.bra and *Stellifer microps* – Ste.mic), with high proportion (25 to 75%) of crustaceans (e.g., shrimps, crabs and polychaetes), followed by piscivores (6 spp) and zooplanktivores (6 spp) (Figure 11). In contrast, few species feed preferentially on detritus (2 spp) and phytoplankton (2 spp) (Figure 11 and Table S2).

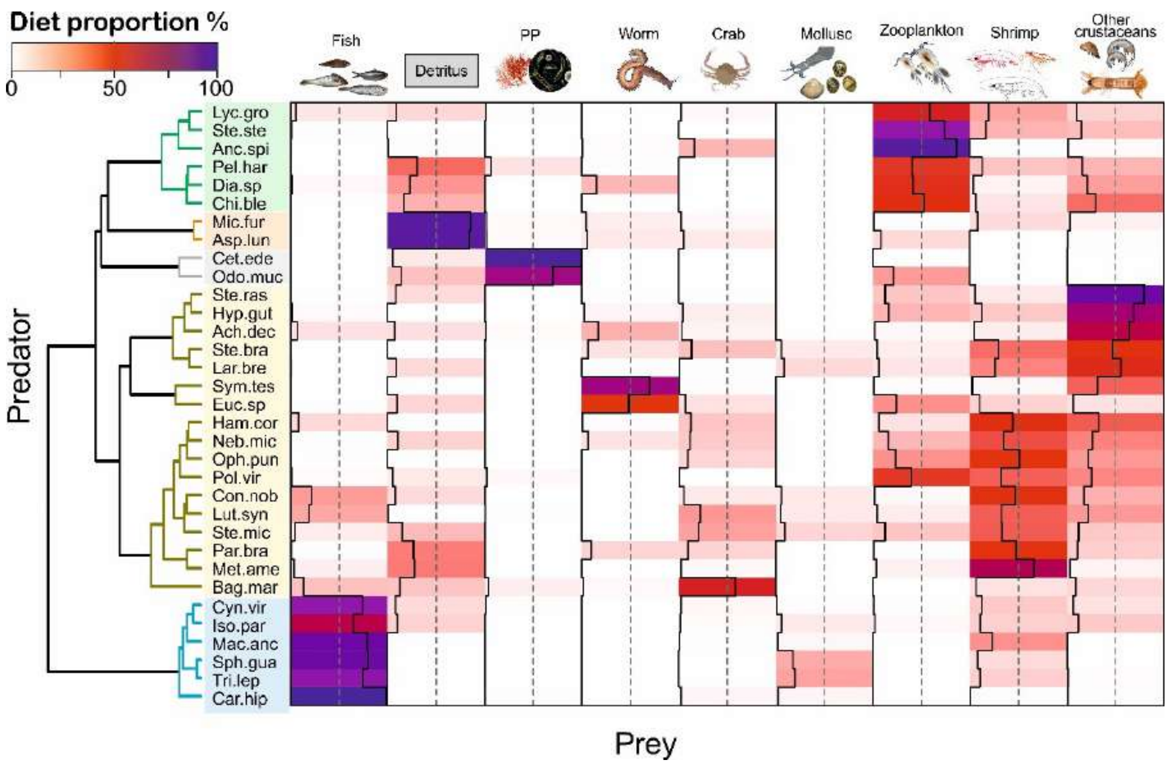


Figure 11. Weight contribution (%) of diet for fish caught as bycatch off the coast of Sirinhaém, Northeastern Brazil. The dendrogram on the left was performed using hierarchical cluster analysis based on the proportion of the predators' diet. Species abbreviation and diet sources may be access in Table S2.

Discussion

The Ecosystem-Based Fishery Management (EBFM) or Ecosystem Approach to Fishery (EAF) has been successful applied worldwide as a model for several fisheries (Pitcher *et al.*, 2009), such as in the United States- Townsend et al. (2019); Baltic sea- Möllmann et al. (2014); Australia- Smith et al. (2007); Canada- O'Boyle and Jamieson (2006); New Zealand- Reid and Rout (2020); Mexico- Arnott et al. (2012); South African- Shannon et al. (2004); Southern Brazil- Scherer and Asmus (2016). However, this approach requires complementary information that considers the dynamic of ecosystem, fishery, economy, ecology and biology of the target and non-target species (Brodziak and Link, 2002; Pikitch *et al.*, 2004; Babcock *et al.*, 2005; Kroetz *et al.*, 2019; Lidström and Johnson, 2020). In tropical multispecies fisheries, such as the trawls, EAF implementation is extremely complex, given the diversity of the fleet and species caught, and the scarcity and poorly informative nature of the data.

The shrimp fishery and the environmental factors

The shrimp fishing in Northeast Brazil, specifically in Sirinhaém, is a non-regulated activity and the largest and most productive motorized fishing fleet in Pernambuco. In order to provide information that could be useful for fishery regulation under the EAF paradigm, for the first time in the region, this study reported an integrative study of this fishery, encompassing the characteristics of environment and fishing aspects, and the dynamics of the target and bycatch species.

In general, the trawl fisheries have multiple targets such as fish, mollusks and crustaceans (FAO, 2020b). Often, they occur in diverse depths, covering large dragged areas of unconsolidated bottoms such as sand, mud or gravel (Amoroso *et al.*, 2018; Rijnsdorp *et al.*, 2020). The coastal zone, where the shrimp fleet mainly operates, is formed by a flat bottom area in the first 10 m of depth, followed by a small depression to 20 m, where mud is concentrated. These muddy areas are constituted by the coastal depositional of nearshore sub-tidal locations characterized by relatively low energy hydrodynamic conditions. Shallow estuaries and embayment's containing a high proportion of silt and clay tend to form extensive low-gradient morphological flat surfaces (Healy, 2005; Anthony *et al.*, 2010). Locally, mud is enriched in organic matter (Serrano *et al.*, 2016) due to supply of nutrients from the rivers, providing an ideal habitat for extensive developing of the benthic fauna, favoring the growth of the target species of coastal fisheries (Holland *et al.*, 1977; Thrush *et al.*, 2003, 2004). Given the high density of target species, these are preferred physical habitats of many fisheries, such as the shrimp-fisheries, where intense trawling takes place (Bourguignon *et al.*, 2018; Sciberras *et al.*, 2018). Artisanal trawl fishery in the south of Pernambuco is carried out at depths among 10 to 20 m, concentrating the effort mainly in the thinner mud layers that are extremely rich in silt. In contrast, the sandy and sandy-gravel bottoms are weakly exploited.

Small-scale fishing has often low level of technology, capture, profit and fishing trip autonomy in terms of time at sea, using mainly fishing areas close to their usual landing site (Dias-Neto, 2011), facilitating the fishery production chain, since the largest part of production is commercialized locally fresh. In addition, in Northeastern Brazil, the fished muddy bottoms are small and close to the coast not extending to deeper areas, mainly associated to the mouth of the rivers, where most shrimp biomass are concentrated, and consequently the fleet in these shallow coastal areas (Santos, 2010). Hence, these fishing grounds and consequently the distribution and abundance of the target species are high associated to coastal weather patterns, such as the rainfall.

The monthly rainfall fluctuations in Sirinhaém area are well defined reflecting the nutrient flows from the rivers to the associated coastal zone. During the periods of high precipitation (April to August) the highest chlorophyll concentration ratios are reported in shallow waters near the mouth of river. In tropical ecosystems, especially the coastal ones, the patterns of precipitation are considered as one most important environmental predictors of the distribution, diversity, and abundance of the species across multiple habitats (e.g., estuary, reef, shelf break, beach and etc.) (Madduppa *et al.*, 2012; Vilar *et al.*, 2013; Mason-Romo *et al.*, 2017; Silva Júnior *et al.*, 2019; Molina *et al.*, 2020). Thus, rainfall influences and defines the dynamics and, consequently, the tropical coastal fishery yields (Eduardo *et al.*, 2016; Barange *et al.*, 2018; Souza *et al.*, 2018; Lira *et al.*, 2021b).

Penaeidae shrimps are widely exploited in the Northeastern of Brazil, particularly the seabob shrimp (*X. kroyeri*), the most abundant one, and the pink (*P. subtilis*) and white shrimp (*P. schmitti*), with high market-values (Santos, 2010). In Pernambuco, the months with highest abundances and catches of the target and non-target species are also those with highest rainfall, while the lowest abundances and catches are related to dry periods (Figure 12), which correspond to the peak of reproduction of these species and the main bycatch (Silva Júnior *et al.*, 2015, 2019; Lopes *et al.*, 2017; Eduardo *et al.*, 2018a; Peixoto *et al.*, 2018; Silva *et al.*, 2018). Hence, given the low abundance during the dry season, the trawling activities are basically inactive or economically unprofitable and, due to the decline in production (Tischer and Santos, 2003; Silva Júnior *et al.*, 2019), barely cover the operating costs of the fishery. This phenomenon could be considered as a "natural closed season" (Lira *et al.*, 2021b).

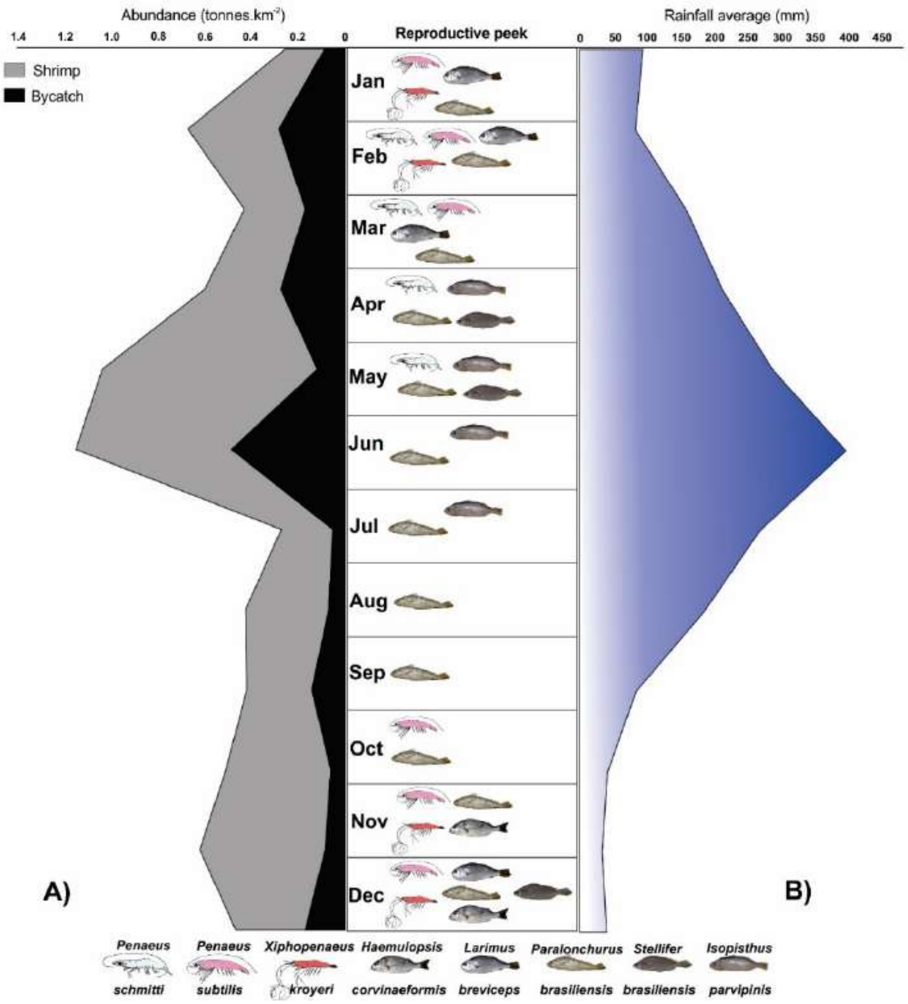


Figure 12. Reproductive season of shrimps and some of main fish bycatch species caught in Sirinhaém-PE coast, Northeastern Brazil (sources: Eduardo et al. (2018); Lopes et al. (2017); Peixoto et al. (2018); Silva et al. (2016); Silva Júnior et al. (2015). A) Monthly abundance (CPUA – tonnes.km^{-2}) of the fish bycatch and shrimp species based in Silva Júnior et al. (2019); B) Monthly rainfall average (mm) (1993-2017; APAC).

The target and by catch species population parameters and status

Recent traditional stock assessments in the region did not indicate overexploitation of the target species (Silva *et al.*, 2015, 2018; Lopes *et al.*, 2017). *Xiphopenaeus kroyeri* is the most abundant among the target species. It presents the fastest growth rates, smallest size and higher natural and fishing mortality ratios. However, the length at first capture is below the length of first sexual maturation for all species. Depending on the situation, this can be considerably harmful to sustainability of the stocks, given that most of harvest individuals might have not been able to reproduce and contribute for population renewal (Blaber *et al.*, 2000). However, since these species are r strategists with small-size, fast-growing, early maturity, high spawning potential and resilient, in the current mortality levels, growth overfishing is unlikely.

The bycatch species of the Scianidae and Pristigasteridae families were the most important fish bycatch in terms abundance and biomass. The incidental catch in Sirinhaém primarily removes juveniles

(Eduardo *et al.*, 2018a; Lira *et al.*, 2019; Silva Júnior *et al.*, 2019). The occurrence of these groups is constantly reported to small and large-scale trawl fisheries from many regions in Brazil, from both Southern and Southeastern regions (Vianna and Almeida, 2005; Branco and Verani, 2006; Bernardo *et al.*, 2011; Branco *et al.*, 2015; Rodrigues-Filho *et al.*, 2015), and Northern and Northeastern of Brazil (Isaac and Braga, 1999; Silva Júnior *et al.*, 2013; Bomfim *et al.*, 2019; Marceniuk *et al.*, 2019; Passarone *et al.*, 2019). The high bycatch capture rates were considered as one of the main threats for Brazilian sciaenids (Chao *et al.*, 2015), which have been widely exploited in the southern coast of Brazil, leading to strong decreases in biomass and catch size of their stocks (Vasconcellos and Haimovici, 2006; De Miranda and Haimovici, 2007).

Although the problem of high non-target catch rates is well known, most bycatch species are still extremely poorly studied in terms of biological characteristics such as population dynamics, breeding season and feeding behavior. Most species in the present study are classified as Data Deficient (DD), due to the lack of available data (ICMbio, 2018), but with a high probability for some of them to be threatened. From the 93 fish bycatch species reported in the present study, less than half of them have information available about diet (34 species – 37%) or growth parameters (37 species - 40%), while 58% (54 species) have some estimations of reproductive aspects, such as L_{50}.

Lessons learned from the integrated analysis and insights for the management in an EAF

Shrimp fisheries management often focus on the target species and measures associated to fleet or gear limitations and closed season fishing, as in Brazil. The only regulation that considers non-target species, is the TED for industrial fishing (Santos, 2010; Dias-Neto, 2011). For our study case (Pernambuco, Northeast Brazil), although there are currently no management measures for the fishery, it was observed that the shrimp stocks do not appear to be in risk. However, the scenario is uncertain when considering non-target species (bycatch), where any knowledge is available.

The absence of basic information (e.g., growth, mortality, breeding season, and feeding behavior) hampers any conservation and assessment action for bycatch species and is considered one of the main barriers for an effective ecosystem-based management (Jacobson *et al.*, 2006; Ruckelshaus *et al.*, 2008; Pita *et al.*, 2020). The empirical relationships to estimate lacking parameters (such as asymptotic length, growth coefficient and length at first maturity) (Pauly, 1980, 1986; Froese and Binohlan, 2000, 2003; Le Quesne and Jennings, 2012; Froese *et al.*, 2014) are often used to overcome the absence of data, however, there is a considerable amount of uncertainty in those empirical formulae, thus they should be used with caution. Population parameters are crucial for fishery management since they are required to the application of assessment and ecosystem approaches. Obtaining the population parameters of bycatch species should be priority for most multispecies fishery, such as the small-scale shrimp fishery.

Some species of the bycatch of our stuy case may be of specific concern, such as the Sciaenidae which present evidence of overexploitation in some regions of Brazil. However, overfishing has not been necessarily considered as the sole or even as the main causes of stocks decrease. Recently, Verba et al. (2020) evaluated the cumulative effect of the climate change (e.g., the increasing of the sea temperature), fishery exploitation and specific life-history traits, classifying many of these Sciaenidae species as fully exploited in the Brazilian Exclusive Economic Zone. In our study area, the environmental drivers were strongly dominant and decisive in the identification of fishing areas and periods, also indicating that they may be key elements in the management of the ecosystem. Moreover, a high correlation between the patterns of abundance and reproduction with the rainfall and chlorophyll concentration exist, indicating that the success of the fishing harvests may be related to environmental drivers. Lira et al. (2021a) modeled that the decreased trawling efforts up to 10% were promising, with better fishing management performance than the closed season which did not present significant improvements in terms of ecosystem functioning. However, the environmental changes caused significant adverse impacts, indicating that environmental factors were more decisive than the effort control.

The challenges of small-scale fisheries management are multiple (Arthur, 2020; Jimenez *et al.*, 2020), especially considering the highly heterogeneous social, political, economic and conservation factors of the fishery. Large-scale and small-scale shrimp trawling fisheries are inherently different, not only in the amount and proportion of bycatch, but also on its destination. In the small-scale shrimp trawling fishery carried out in Northeast Brazil, specifically in the Sirinhaém, the amount of bycatch is lower than the reported in other regions in Brazil and around the world (Silva Júnior *et al.*, 2019), with most of it being used by the local community, as additional source of food and income (Figure 13). In this way, the impact of the fishing activities on the ecosystems appears to be counter-balanced by the beneficial role of the bycatch in the local community (Carvalho *et al.*, 2020). The commercialized bycatch consists of larger-sized and/or species with market value, while the bycatch consumed by the local community is mainly made up by small-sized individuals and/or species, abundant in the shrimp trawl fishery in the region. However, even with the importance of this fishery bycatch for the local food security, we cannot disregard the fact that several bycatch fish species are crucial for the balance of the food web and/or has long life history, some with low spawning potential, and high commercial value when adults. In addition, many of them are poorly studied and, for an appropriate evaluation of the fishery in terms of ecosystem management, there is a need for the development of approaches that adapts to scenario of data scarce and often poorly informative (Chrysafi and Kuparinen, 2016; Zhou *et al.*, 2019).

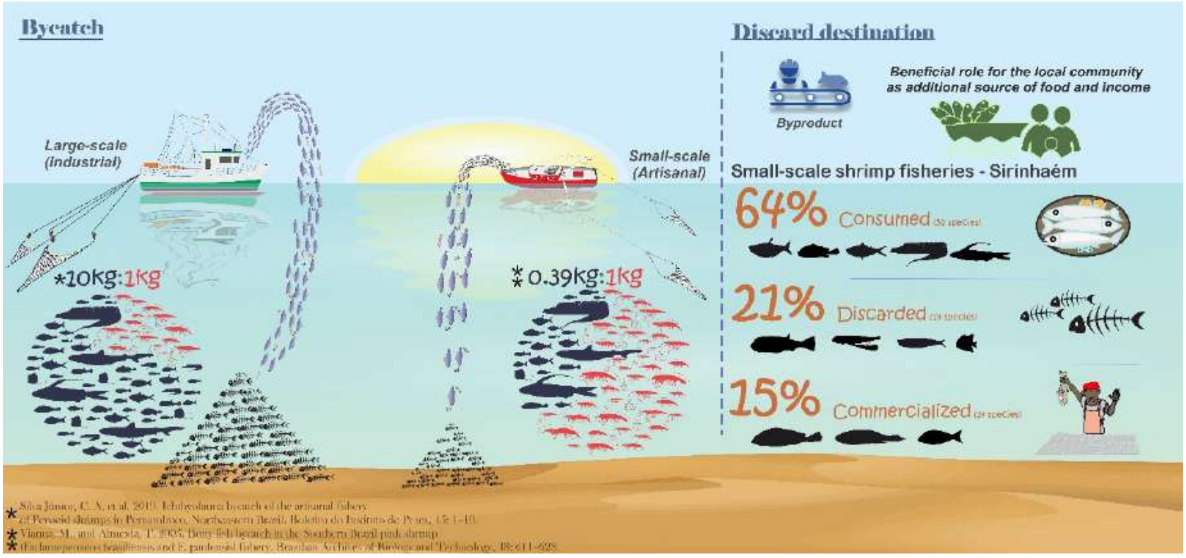


Figure 13. Bycatch volume comparison between large and small-scale bottom trawling fishery; proportion bycatch: shrimp by and discard destination for species caught by small-scale shrimp fisheries in Sirinhaém-Pernambuco coast, Northeastern Brazil.

The absence of a suitable management, in terms of EBFM or EAF, is often related to inability to observe in an integrated the different elements within the ecosystem (physical, biological, economic and social). In this study, we have reported that the main fishing grounds were small and restricted to muddy beds close to the coast. Thus, given its extension, spatial management approaches (e.g., Marine Protect Area – MPA or no-take zones) may be not very effective in a possible fisheries management in the region. The closed season for target species did not display significant improvement to the ecosystem and fishery given the seasonal pattern of the species ("natural closed season"; Lira *et al.* (2021b)). In addition, the effort decrease or the definition of size and gear limitations did not appear to be necessary measures, considering that, according to the traditional stock assessment, the target species are being exploited at biologically accepted levels. The permanent conflict between conservation of species and ecosystem and the need to maintain the income and social condition of fishermen is always discussed, especially in the small-scale fishing. The non-target species are often disregarded in the management measures and given the high socio-economic importance of the bycatch for local community in the region, they need to be better assessed under the Ecosystem Approach to Fishery (EAF) taking into account the effect in whole trophic dynamic and the bycatch sustainability, essential for the food security.

Chapter main findings and Thesis outlook

An integrative study of the fishery was carried out in this first Chapter, encompassing the characteristics of environment and fishing aspects, and the dynamics of the target and bycatch species in order to promote support for ecosystem management. The trawl fishing ground in Sirinham, Northeast Brazil is restricted to muddy beds close to coast and the patterns of abundance and reproduction of the species, as well as the fishing dynamic, is mainly controlled by the environmental drives (e.g., rainfall, chlorophyll) (Figure 14).

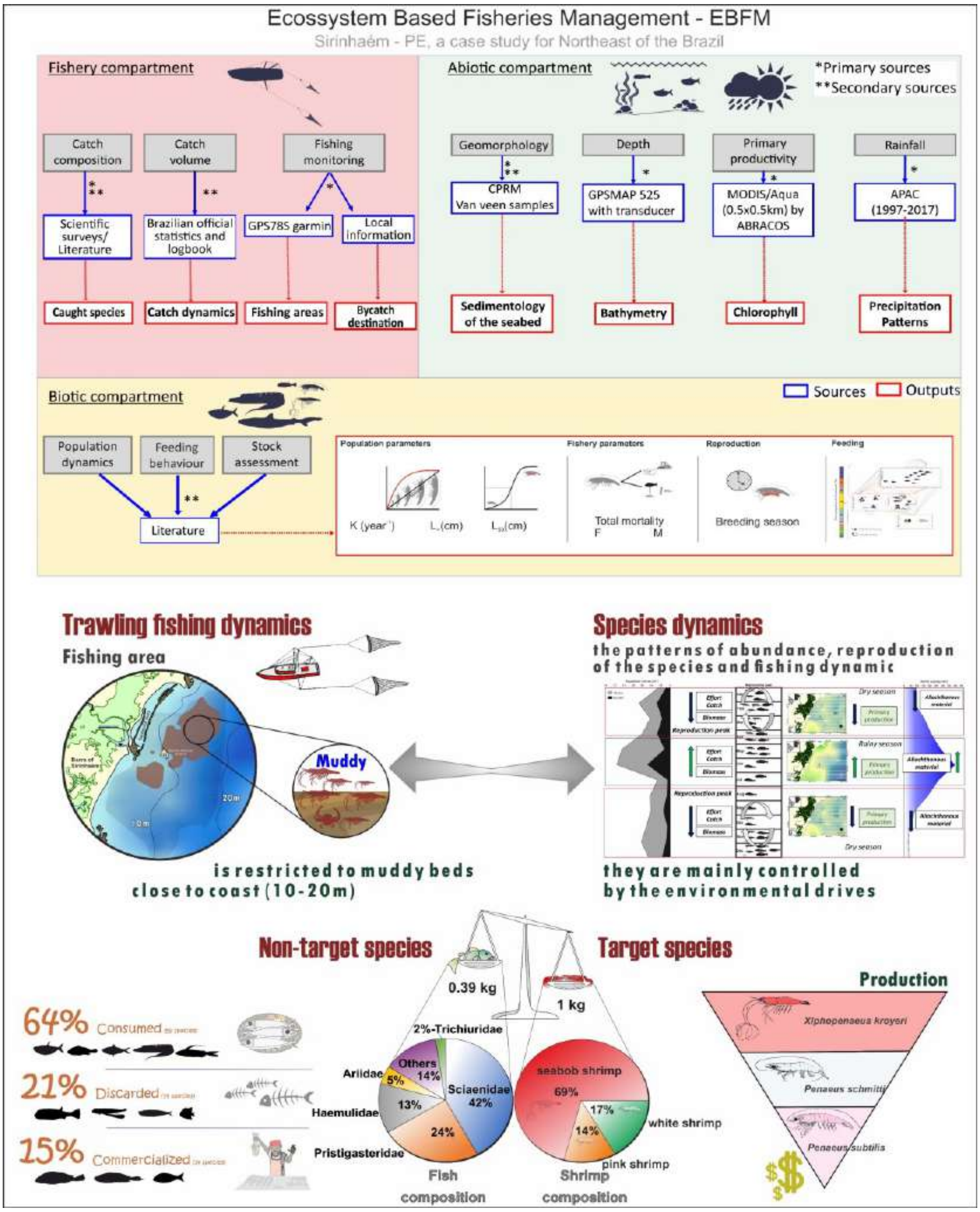


Figure 14. Overview information according by Ecossystem Based Fisheries Management (EBFM) and main findings in terms of dynamic of fishing and species on Sirinhaém-PE coast, Northeastern Brazil.

In the small-scale shrimp trawling fishery carried out in Northeast Brazil, specifically in the Sirinhaém, the amount of bycatch (no-target) caught (in weight) is lower than of shrimp (target). Among the shrimps, the seabob (*Xiphopenaeus kroyeri*), the most abundant one, and the pink (*Penaeus subtilis*) and white shrimp (*Penaeus schmitti*), with high market-values, dominates. Although widely captured, these target species, r strategists with small-size, fast-growing, early maturity, high spawning potential and resilient, according to the traditional stock assessment, they are being exploited at biologically accepted levels.

Among the fish bycatch (93 species regarded as bycatch), those of the Scianidae and Pristigasteridae families were the most important in terms abundance and biomass, with most of them being used by the local community, as additional source of food and income. However, these non-target species are often disregarded into the management measures and, given their high socio-economic importance for local community in the region, they need to be better assessed under the Ecosystem Approach to Fishery (EAF) taking into account their interactions in the trophic chain, essential to evaluate the species conservation and fishery sustainability. In the next three Chapters of thesis it is used, at different levels, the information from the present Chapter as input to evaluate aspects of the ecosystem structure and fishing, taking into account not only target species but also the non-target ones and the lessons learned from the influence of the environmental factors into the fishery and species dynamic.

Chapter 1 also provided a general description of the diet of the main bycatch species, limited by the restrictions of prey quantification. These species potentially play a key role in balancing the ecosystem structure and trophic functioning. In the next Chapter (Chapter 2), we used the combined approaches of stomach content and stable isotopes that has been widely useful for the description of the organism diets, aiming at evaluating the importance of the benthic preys, especially shrimp species as food to coastal fauna, as well as the potential effect caused by trawling on the trophic functioning of the ecosystem.

Chapter 2

Trophic structure of a nektobenthic community exploited by a multispecific bottom trawling fishery in Northeastern Brazil

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Article "*Trophic structure of a nektobenthic community exploited by a multispecific bottom trawling fishery in Northeastern Brazil*"

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RESEARCH ARTICLE

Trophic structure of a nektobenthic community exploited by a multispecific bottom trawling fishery in Northeastern Brazil

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Abstract

We used complementary stable isotope (SIA) and stomach content (SCA) analyses to investigate feeding relationships among species of the nektobenthic communities and the potential ecological effects of the bottom trawling of a coastal ecosystem in Northeastern Brazil. Carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) compositions were determined for five basal sources and 28 consumers, from zooplankton to shrimp and fish species. Fishes and basal sources showed a broad range of $\delta^{15}N$ (fishes: 6.49–14.94‰; sources: 2.58–6.79‰) and $\delta^{13}C$ values (fishes: -23.86 to -13.71‰; sources: -24.32 to -13.53‰), while shrimps and crabs exhibited similar nitrogen and carbon ratios. Six trophic consumer groups were determined among zooplankton, crustaceans and fishes by SIA, with trophic pathways associated mostly with benthic sources. SCA results indicated a preference for benthic invertebrates, mainly worms, crabs and shrimps, as prey for the fish fauna, highlighting their importance in the food web. In overall, differences between SCA and the SIA approaches were observed, except for groups composed mainly for shrimps and some species of high $\delta^{15}N$ values, mostly piscivorous and zoobenthivores. Given the absence of regulation for bottom trawling activities in the area, the cumulative effects of trawling on population parameters, species composition, potentially decreasing the abundance of benthic preys (e.g., shrimps, worms and crabs) may lead to changes in the trophic structure potentially affect the food web and the sustainability of the fishery.

Introduction

Bottom trawling impacts marine habitats in three main aspects: i) physical, due to direct changes in the seabed structure [1], causing the resuspension of sediment (sediment's matrix disruption) and injury or death of many benthic organisms [2–4]; ii) chemical, affecting the organic carbon mineralization [5,6] and re-inserting into the water column possible

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contaminants such as mercury [7]; and iii) biological, mainly given its high level of non-targeted catch [8–10], mostly composed of small sized individuals, usually juveniles [11,12].

In the food web, the fishing activity may act as regulator of the ecosystem, causing adverse ecological effects that could lead to major changes in the trophic interactions among species, consequently to marine habitat degradation [13–16]. Particularly concerning the bottom trawling, direct food web effects are associated to the reduction of species richness and abundance [17–19], however, important indirect consequences are usually disregarded [20]. The capture of non-targeted species by bottom trawling may be a potential risk for the ecosystem sustainability, not only by removing predators of high trophic level, but also prey of lower trophic levels, as the untargeted invertebrates [14,21–23]. For example, a decline in prey availability for demersal fishes, could potentially reduce food intake and body condition [24], causing a trophic cascade effect, changing the ecosystem control equilibrium, either top-down or bottom-up, or even reaching the extreme collapse of the ecosystem [25–27]. In this context, the effect of the predator-prey interactions into the ecosystem trophic structure may be accessed, either by the diet composition and natural markers (such as isotope analysis) [28], and also though ecosystem models (such as Ecopath) [29].

One of the traditional and most accessible ways to address the feeding habits of fish species is by qualitative and quantitative Stomach Content Analysis (SCA) [28–30]. However, often when considering spatial and temporal variations, this approach may be misleading, providing only "snapshots" of the diet [31,32]. On the other hand, Stable Isotope Analysis (SIA) is one of the newest ecological tools in diet studies, providing information that are incorporated in the consumer tissues over a longer period of time [33], indicating resources poorly quantified by stomach contents methods due to regurgitation and digestion rates of preys [34,35]. Although less subject to temporal bias, the SIA approach are influenced, for example, by the type of tissue sampled, lipid concentration, climate season, life stage and size spectrum [36–38].

However, even if SIA and SCA are inherently different techniques, both with considerable assumptions and caveats [39], the use of the these approaches as complementary tools, has been largely recommended [40–43]. For example, increases of $\delta^{13}C$ and $\delta^{15}N$ may be related to the decrease in the biomass of benthic consumers, while the decrease of biomass of benthic preys causes the reduction in the trophic level of the species [45]. Currently, the assessment of the trawling impacts in the food-wed are restricted to SIA, when evaluating changes in carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) compositions and the trophic level of consumers or prey, and to SCA when considering the biomass of the preys [44–46].

Although the Brazilian Northeastern coast covers an extensive area and encompasses a wide range of environments, few studies of coastal trophic structure have been carried out, often focusing only on describing qualitatively and quantitatively the diet [47–50], and in the functioning of the ecosystem [51–53]. Even of great importance, the probable effect of the "disturbance" in the trophic web by fishing, especially those with high impact in the ecosystem (e.g., bottom trawling), has never been focused. Specifically, in Pernambuco, Northeast Brazil, despite the socio-economic relevance of the shrimp fishery, the activity is completely unregulated. Sirinhaém has the largest and most productive motorized fishing fleet among the coastal cities of Pernambuco, corresponding to 50% of the shrimp catch [54], being extremely important as income source for local population [55].

In this study, we investigated the trophic structure of the nektobenthic community exploited by the shrimp trawl fisheries in the State of Pernambuco, Northeastern Brazil, using stable isotopes (SIA) of carbon and nitrogen and stomach content (SCA) analyses. Our main aim is to determine the importance of the target species (shrimps) as prey for non-target spe- cies (bycatch fishes), also discussing the possible effects of the bottom trawling into the trophic interactions, which may affect the marine local community.

Material and methods

Study area and field sampling

In the west coast of the South Atlantic Ocean, mainly in Brazil, shrimps are exploited by a multispecies fishery along the entire coastline, mainly in shallow areas with motorized bottom trawl nets [56], being the Penaeidae the main target [57]. Three fishery systems, which differ in size, technology and volume of catch occur in the Brazilian waters: (i) the industrial fleet operating mainly in the North region (Amazon river estuarine system), Southeast and South Brazil; (ii) a semi-industrial fleet distributed from north to south of the country with similar technology of the artisanal fleet but with greater fishing power and catches; and (iii) artisanal fleet that operates along the entire coast, but specially in Northeast, characterized by higher number of people involved; low level of technology, capture and profit [58]. This later fishery system is present in our study area, Sirinhaém. This fishery has the proportion of fish bycatch: shrimp as 0.39:1 kg [59]. The fish bycatch is composed of 51 species, 38 genera and 17 families, primarily Pristigasteridae, Sciaenidae and Haemulidae, mostly zooplanktivore and zoobenthivore (e.g., *Pellona harroweri*, *C. bleekerianus*, *Isophistus parvipinnis*, *Stellifer microps*, *Larimus breviceps*, *P. brasiliensis*, *C. nobilis* and *Haemulopsis corvinaeformis*), which are often used as a byproduct (commercially valuable species) or consumed by the crew and local communities [59].

The coastal waters are influenced by nutrient supply from the Sirinhaém river, the climate is tropical, with a rainy season occurring between May and October. In terms of environmental condition, the rainfall ranges monthly from 20 to 450 $\text{mm}\cdot\text{yr}^{-1}$, the mean water surface temperature is 29˚C, and the pH and salinity range between 8.0 and 8.7 and 23–37, respectively [60,61]. The shrimp fishery is artisanal and carried out near the coast [62] between 8 and 20 m depth, mainly inside or close to the Marine Protected Area of Guadalupe, around of Santo Aleixo Island, distant from 1.5 to 3 miles off the coast (Fig 1).

Surveys to collect macroalgae, bycatch fishes and invertebrates (except zooplankton) were carried out quarterly with the approval by the Brazilian authorities, such as the Navy and the

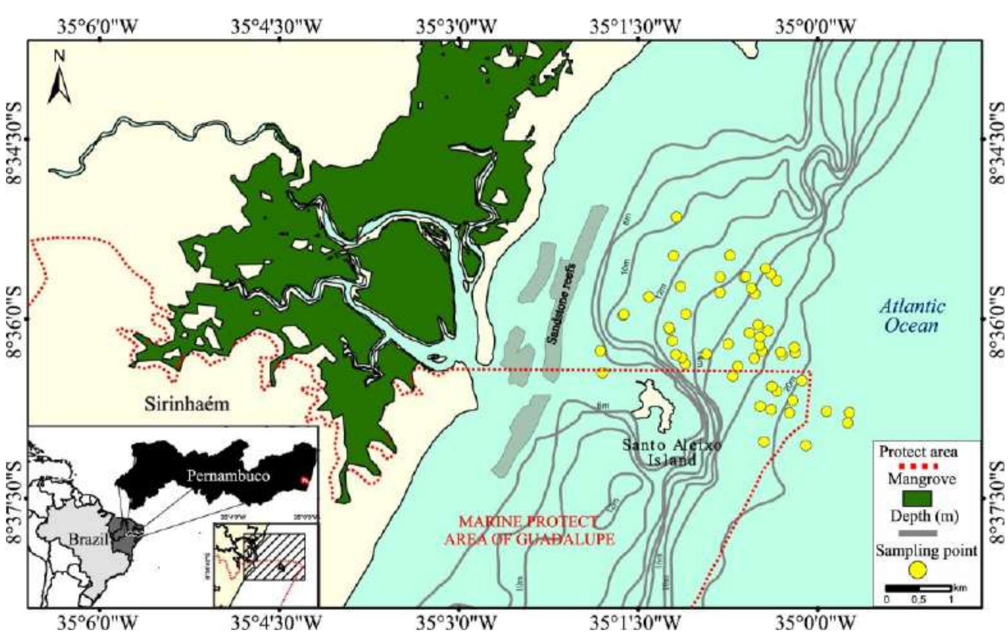

Fig 1. Study area located on the Pernambuco coast in Northeast Brazil. The Sirinhaém area, located on the Pernambuco coast in Northeast Brazil. Depth was obtained from [63].

https://doi.org/10.1371/journal.pone.0246491.g001

Ministry of the Environment (Sisbio—License n˚34125), between 2014 and 2015 using the commercial bottom trawl fishing (length: 10 m; horizontal opening: 6.10 m; mesh size body: 30 mm; mesh size cod end: 25 mm). It was not required the approval by the Brazilian animal ethics committee, since species collected arrive dead onboard without any method of sacrifice and within the authorized fishery activity. In order to improve the data samples with other consumers of the bycatch not previously sampled, complementary data collections were carried out in October to December 2019 (see S1 Table for detail).

At each month, three trawls were performed during the daytime, between 10 and 20 m depth, for about 2 hours, with boat velocity varying between 1.6 and 3.7 knots. Zooplankton was sampled with a 300 μm mesh size plankton net hauled horizontally for 10 minutes at subsurface. In addition, basal food sources included suspended Particulate Organic Matter (POM) obtained by filtering 0.5–1.0 L of water through fiberglass filters (0.75 μm) and Sediment particulate Organic Matter (SOM) collected at low tide in a shallow area near the island from the top 2 mm layer of sediment using a tube core (2 cm of diameter) [37]. All compartments sampled and specimens caught were at once put on ice, then transported to the laboratory and stored in a freezer (-18˚C) until the analysis. In laboratory, they were identified to species level and measured (standard length–SL for fishes and carapace length/diameter for shrimps and blue crabs).

Data analysis

Muscle samples (about 0.5g) from each fish, squid, blue crab and shrimp species were extracted, rinsed with distilled water to remove exogenous materials (e.g., remaining scales, bones and carapace). For POM, SOM and zooplankton (which comprehended only copepods), the whole organism/sample was used. Samples were dried in an oven at 60˚C for 48h. Then, they were ground into a fine powder with a mortar and pestle.

POM, SOM and zooplankton samples were duplicated. The inorganic carbon was removed by acidification process prior to the $\delta^{13}C$ analysis [64]. The sub-samples that were not acidified were analyzed for $\delta^{15}N$ [31]. Samples were analyzed by continuous flow on a Thermo Scientific Flash EA 2000 elemental analyzer coupled to a Delta V Plus mass spectrometer at the Pôle Spectrométrie Océan (Plouzané, France). Results are expressed in standard δ notation based on international standards (Vienna Pee Dee Belemnite for $\delta^{13}C$ and atmospheric nitrogen for $\delta^{15}N$) following the equation:

$$\delta^{13}\mathrm{C} \text{ or } \delta^{15}\mathrm{N} = \left[\left(\frac{R_{sample}}{R_{standard}}\right) - 1\right] \times 10^3 \text{ (in ‰), where R is } {}^{13}\mathrm{C}/{}^{12}\mathrm{C} \text{ or } {}^{15}\mathrm{N}/{}^{14}\mathrm{N} \text{ (eq.1)}$$

Reference materials of known $\delta^{15}N$ and $\delta^{13}C$ were analyzed: USGS61, USGS62 and USGS63. The recommended values of the standards were reproduced within the confidence limits. For every six samples, a home standard (Thermo Acetanilide) of experimental precision (based on the standard deviation of the internal standard replicates) was used, indicating an analytical precision of ± 0.11‰ for $\delta^{13}C$ and ± 0.07‰ for $\delta^{15}N$.

The carbon and nitrogen values of basal food sources and consumers of different trophic guilds [65] in Sirinhaém coast were investigated by the biplot of mean $\delta^{13}C$ (±Standard deviation (SD)) and $\delta^{15}N$ (±SD) values of each group/species. Due to the non-normality (Kolmogorov-Smirnov test) and non-homogeneity of variance (Bartlett test), the statistical significance of differences between individual $\delta^{13}C$ and $\delta^{15}N$ values of food sources, shrimp and fish bycatch species was assessed with the non-parametric Kruskal-Wallis test and pairwise multiple comparisons tested for subsequent comparisons in case of significant differences (*p-value*<0.05) [66].

From the mean values of $\delta^{13}C$ and $\delta^{15}N$ (objects) for each consumer species (descriptors), an Agglomerative Hierarchical Cluster (AHC) using the Ward's minimum variance method based in Euclidian similarity resemblance matrix was performed in order to identify trophic groups of species [67,68]. To determine optimal number of clusters, the NbClust method pro- posed by Charrad et al. [69] was carried out. This method provides 30 indices to evaluate the relevant number of Clusters. In addition, the trophic groups obtained with AHC were compared using a Nonparametric multivariate permutational analysis of variance (PERMANOVA) [70]. All statistical analyses were performed considering a 5% significance level.

Stomach Content Analysis (SCA) were accessed for 52% of species (13 species, 52% of the total) caught in the same area, including fishes and shrimps from unpublished laboratory database, except *Conodon nobilis* [71]. For the remaining species (12), diet information was obtained from literature and detailed in the Tables 2 and S2. For local collected species, the stomachs were removed and weighed to the nearest 0.01 g and fixed in 10% formaldehyde within 48 h and then conserved in 70% alcohol. The contents of the individual stomachs were sorted, counted, weighed (g), and identified to the lowest possible taxonomic level.

To describe the diet composition of the consumers, the stomach content items were gath- ered in 9 prey groups (detritus, phytoplankton, zooplankton, worm, crab, mollusk, other crustaceans, shrimp and fish). The similarity of diet among species was accessed by AHC as explained earlier, using prey weight proportion (objects; %W) [55] for each consumer (descriptors).

To provide an overview comparison among SIA and SCA, the stomach contents data was graphically displayed through heatmaps (consumer x prey) along with a AHC, using prey weight proportion (%W) [72] for each consumer. In the heatmap approach, the individual values contained in a matrix were represented as color ramp within a range of %W value scale. In addition, the hierarchical cluster obtained from SIA was compared graphically to SCA and quantified by Baker's Gamma Index (BGI) with permutation test [73,74] to identify the possible level of similarity among the dendrograms, and consequently the two approaches. BGI value ranges from -1 to 1, values close to 0 represents statistic difference between the two dendrograms ($p<0.05$), and values close to -1 and 1 reveals identical dendrogram.

All analyses were performed using the R environment [75], with packages vegan [76], cluster [77], NbClust [69] and dendextend [73] for the estimation the clusters, to identify the optimum cluster number and to measure the association between the two trees of hierarchical clustering respectively. Additionally, ggplot2 [78] and gplots [79] were used to generate graphics.

Results

Stable isotope compositions were analyzed in six invertebrate species and eighteen consumers —fish (167 samples), one zooplankton group (6 samples) and five basal sources (31 samples) (Table 1). Fishes and basal sources showed a broad range of $\delta^{15}N$ (fishes: 6.49–14.94‰; sources: 2.58–6.79‰) and $\delta^{13}C$ values (fishes: −23.86 to −13.71‰; sources: −24.32 to −-13.53‰), while shrimps and *Callinectes* species exhibited similar values of nitrogen and carbon ratios (Table 1).

Basal sources exhibited significant difference within the medians for both $\delta^{13}C$ values (Kruskal-Wallis: $\chi^2 = 17.814$, *p-value* = 0.001) and $\delta^{15}N$ (Kruskal-Wallis: $\chi^2 = 23.668$, *p-value* < 0.001) (Fig 2), for example between POM and SOM in $\delta^{15}N$, and the macroalgae *Lobophora variegate* and *Gracilaria cervicornis* in $\delta^{13}C$. The medians of $\delta^{13}C$ values for the three shrimp species (*Penaeus subtilis*, *P. schmitti* and *Xiphopenaeus kroyeri*) were similar (Kruskal-Wallis: $\chi^2 = 1.555$, *p-value* = 0.459), as well as for $\delta^{15}N$ values (Kruskal-Wallis: $\chi^2 = 2.6428$, *p-value* =

Table 1. Stable isotopes compositions of basal sources and consumers.

Groups/species	Code	Guilds	N	$\delta^{13}C$ (‰)	Min-Max	$\delta^{15}N$ (‰)	Min-Max
Basal sources							
Sedimentary organic matter	SOM	-	8	-16.51 ± 0.60	[-17.35 to -15.84]	3.67 ± 0.55	[2.85 to 4.37]
Lobophora variegata	lob.var	-	6	-15.02 ± 0.84	[-15.74 to -13.53]	4.36 ± 0.44	[3.88 to 4.89]
Gracilaria cervicornis	gra.cer	-	6	-21.98 ± 1.92	[-24.32 to -18.63]	4.44 ± 1.09	[3.59 to 6.58]
Sargassum sp.	sar.sp	-	6	-17.50 ± 1.41	[-19.34 to -15.69]	4.44 ± 0.24	[4.07 to 4.73]
Particulate organic matter	POM	-	5	-21.60 ± 0.65	[-22.35 to -20.61]	6.39 ± 0.36	[5.90 to 6.79]
Invertebrates							
Zooplankton	zoo	Filter-feeder	6	-18.65 ±0.51	[-19.32 to -17.84]	7.26 ±1.14	[6.45 to 9.49]
Penaeus subtilis	pen.sub	Omnivore	14	-16.71 ±1.89	[-21.59 to -14.69]	8.83 ±2.19	[7.38 to 11.72]
Penaeus schmitti	pen.sch	Detritivore	20	-16.29 ±1.18	[-18.45 to -13.60]	8.98 ±1.51	[6.85 to 11.18]
Callinectes danae	cal.dan	Omnivore	5	-15.14 ±0.61	[-16.01 to -14.45]	9.07 ±0.62	[8.52 to 9.75]
Callinectes ornatus	cal.orn	Omnivore	3	-14.87 ±0.67	[-15.41 to -14.12]	9.27 ±0.86	[8.47 to 10.18]
Xiphopenaeus kroyeri	xip.kro	Omnivore	17	-15.95 ±0.59	[-17.01 to -15.14]	9.27 ±0.48	[8.05 to 9.76]
Lolliguncula brevis	lol.bre	Piscivore/Zoobenthivore	5	-16.77 ±0.17	[-16.91 to -16.58]	12.60 ± 0.10	[12.53 to 12.75]
Fishes							
Citharichthys spilopterus	cit.spi	Zoobenthivore	3	-21.59 ±2.65	[-23.86 to -18.68]	8.85 ±1.59	[7.91 to 10.68]
Diapterus auratus	dia.aur	Zoobenthivore	7	-17.52 ±2.88	[-21.44 to -13.71]	8.84 ±1.23	[7.74 to 11.47]
Opisthonema oglinum	opi.ogl	Zooplanktivore	8	-17.07 ±0.47	[-17.60 to -16.19]	9.58 ±1.01	[8.35 to 11.83]
Symphurus tessellatus	sym.tes	Zoobenthivore	6	-21.56 ±1.54	[-23.20 to -19.08]	9.69 ±1.22	[8.71 to 11.86]
Diapterus rhombeus	dia.rho	Zoobenthivore	8	-19.22 ±2.19	[-22.50 to -17.06]	9.71 ±1.49	[7.11 to 11.41]
Lutjanus synagris	lut.syn	Zoobenthivore	6	-15.74 ±0.81	[-16.77 to -14.75]	10.21 ± 1.50	[8.71 to 11.76]
Bairdiella ronchus	bai.ron	Zoobenthivore	3	-16.02 ±0.08	[-16.11 to -15.95]	10.54 ±0.1	[10.36 to 10.70]
Chirocentrodon bleekerianus	chi.ble	Zoobenthivore	4	-16.84 ±0.23	[-17.15 to -16.64]	10.59 ±0.80	[8.28 to 11.81]
Eucinostomus argenteus	euc.arg	Omnivore	14	-16.11 ±1.21	[-18.87 to -14.99]	10.98 ±1.51	[6.49 to 13.19]
Bagre bagre	bag.bag	Zoobenthivore	3	-16.28 ±0.09	[-16.38 to -16.21]	11.62 ±0.40	[11.13 to 11.93]
Caranx hippos	car.hip	Piscivore	8	-17.13 ±1.56	[-19.73 to -15.83]	11.75 ±0.50	[10.36 to 10.70]
Micropogonias furnieri	mic.fur	Omnivore	7	-16.59 ±1.32	[-18.18 to -15.12]	12.07 ±0.60	[11.15 to 12.82]
Bagre marinus	bag.mar	Zoobenthivore	8	-16.18 ±0.23	[-16.59 to -15.84]	12.18 ±0.70	[11.33 to 13.47]
Larimus breviceps	lar.bre	Zoobenthivore	3	-16.29 ±0.51	[-16.61 to -15.7]	12.19 ±1.00	[11.18 to 13.18]
Stellifer microps	ste.mic	Zoobenthivore	4	-16.26 ±0.81	[-17.32 to -15.44]	12.21 ±1.60	[10.40 to 13.64]
Isopisthus parvipinnis	iso.par	Piscivore	4	-15.93 ±0.28	[-16.15 to -15.56]	12.50 ±0.19	[12.33 to 12.74]
Conodon nobilis	con.nob	Piscivore/Zoobenthivore	4	-15.58 ±0.31	[-15.93 to -15.21]	12.71 ±1.50	[11.45 to 14.94]
Paralonchurus brasiliensis	par.bra	Zoobenthivore	3	-15.20 ±1.20	[-16.58 to -14.44]	12.89 ±1.60	[11.23 to 14.45]

Groups/species names, codes, trophic guilds, numbers of samples (n), $\delta^{13}C$ means ± standard deviation, minimum and maximum, $\delta^{15}N$ mean ± standard deviation, and minimal and maximum of basal sources and consumers (invertebrates and fishes) sampled off the Sirinhaém coast, Northeastern Brazil.

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0.266). Significant differences were observed in $\delta^{15}N$ and $\delta^{13}C$ values (Kruskal-Wallis: χ^2 = 63.44, *p-value* < 0.001; χ^2 = 52.083, *p-value* < 0.001 respectively) for fish species, mostly due to *Citharichthys spilopterus*, *Symphurus tesellatus*, *Eucinostomus argenteus* and *Diapterus auratus* which showed the more depleted $\delta^{15}N$ and $\delta^{13}C$ values.

Among the basal sources, POM and SOM had maximum and minimum $\delta^{15}N$ values respectively (6.79 and 2.85‰), while *G. cervicornis* and *L. variegata* showed the most depleted and enriched $\delta^{13}C$ values, respectively (Fig 2). Between consumers, flatfish species (*C. spilopterus* and *S. tesellatus*) had the most depleted $\delta^{13}C$ values and blue crab species (*Callinectes danae* and *C. ornatus*) were the most enriched. For the $\delta^{15}N$ rates, zooplankton had the lowest, while *Conodon nobilis*, *Paralonchurus brasiliensis* and *Lolliguncula brevis* showed the highest values (Fig 2).

Table 2. Weight contribution (%) of each prey group in the diet of consumers off the Sirinhae´m coast, Northeastern Brazil.

Consumers	Weight contribution of preys (%W)									
	Det	Phy	Zoo	Cra	Shr	Wor	Mol	Oth.crus	Fis	Sources
Zooplankton (zoo)	0.15	0.80	0.05							[80]
Penaeus subtilis (pen.sub)	0.12	0.08	0.30			0.30		0.20		unpublished data
Penaeus schmitti (pen.sch)		0.50	0.06			0.24		0.20		unpublished data
Callinectes danae (cal.dan)	0.04			0.26	0.01	0.35	0.23	0.03	0.08	[81]
Callinectes ornatus (cal.orn)	0.1	0.04	0.02	0.25	0.18	0.04	0.12	0.02	0.22	[82]
Xiphopenaeus kroyeri (xip.kro)	0.22	0.07	0.37	0.03	0.11	0.08	0.05	0.04	0.05	unpublished data
Lolliguncula brevis (lol.bre)	0.15	0.01	0.01	0.24	0.24	0.01	0.02		0.32	[83]
Citharichthys spilopterus (cit.spi)			0.09	0.02	0.21	0.29		0.01	0.38	[84]
Diapterus auratus (dia.aur)			0.01			0.96	0.01		0.02	unpublished data
Opisthonema oglinum (opi.ogl)	0.05	0.42	0.41					0.11	0.01	[85,86]
Symphurus tessellatus (sym.tes)			0.31	0.01	0.03	0.66				[84]
Diapterus rhombeus (dia.rho)		0.02	0.82			0.16				unpublished data
Lutjanus synagris (lut.syn)		0.01	0.16	0.39	0.18	0.05		0.11	0.10	[87]
Bairdiella ronchus (bai.ron)	0.04			0.18	0.22			0.26	0.29	unpublished data
Chirocentrodon bleekerianus (chi.ble)	0.01		0.12	0.10	0.32	0.01		0.30	0.14	[88]
Eucinostomus argenteus (euc.arg)	0.02		0.13	0.03	0.14	0.52		0.02	0.14	unpublished data
Bagre bagre (bag.bag)		0.01	0.01	0.21	0.23	0.15			0.39	[89]
Caranx hippos (car.hip)	0.02		0.02	0.01	0.15	0.01	0.01	0.22	0.57	unpublished data
Micropogonias furnieri (mic.fur)			0.02		0.35	0.60			0.03	[90]
Bagre marinus (bag.mar)	0.12	0.03		0.54	0.14	0.01		0.02	0.15	unpublished data
Larimus breviceps (lar.bre)	0.03		0.01		0.80	0.16				[91]
Stellifer microps (ste.mic)				0.19	0.60	0.02	0.01	0.06	0.02	unpublished data
Isopisthus parvipinnis (iso.par)				0.01	0.16			0.01	0.82	unpublished data
Conodon nobilis (con.nob)					0.62	0.01			0.31	unpublished data
Paralonchurus brasiliensis (par.bra)					0.40				0.01	unpublished data

The values represent the percentage of weight contribution of each prey group. Acronyms for each prey are: Det–Detritus; Phy–Phytoplankton; Zoo–Zooplankton; Cra–Crab; Shr–Shrimp; Wor–Worm; Mol–Mollusc; Oth.cru—Other crustaceans and Fis–Fish.

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Cluster analysis performed on mean stable isotope ratio values for the consumer group significantly gathered species in 3 main groups (GR), divided on 2 to 3 sub-groups (Fig 2 inset) (PERMANOVA: F = 49.12; *p-value* < 0.001). Zooplankton, the only member of GR6, had the lowest $\delta^{15}N$.

Fish species associated to the seabed had relatively lower $\delta^{13}C$ compared to the others and were separated into two groups, mojarras (*D*. *rhombeus* and *D*. *auratus*; GR5) and flatfish species (*S*. *tesselatus* and *C*. *spilopterus*; GR4) (Fig 2). The cluster GR3 regrouped the species of highest $\delta^{15}N$ values, greater than 11‰, as piscivorous and zoobenthivore, while GR2 represented zooplanktivore, omnivore and zoobenthivore fishes of intermediate values of carbon ($\delta^{13}C$: -17.04 to -15.74‰) and nitrogen ($\delta^{15}N$: 9.58 to 10.98‰) (Fig 2 and Table 1). GR1 gathered the omnivorous or detritivores invertebrates, as shrimp and blue crab, with low $\delta^{15}N$ values and enriched $\delta^{13}C$ (Fig2).

The diet description of the 25 consumers species/groups through SCA may be accessed in Table 2. Omnivorous and detritivores species, including shrimp (e.g., *P*. *schmitti*) and blue crabs (e.g., *C*. *ornatus*), showed high trophic plasticity, feeding from phytoplankton to fishes in proportions ranging, in average, from 8 to 25% for each group of prey (Table 2). Omnivorous fishes (e.g., *E*. *argenteus* and *Micropogonias furnieri*) were an exception, feeding predominantly

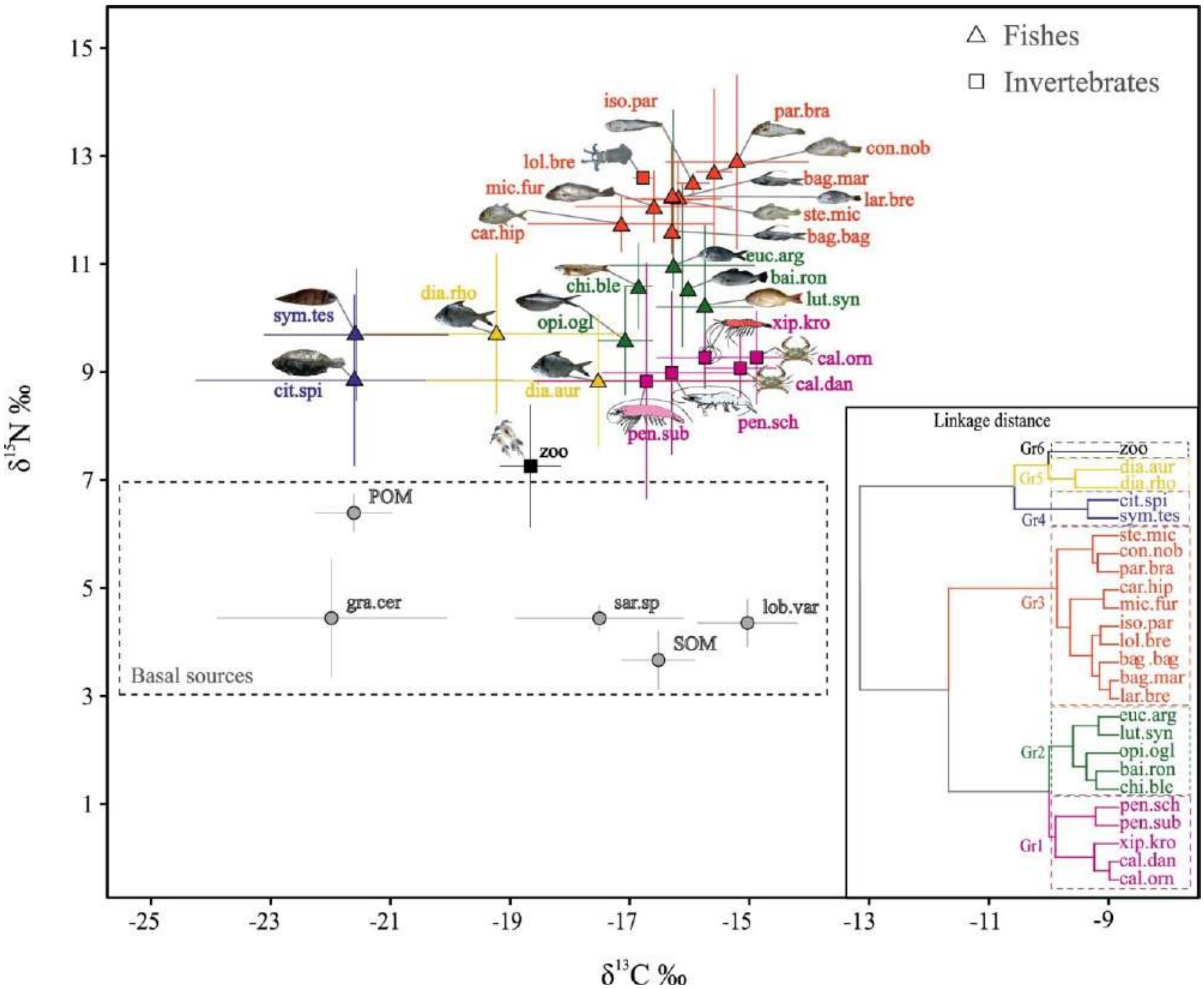

Fig 2. Biplot of carbon and nitrogen for basal sources and consumers. Biplot of $\delta^{13}C$ (‰) and $\delta^{15}N$ (‰) values (mean ± SD) for basal sources (grey circles) and consumers (invertebrates and fishes) sampled off the Sirinhaém coast, Northeastern Brazil. The dendrogram inserted in the right corner is from agglomerative hierarchical clustering (AHC) for 25 consumers representing the trophic groups, indicated by colours, where each node represents an individual species. Species abbreviations are: Sedimentary organic matter (SOM), *Lobophora variegata* (lob.var), *Gracilaria cervicorni* (gra.cer), *Sargassum* sp.(sar.sp), Particulate organic matter (POM), Zooplankton–(zoo), *Penaeus subtilis* (pen.sub), *Penaeus schmitti* (pen.sch), *Callinectes danae* (cal.dan), *Callinectes ornatus* (cal.orn), *Xiphopenaeus kroyeri* (xip.kro), *Lolliguncula brevis* (lol.bre), *Citharichthys spilopterus* (cit.spi), *Diapterus auratus* (dia.aur), *Opisthonema oglinum* (opi.ogl), *Symphurus tessellatus* (sym.tes), *Diapterus rhombeus* (dia.rho), *Lutjanus* synagris (lut.syn), *Bairdiella ronchus* (bai.ron), *Chirocentrodon bleekerianus* (chi.ble), *Eucinostomus argenteus* (euc.arg), *Bagre bagre* (bag.bag), *Caranx hippos* (car.hip), *Micropogonias furnieri* (mic.fur), *Bagre marinus* (bag.mar), *Larimus breviceps* (lar.bre), *Stellifer microps* (ste.mic), *Isopisthus parvipinnis* (iso.par), *Conodon nobilis* (con.nob) and *Paralonchurus brasiliensis* (par.bra).

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on benthic fauna, as shrimp and worms, totalizing 60% and 95% of their diet, respectively (Table 2), while *Opisthonema oglinum*, classified as zooplanktivore, fed mainly on phytoplank- ton and zooplankton, which represented 83% of the diet (Table 2).

Shrimps, fishes, and worms were the main preys, contributing on average 50% of the stomach content of fishes and squids (*L. brevis*) (Table 2). In this group, *P. brasiliensis* was an excep- tion, with a diet composed basically of detritus (58%) and shrimp (40%), similar to detritivorous species. Species classified as piscivores, *Caranx hippos* and *Isopisthus parvipinnis*, presented high percentage of fish in their diet, 82% and 57% respectively (Table 2).

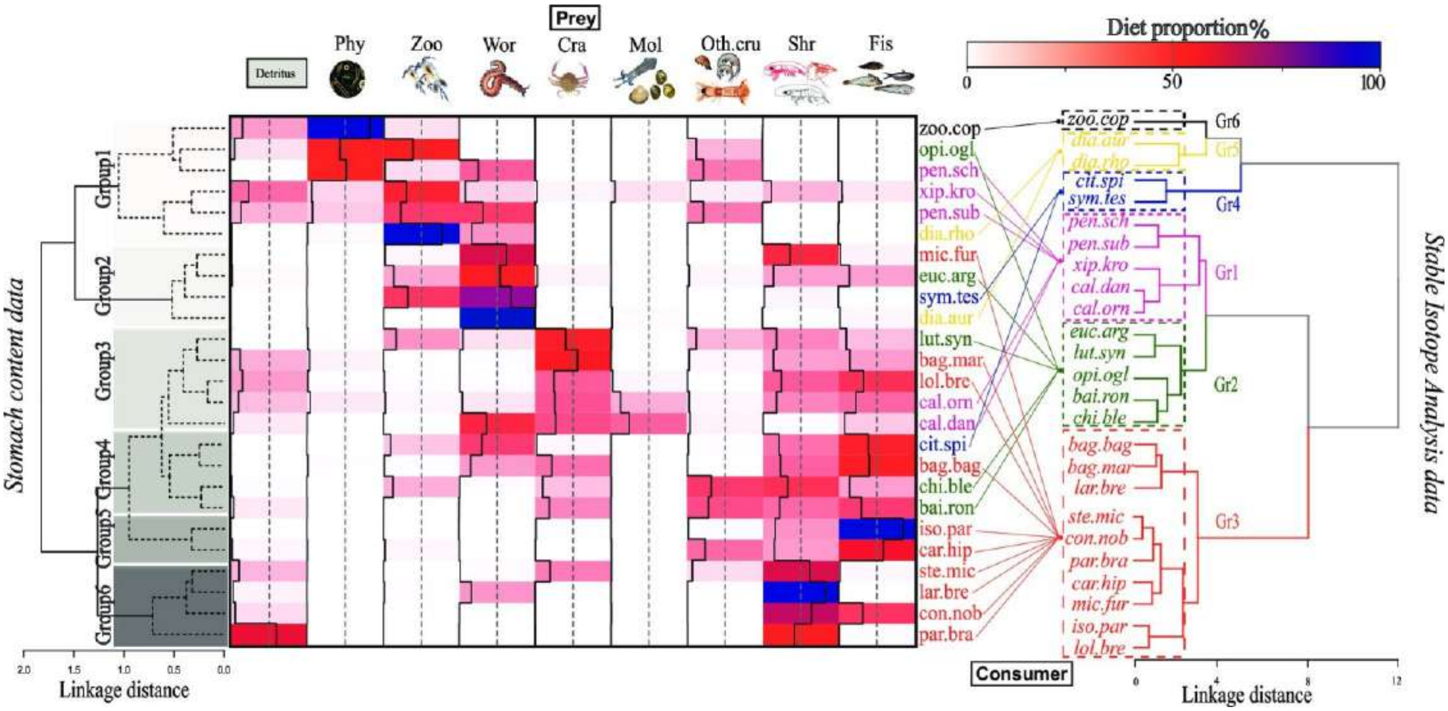

Fig 3. Heatmap of the diet proportion among consumers and prey. The dendrograms inserted in the corners were made with agglomerative hierarchical clustering (AHC) based on diet proportion by stomach content data (left) and isotope composition data (right) off the Sirinhaém coast, Northeastern Brazil. The grey boxes represent different groups based on stomach content data. Consumer abbreviations are given in Table 1 and colours based on clustering by isotope composition data. Acronyms for each prey are: Det–Detritus; Phy–Phytoplankton; Zoo–Zooplankton; Cra–Crab; Shr–Shrimp; Wor–Worm; Mol–Mollusc; Oth.cru—Other crustaceans and Fis–Fish.

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Cluster analysis of SCA emphasized 6 significantly different main consumer groups (Fig 3) (PERMANOVA: F = 6.50; *p-value* < 0.001). Group 1 (six species) had diet based mainly on detritus, phytoplankton and zooplankton and worms, while the second group was composed of four species (e.g., flatfish and croaker) that fed mainly on worms (Fig 3 left). The group 3 (five species) and group 4 (four species) (e.g., *Bagre marinus*, *Chirocentrodon bleekerianus* and

L. synagris), showed considerable variability in dietary items in the stomach contents dominated by crustaceans and fishes (Fig 3). In the last clusters of two (Group 5) and four species (Group 6), composed by piscivores or zoobenthivore species of high $\delta^{15}N$ values (Fig 2 and Table 1), the main preys were fish or shrimps (Fig 3).

The species with high $\delta^{15}N$ values (e.g., *P. brasiliensis*, *C. nobilis* and *C. hippos*), as well as shrimps (*P. schmitti*, *P. subtilis*, *X. kroyeri*) showed a similar grouping between the two approaches (SIA and SCA). However, in overall, differences in diagram clusters between stomach contents and the SIA approach were observed (Baker's Gamma correlation coefficient = 0.20). Some species presented large grouping differences between the two approaches, mainly for species of the GR4 (e.g., *C. spilopterus*, and *S. tesselatus*) and zoobenthivores of the GR2 (e.g., *O. oglinum*, and *E. argenteus*), based in SIA clusters (Fig 3).

Discussion

The trophic ecology has long been assessed from diet composition to evaluate level of complexity, health and alterations of communities on aquatic ecosystems (e.g., rivers, estuaries, reefs and deep oceans) [47,92–95]. Additional tools as the trophic natural markers provide information on the assimilated food, while the traditional approach of diet composition is based only on food intake. Comparing the two approaches improves the description and potentially minimizes errors in measuring the organism diets. Thus, by applying complementary methods—stable isotope and stomach content composition—we examined the trophic structure of a tropical ecosystem affected by shrimp bottom trawling, aiming to evaluate the importance of the shrimp species as food to coastal fauna and how the fishery exploitation of these resource may affect the ecosystem trophic functioning.

Firstly, some considerations should be made before the interpretation of our results. Although we have used most data from the study area and similar periods, we also utilized stomach content data from the literature, as proxy of the diet of some local species, which did not allow a direct comparison between methods (SCA and SIA), but rather a complementary approach. In addition, we decided not to apply the models to quantify the source importance in isotope approach (e.g., bayesian mixing model), given that our sampling did not take into account some of the known basal sources and benthic invertebrates, which could lead to potential misinterpretation of our results and conclusion as reported by [96]. Therefore, the results presented here are not intended to exhaustively describe the trophic dynamic of the study, but, despite their limitations, we were able to identify the predator and prey groups with major roles in the food-web, and how they could influence the ecosystem trophic dynamic in response to the shrimp fishery in Sirinhaém, Northeast Brazil.

Differences on isotopic ratios occurred between SOM and POM. These variations among basal sources are expected [97] and reflects, for example, different contributions to organic deposition in coastal sediments [98–100], which can be seasonally intensified with the increase of fluvial discharges during periods of heavy precipitation [101]. These differences allow the discrimination of two trophic pathways based on benthic or pelagic sources [102]. However, it usually can result in high range of isotopes ratios, given the high diversity of trophic guilds, [103,104]. In general, we found differences and similarities between SCA and the SIA approaches. For example, for shrimps and species of high δ^{15}N values, mostly piscivorous and zoobenthivores, the two approached converged. However, we noticed some mismatches in our results for some zooplanktivore (e.g., *O. oglinum*), omnivore (e.g., *C. ornatus* and *C. danae*) and zoobenthivores species (e.g., *B. marinus*, *L. synagris* and *Bairdiella ronchus*). Generalist trophic habits associated with omnivores that feed on multiple trophic levels and taxonomic groups, introduce considerable uncertainty into diet patterns by SCA and SIA [105], mainly related to age-dependent trophic shifts [106]. Some studies report wide variations and even lack of correlation between SIA and SCA approaches [35,39,42], mainly related to aspects of differential size range [107], life stage [105], season [108], isotopic fractionation [109] and spa- tial-temporal scale [34].

For some zoobenthivores, isotopic niches often overlap with piscivorous [110], reflecting the opportunistic behavior of this group in an environment where food sources are highly available. Zoobenthivore fishes had wide feeding preferences [65,111], which would possibly provide large variations of δ^{15}N composition [112,113]. However, the nitrogen ratios for this group slightly varied, indicating that they feed on food sources that have similar isotopic composition, consisting mostly of penaeid shrimps, small crabs and fishes in lower proportion. The availability and consequently the aggregation of prey can strongly influence the species feeding habitat patterns [114,115]; the predator would feed on prey largely available. Penaeidae shrimps are widely explored in the region, particularly the seabob shrimp (*X. kroyeri*), the most abundant one, and the pink (*P. subtilis*) and white shrimp (*P. schmitti*), with high mar- ket-values [62]. Although we have not evaluated the worms isotopic compositions, fish diet revealed a relative high contribution of this taxonomic group, mostly polychaets for some spe- cies (e.g., *Eucinostomus argenteus*–present study and *Symphurus tesselatus* – Guedes et al.

[84]). Thus, polychaets should be considered as an additional important source of energy for the higher trophic levels.

Our findings with two complementary tools (SCA and SIA) helped to understand the contribution of benthic sources, the importance of crustaceans, especially shrimps, in transporting energy from food web base to upper trophic levels and bycatch species of high $\delta^{15}N$ values, such as the top predators (e.g., *I. parvipinis* and *C. nobilis*), thus providing support to coastal food-web in Sirinhaém. The importance of the benthic community for the trophic functioning of the coastal zone, specifically crustaceans, has been reported in other ecosystems affected by bottom trawl fishing, for example, in southeast Brazil [116–120], and in other parts of the world, such as Australia [121], Irish Sea [24] and North Sea [122]. The presence of large mud banks in these coastal areas, which usually favors large occurrences of benthic invertebrates, such as worms and crustaceans, explains this huge importance. In our study case, the fishing area in Sirinhaém is close to river mouth with depths ranging from 4 to 20 m, the seabed is composed of sand and predominantly mud zones, where most of the organisms and fishing effort is homogeneously concentrated. Hinz et al. [45] highlighted the negative effect of fishery trawling, removing not only fish and benthos, but also changing prey and predator relation- ships. The resuspension of sediment from trawling may cause death of a wide range of benthic organism [13], including benthic invertebrate preys of major role in energy transfer for the food-web, as for example in our case, the shrimps (e.g., *X. kroyeri*, *P. subtilis* and *P. schmitti*), crabs (e.g., *C. ornatus* and *C. danae*) and worms. The food-web dependence of the benthic invertebrates should also be considered in ecosystem approach to fisheries, since any regula- tion may therefore have consequences on both benthic prey and the consumers [45,123].

Specifically in Sirinhaém, since there are no fishing regulations [59], the cumulative effects of trawling on population parameters (e.g., size and food intake), species composition [124,125], potential decreasing the abundance of benthic preys and fish species may lead to intense changes in the trophic structure of the ecosystem, which may cause the trophic cascade effect (top-down or bottom-up) and potentially affect the food web and the sustainability of the fishery.

Supporting information

S1 Table. Complementary sampling information. Mean, minima, maxima size, number of samples (n) in each quarter/year by species/group considered off the Sirinhaém coast, north- eastern Brazil. For fish the size is related to standard length (cm); *for shrimps, carapace length (cm) and ** for mollusk, mantle length (cm).
(DOCX)

S2 Table. Additional diet data information considered to present study off the Sirinhaém coast, Northeastern Brazil. Location and year of data, total length range used and whether sea- sonal or ontogenic characteristics were considered (yes (y) or no (n)).
(DOCX)

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Chapter main findings and Thesis outlook

In this Chapter, using two complementary tools (SCA and SIA), we described the contribution of benthic sources and the importance of crustaceans, especially shrimps, in transporting energy from food web base to upper trophic levels providing support to coastal food-web in Sirinhaém (Figure 4). The presence of large mud banks in these coastal areas, which usually favors large occurrences of benthic invertebrates, such as worms and crustaceans, explains this huge importance. Given the absence of regulation for bottom trawling activities in the area, the cumulative effects of trawling on population parameters (e.g., size and food intake), potentially decreasing the abundance of benthic preys, may lead to changes in the trophic structure of the ecosystem, which may cause the trophic cascade effect (top-down or bottom-up) and potentially affect the food web and the sustainability of the fishery.

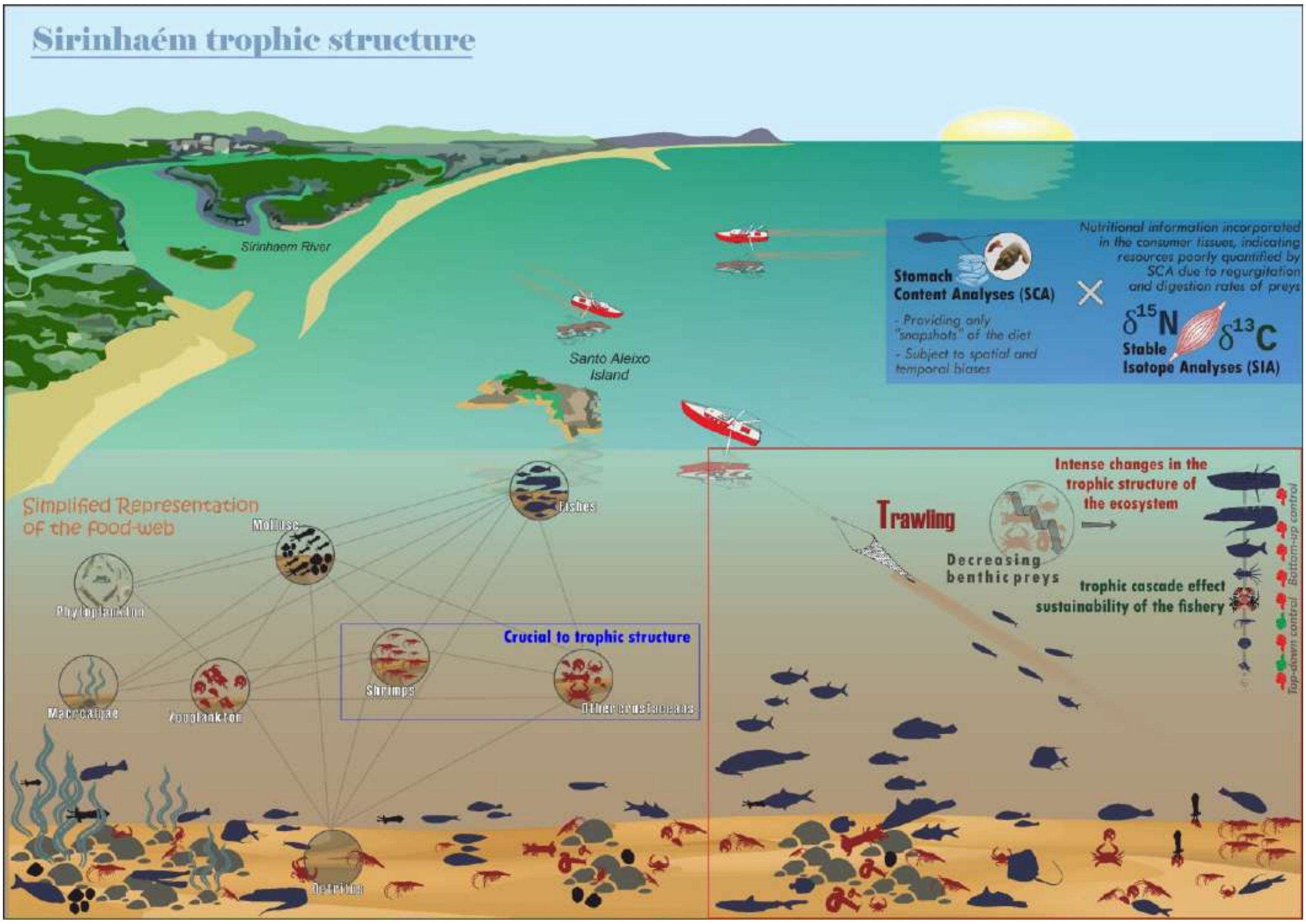
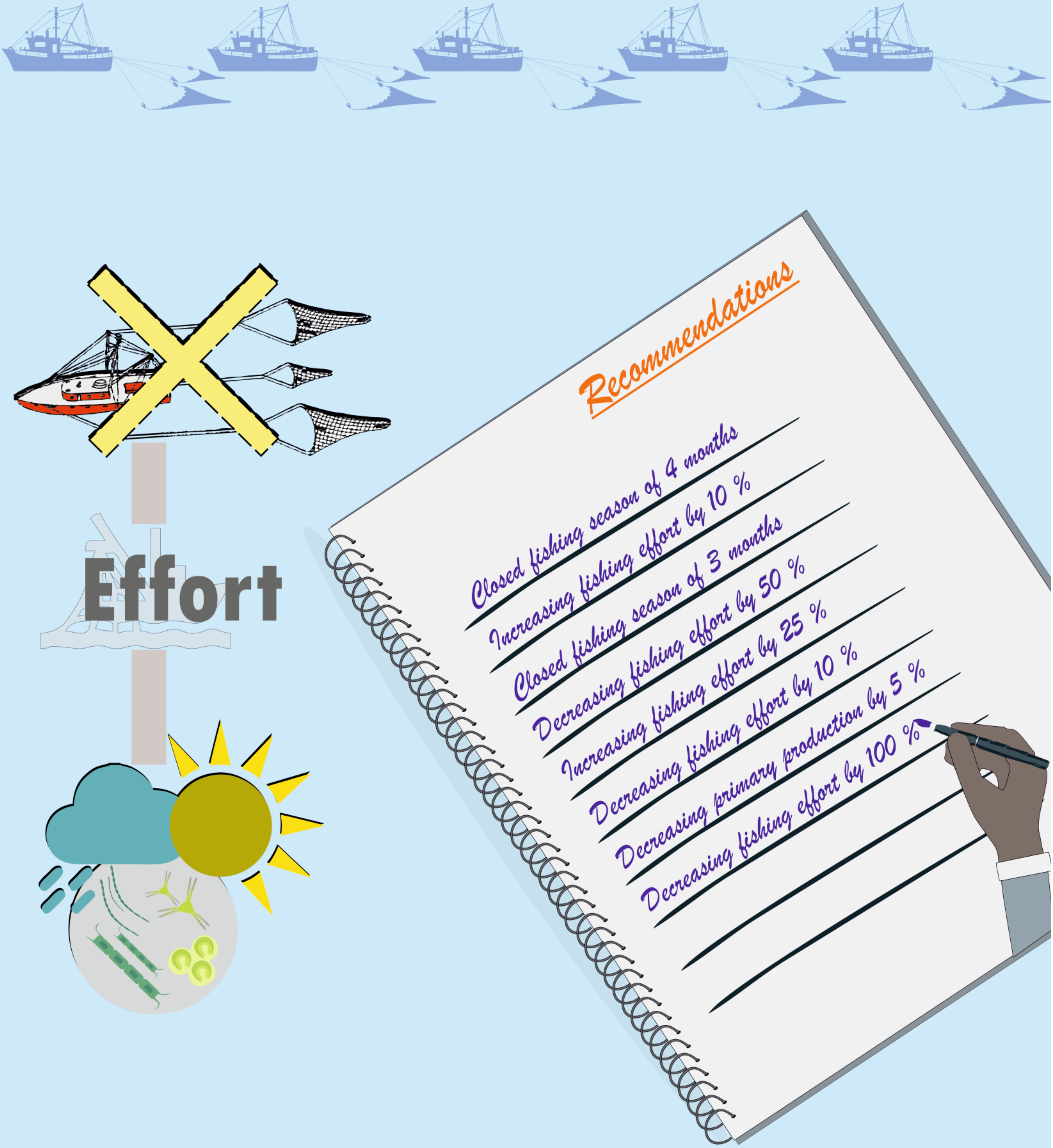


Figure 4. Diagram of the ecosystem structure in the Barra of Sirinhaém, Pernambuco, north-eastern Brazil.

For the Chapter 3, we used information obtained on both Chapters 1 and 2. The results found specifically in the Chapter 1, helped to define the main compartments and to summarize the available biological information of the target and non-target species, as well as to identify the decisive role of the environmental drivers in the biological and fishing seasonality of the species. These results, added to those of Chapter 2, contributed to the knowledge of trophic interactions and how the structure of the ecosystem could be affected by trawling. In another hand, the information from Chapter 1 was also useful as primary and secondary input data for development of the Ecopath and Ecosim (EwE) model presented in this Chapter (Figure 4), since the relationship between the stomach contents and stable isotope analysis was used as to validate the quality of the input diet data of this model. The previous Chapters provided an overview of the ecosystem by defining where, how and which species are caught by bottom trawling. However, they do not quantify the possible effects of this fishery and environmental factors at the individual or ecosystem level, considering the current framework of non-regulation of trawling, the importance of the climate season (e.g., rainfall, primary productivity) as a regulator of abundance patterns of target and non-target species, and the potential changes on trophic structure of the ecosystem in the region as highlighted in these previous Chapters.

In the next Chapter, we built an Ecopath model and applied a temporally dynamic model (Ecosim) to evaluate the potential isolated and combined effects of different fishing effort control policies and environmental changes on marine resources and ecosystem for our case study, novelty for tropical region.



Chapter 3

How the fishing effort control and environmental changes affect the sustainability of a tropical shrimp small scale fishery

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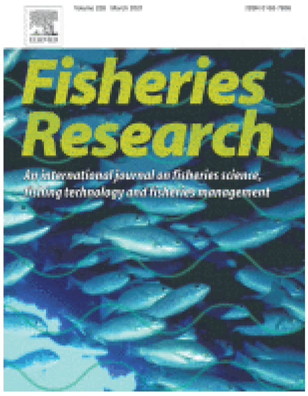

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How the fishing effort control and environmental changes affect the sustainability of a tropical shrimp small scale fishery

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ABSTRACT

Global shrimp catches are reported primarily in association with large industrial trawling, but they also occur through small-scale fishing, which plays a substantial role in traditional communities. We developed an Ecopath model in north-eastern Brazil, and applied a temporally dynamic model (Ecosim) to evaluate the potential effects of different fishing effort control policies and environmental changes on marine resources and ecosystem be- tween 2015 to 2030 with a case study for small-scale shrimp fishing, novelty for tropical region. These scenarios included different management options related to fishing controls (changing effort and closed season) and environmental changes (primary production changes). Our findings indicate that it is possible to maintain the same level of landings with a controlled reduction of bottom trawlers activities, for example, close to 10 %, without compromising the ecosystem structure. This scenario provided better results than 3–4 months of closing the fishing season, which led to significant losses in catches of high market-value target species (white shrimp, *Penaeus schmitti* and pink shrimp, *Penaeus subtilis*). However, intense negative effects on biomass, catch and biodiversity indicators were reported in scenarios with decreasing primary production, from 2 %, reinforcing the need to simulate and project the possible impacts caused by environmental change. However, the control of bottom trawling activity may help to reduce, even at low levels, the highly adverse effects due to primary production reduction. The impacts of climate change in a near future on organisms and ecosystems is an imminent reality, and therefore the search for measures for mitigating and even minimizing these impacts is crucial.

1. Introduction

Marine resources are one of the primary food sources in the world, contributing significantly to the food security and well-being of human society (Oyinlola et al., 2018); these resources are highly associated with environmental patterns or cycles and are frequently sensitive to anthropogenic pressures. Global climate change has modified local biodiversity in terms of the distribution, growth, fecundity, and recruitment of species, consequently affecting the catch amount and composition (Pörtner and Farrell, 2008; Roessig et al., 2004). Acceler- ated human population growth also implies an increase in the global food demand, which has consequently intensified the search for more effective methods of production, often unsustainable.

The reconstruction of global fishing trends (Cashion et al., 2018; Zeller et al., 2017), including Illegal, Unreported and Unregulated Fisheries (IUU) and discards, has revealed that purse seining and trawling fisheries are responsible for more than half of global catches. Despite having high levels of non-targeted catches, these fisheries may also have substantial adverse implications for marine habitats, particularly in the seabed structure and community biodiversity (Davies et al., 2018; Johnson et al., 2015; Ortega et al., 2018). The non-target catch (bycatch) may be divided into the part that is rejected at port or at sea, the one used for bait (industrial fisheries), or byproduct (commercially valuable species), as well as the amount consumed by the crew and local communities, primarily from small-scale fisheries (Davies et al., 2009; Gilman et al., 2014). Thus, the impact of fisheries on ecosystems appears to be counter-balanced by the beneficial role of the bycatch in the local community.

Global shrimp catches are reported primarily by large industrial trawlers, but some are also based on small-scale fishing, including non- motorized boats operating in estuaries and coastal waters, which play a major role in traditional communities (Gillett, 2008). Although their

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contribution to global discards are considered small (Zeller et al., 2017) mainly due to the remoteness of their landing sites and the decentralized nature of their activities, this sector provides an important source of income, employment and food to millions of people, making it one of the major economic activities in coastal communities around the world (Chollett et al., 2014). The lack of basic information (e.g., on species biology, catches, biomass, etc.) prevents researchers from evaluating the real impact of this activity on the ecosystem, posing a threat to its future sustainability (Andrew et al., 2007; Jeffers et al., 2019).

Frameworks and approaches have been developed to help the fishing impacts of multi-factor scenarios (Goti-Aralucea, 2019; Jones et al., 2018; Rezende et al., 2019; Rice, 2000), since human activities, marine organisms, and ecosystem changes interact and influence one another (Corrales et al., 2018). To address this challenge, a more comprehensive analysis and management of human activities and the environment is needed in accordance with an ecosystem-based management approach (Rosenberg and McLeod, 2005). In this context, strategies based on the principles of adaptive co-management and the Ecosystem Approach to Fisheries (EAF) (Guanais et al., 2015) have become very promising in recent years (Serafini et al., 2017). The EAF is an effective framework for ecosystem management that considers "the knowledge and un- certainties about biotic, abiotic, and human components of ecosystems and their interactions, applying an integrated approach to fisheries within ecologically meaningful boundaries" (Garcia et al., 2003).

Studies, methods or policies based on EAF are recommended to un- derstand and eventually mitigate the impacts of trawling. They have being applied to different countries (Jennings and Rice, 2011), fisheries (Gianelli et al., 2018), resources (Cuervo-Sánchez et al., 2018) and en- vironments (Rosa et al., 2014). The Code of Conduct for Responsible Fisheries (FAO, 1995) recommends that the entire catch, not only the targeted species, should be managed in an ecologically sustainable manner. To achieve this goal, the first step is to describe the fishing zones, target species, bycatch, and the factors that influence its varia- tion, and how they are related. This knowledge is essential for assessing the measures used for appropriate management (e.g., closed fishing seasons, Marine Spatial Planning (MSP) or Bycatch Reduction Devices (BRD)) (Bellido et al., 2011).

Among the tools considered within the EAF, the Ecopath with Ecosim (EwE) model (Christensen and Walters, 2004; Wolff et al., 2000) has been widely applied to characterize the trophic interactions and changes at the community level (Lira et al., 2018; Zhang et al., 2019) as well as to evaluate the effect of management policies on the environment and on ecosystem compensation (Halouani et al., 2016; Vasslides et al., 2017). In addition, the use of these approaches to forecast future cumulative impacts of human activities on aquatic food webs, such as fishing (Adebola and Mutsert, 2019; Piroddi et al., 2017) and stressors related to climate change (Bentley et al., 2019; Corrales et al., 2018; Serpetti et al., 2017), may be an interesting alternative to help manage ecosystems and their resources. However, particularly in countries with poorly managed fisheries (e.g., Brazil), studies are scarce.

In Brazil, shrimp are exploited by a multispecies fishery along the entire coastline and are caught primarily in shallow areas using motorized bottom trawl nets (Costa et al., 2007). Penaeidae species are the primary targets in Brazilian waters (Lopes, 2008). Shrimps of this family are captured by three fishery systems that differ in the size, technology and volume of the catch: the industrial, semi-industrial, and artisanal fleets (Dias-Neto, 2011). In the north-eastern region of Brazil, shrimp fishing is primarily performed by artisanal boats operating in shallow muddy coastal waters (Dias-Neto, 2011), involving more than 100,000 people and approximately 1700 motorized and 20,000 non-motorized boats (Santos, 2010), representing around 10 % of the total landed marine fishery resources in the country (IBAMA, 2008).

Despite their socio-economic importance, the effects of policy regu- lations and environmental variations in the Brazilian shrimp fishery have never been assessed with EAF models, specifically in terms of the EwE approach. Therefore, in this study, we developed an Ecopath with Ecosim (EwE) food web model approach to the Sirinhaém coast as a case study of north-eastern Brazil, in order to evaluate the potential isolated and combined effects of different scenarios related to closed seasons, fishing effort and environmental changes, simulated up to 2030. We expect that our results could provide straightforward responses to the decision makers, specifically those related to small scale bottom trawlers, with solutions that meet both fisheries and conservation objectives.

2. Methods

2.1. Study area

The Barra of Sirinhaém (BSIR), which is located on the southern coast of Pernambuco, in north-eastern Brazil (Fig. 1), is influenced pri- marily by the nutrient supply of the Sirinhaém river. The climate is tropical, with a rainy season that occurs between May and October. The rainfall ranges from 20 to 450 mm·month^{-1}, the mean water tempera- ture is 29 °C, and the pH and salinity range between 8.0 and 8.7 and 23 and 37, respectively (APAC, 2015; Mello, 2009). Fishing, the sugar cane industry and other farming industries are considered the primary pro- ductive activities in the region (CPRH, 2011). Fishing is performed near the coast (Manso et al., 2003) and the main fishing zones are inside or close to the Marine Protected Areas around Santo Aleixo Island (MPAs of Guadalupe and Costa dos Corais) (Fig. 1). The spatial extent of the model corresponds to the shrimp fishing areas in the BSIR with depths ranging from 4 to 20 m, covering a total area of 75 km^2.

2.2. Trawl fishery

Bottom trawling in the BSIR of north-eastern Brazil, the main fishery assessed in this study, has the largest and most productive motorized fishing fleet in Pernambuco, corresponding to 50 % of the shrimp pro- duction (Tischer and Santos, 2003), being an important source of in- come and food for the local population (Lira et al., 2010). This fishery is operated with fleet of twelve boats, from 1.5 to 3.0 miles off the coast, mainly between 10 and 20 m depth, with set duration of 4–8 hours and boat velocity varying between 2 and 4 knots. Boats often have 8 10 m of length, horizontal opening net of 6.1 m, mesh sizes of body and cod end of 30 mm and 25 mm, respectively. In Brazil, the regulations of this modality of fishery mostly involve a closed season (Dias-Neto, 2011; Santos, 2010) and fishermen and fisherwomen have the right to eco- nomic assistance during this time. However, despite its high relevance, Pernambuco is the only state in the region with no regulation. Shrimps of the Penaeidae family are the main targets: the pink shrimp (*Penaeus subtilis*), white shrimp (*Penaeus schmitti*), and seabob shrimp (*Xiphope- naeus kroyeri*) and the proportion of fish bycatch is 0.39 kg of fish captured for each 1 kg of shrimp (Silva Júnior et al., 2019). The fish bycatch is composed of 51 species, 38 genera and 17 families (Silva Júnior et al., 2019). The target shrimps and the most relevant non-target species were selected for model construction (Table S1).

2.3. Modelling approach

The Ecopath with Ecosim (EwE) version 6.6 (www.ecopath.org) approach has three primary modules: the mass-balance routine (Eco- path), the time dynamic routine (Ecosim) and the spatial-temporal dy- namic module in Ecospace. Initially, an Ecopath model was developed to quantify the trophic flows among compartments of the BSIR.

The Ecopath model simplifies the complexity of marine ecosystem dynamics through a mass balance approach on a system of linear equations that considers parameters such as the biomass, production and consumption of the species to describe the trophic flows between biological compartments, thus allowing the investigation of the possible responses of the ecosystem to anthropogenic impacts such as habitat degradation and/or fishing (Christensen and Pauly, 1992; Christensen

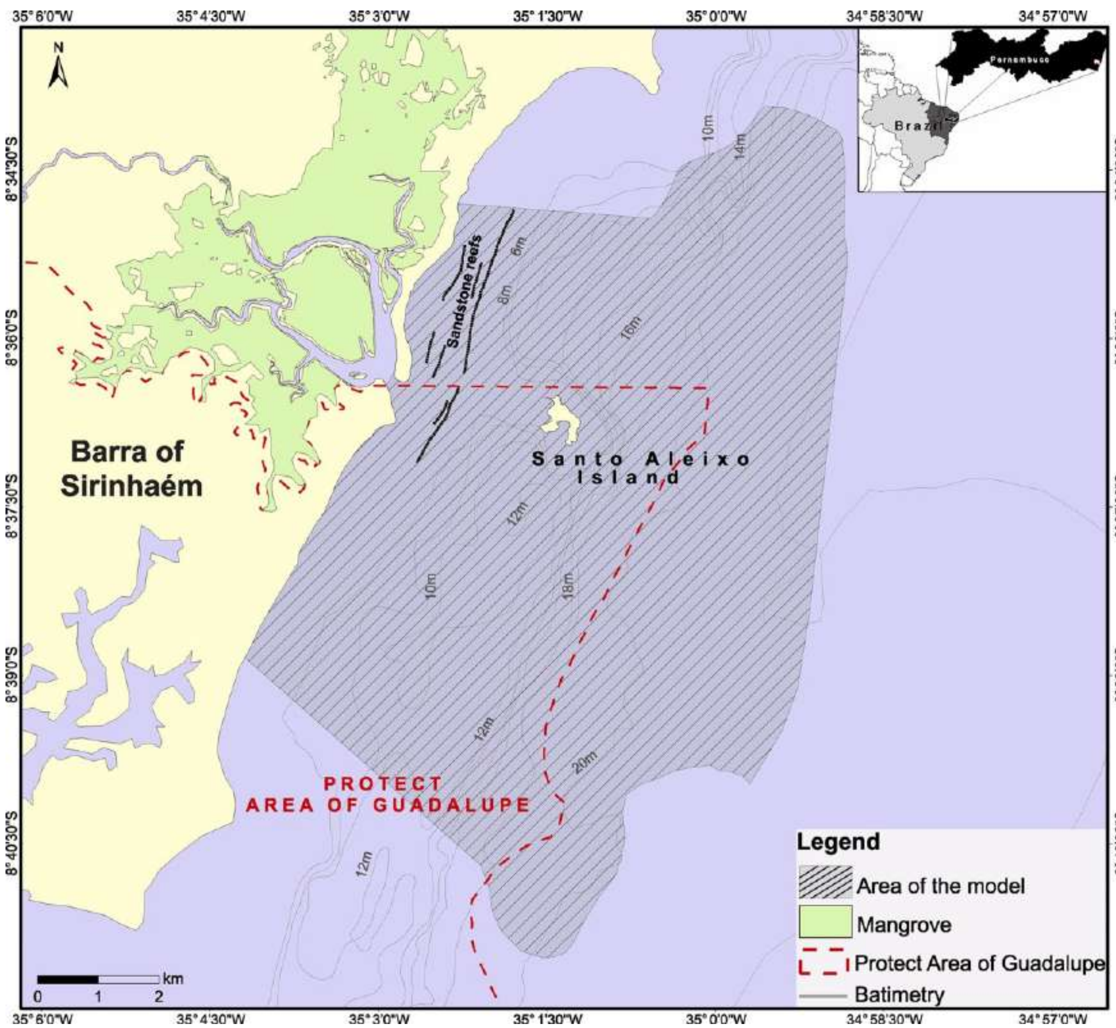

Fig. 1. Barra of Sirinhaém, Pernambuco, north-eastern Brazil, the area of the model (hachured area 75 km^2).

and Walters, 2004) (Appendix 1 for further details). The balanced Ecopath model (2011−2012) included 50 trophic groups with two pri- mary producer groups, one zooplankton compartment, twelve macro- benthos groups, 35 fish groups, and one group of birds, turtles and detritus (Fig. 2). The fish groups were selected given the importance of their biomass and landings, their position in the water column (pelagic, demersal, and benthic) and their trophic guilds (Elliott et al., 2007; Ferreira et al., 2019) (Table S1). This model accounted for the landings and bycatch of the primary fleets operating in the area, including bottom trawlers, gillnets and line. Following Heymans et al. (2016) and Link

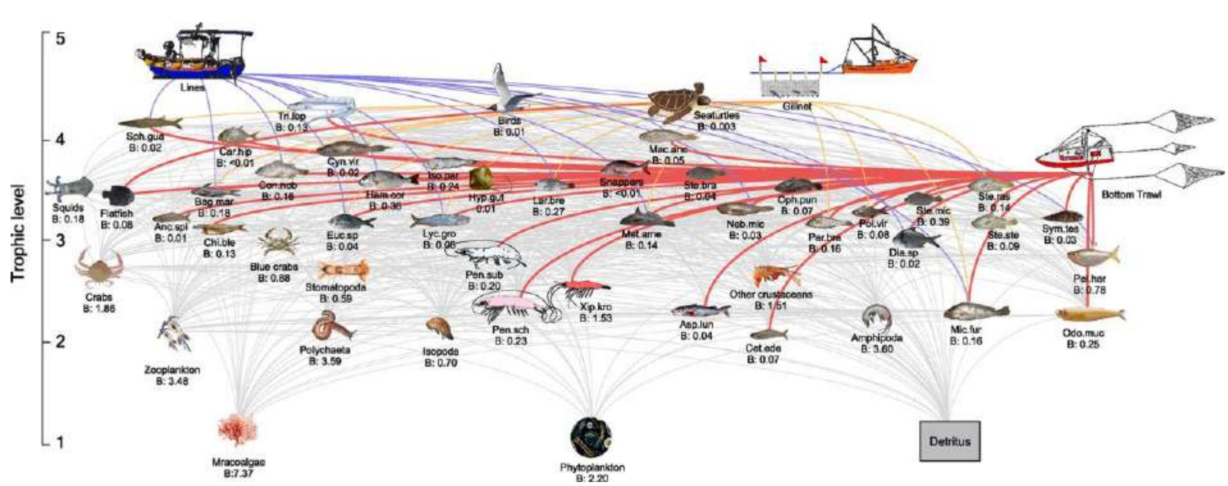

Fig. 2. Food web of the Barra of Sirinhaém Ecopath model (BSIR). The grey lines are the trophic paths and the orange, red and blue lines are the catches of the fleets of line, gillnet and bottom trawl, respectively. B is biomass in t km^{-2}. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

(2010), we analyzed the balance and confidence of our model by observing a set of criteria and assumptions using the pre-balanced (PREBAL) diagnostics routine (Link, 2010) (Table S4 and Fig. S2 for further details). A full description and the sources of information for the input and output parameters in the baseline Ecopath model are presented in Appendix 2 (Tables S1–S5 and Figs. S1–S5).

Based on the Ecopath model, the Ecosim time dynamic module was applied and fitted to a time series from 1988 to 2014. This model is a time-dynamic approach based on initial parameters from Ecopath that simulate changes in the estimates of biomass and catch rates over time, given the changes primarily exerted by fishing and the environment (Christensen and Walters, 2004; Walters et al., 1997). These estimates are performed by multiple coupled differential equations derived from the Ecopath equation.

$$\frac{dBi}{dt} = g_i \sum_{j=1}^{n} Q_{ji} - \sum_{j=1}^{n} Q_{ij} + I_i - (M_i + F_i + e_i)Bi \qquad (1)$$

where $\frac{dBi}{dt}$ is the growth rate in terms of biomass (B_i) over time for group i, g_i is the net growth efficiency (production/consumption ratio), I_i is the immigration rate, M_i is the natural mortality rate (unrelated to predation), F_i is the fishing mortality rate and e_i is the emigration rate (Christensen et al., 2008). Q_{ij} and Q_{ji} are the total consumption by group i and the predation by all predators on group i, respectively. The consumption rate calculations are based on the "foraging arena" theory (Ahrens et al., 2012; Walters et al., 1997) in which biomass B_i of prey is divided into two fraction: available prey (vulnerable) and unavailable prey (invulnerable fraction) which depend of the transfer rate (v_{ij}). The vulnerability parameter in Ecosim represent the degree to which an increase in predator biomass will cause in predation mortality for a given prey, determining the food web controls (top-down vs. bottom-up) (Christensen et al., 2008). Values close to 1 (low vulnerability) lead to bottom-up control, since the growth of the predator biomass will not cause a substantial increase in predation mortality on its prey. In the opposite, vulnerability values higher than 10 may lead to top-down control in the food web, and the positive variation in predator biomass causes significant impacts in the biomass of its prey due to predation mortality (Christensen et al., 2008).

2.4. Model fitting

The Ecosim model was fitted to the shrimp species trawl catch data based on the official fishery reports, which is the longest and more accurate time series available for the 1988–2014 period in the study area. The near-surface chlorophyll-*a* concentration was applied as a pri- mary production proxy from satellite image-processed data (Level-3) (source: https://oceancolor.gsfc.nasa.gov/) using an empirical relationship derived by *in situ* measurements and remote sensing (see Hu et al. (2012) for algorithm details). The mean chlorophyll-*a* data converted to t. km^{-2} was monthly obtained for October 1997 to December 2014 (*SEAWIFS* and *MODIS/AQUA* with resolutions of 9 km and 4 km, respectively) for the study area (8.56 °S/8.68 °S; 35.10 °W/34.95 °W) (see Fig. S6 for details). Therefore, the historical chlorophyll-*a* data was implemented as a forcing function of the primary production.

The vulnerabilities for each species/group that provided the best fit (measured by the weighted sum of squared deviations SS), was obtained, in three steps, using an iterative procedure of the "Fit to time Series" module of Ecosim. The first step determined the sensitivity of SS to vulnerabilities associated only with individual predator-prey in- teractions (Christensen et al., 2008). Secondly, anomalous patterns based on the time series values of relative primary productivity (forcing data, see above) were compiled. For the last step, both the vulnerability values and anomaly patterns were applied to reduce the SS. To assess the robustness of the fitted model, the landings estimates were compared using both the reported official and non-official catch statistics. The final vulnerability values used to provide the best fit are presented in Table S6.

2.5. Measuring the uncertainty

To assess the sensitivity of the Ecosim output, the Monte Carlo routine was applied (Heymans et al., 2016), assuming changes based on the pedigree indicator (Corrales et al., 2018; Serpetti et al., 2017) for each basic Ecopath input parameters (B, P/B, Q/B, and EE). We per- formed 1000 Monte-Carlo simulation trials for each species/group of the model in order to determine the confidence intervals (CI: 5 and 95 %) for the Ecosim outputs (fitted results and ecological indicators).

2.6. Scenario simulation

We proposed a simulation and evaluation of the fishing management scenarios (FMS) and the responses of the target species (shrimps), bycatch and whole ecosystem using the Ecosim temporal dynamic module from the BSIR base model (2011–2012). Seventeen scenarios were simulated. These scenarios were related to closed period of the trawling fishery based on the number of months of maximum reproduction/recruitment activity of shrimp species and bycatch and on the current shrimp regulation in Brazil (Normative N°14 MMA/2004); in- crease and decrease of trawl fishing effort; and environmental drivers using primary production changes as proxy (Table 1). Thus, we evalu- ated scenarios with 4 (clos1s) and 3 months (clos2s) of closed fishing periods; scenarios (scenarios "inc" and "dec") with increase (inc) and decrease (dec) in fishing effort by 10, 25, 50 and 100 %; and scenarios with a decrease in the primary production from 0.5–10 % (scenarios env1-env3), considering the expected variation, in our region, of the primary productivity given the predictable decreasing trend in the rainfall caused by climate change (Blanchard et al., 2012; Krumhardt et al., 2017; Lotze et al., 2019; Reay et al., 2007) (Table 1).

We considered a two-tiered approach, first looking at individual strategies (fishing and environmental drivers as reported above) then by the combination of these factors (fishing environmental drivers). For this, the combined scenarios involving closed seasons and effort control that supplied the best results considering the balance between increasing the catch and maintaining conservation indicators (e.g., biomass) were incorporated into the scenarios concerning the primary productivity to evaluate the cumulative effects of the three factors, into management measures. From the original configuration of the fitted model, here considered as the baseline simulation (Stand), the 17 scenarios were performed to assess the responses of the marine resources and ecosystem conditions to fifteen years, between 2015 to 2030 (Table 1).

2.7. Indicator analysis

The absolute values of the biomasses and catches for each trophic group in each simulated scenario from 2015 to 2030 were compared to the baseline model of constant effort (scenario-stand). The average ratio values (e.g., final biomass/initial biomass) for each scenario are represented by colour heatmaps indicating the increases or decreases in the biomass and catches from 2015 to 2030. Additionally, several indicators associated with the biomass, catch, size and trophic level were assessed to evaluate the response of the ecosystem to the different simulations over time (Table 2) (Coll and Steenbeek, 2017). These indicators were then correlated over the period from 2015 to 2030 by the Spearman's rank correlation (see Corrales et al. (2018); Piroddi et al. (2017)).

3. Results

3.1. Ecopath model

A balanced Ecopath model was developed to represent the ecosystem function and to characterize the food web structure in the BSIR from

Table 1
Fishing management scenarios simulated to Barra of Sirinhaém Ecosim model between 2015 to 2030.

	Scenarios	Description	Axis	Justification	Source
1	Stand	Baseline model without change of fishing months		-	-
2	clo1s	Included closed fishing season based on the peak of reproduction and recruitment of the shrimp species (4 months)	Temporal	Shrimp and *bycatch* species present specific breeding and recruitment seasons from December to July	1, 2, 3
3	clo2s	Included closed fishing season based on the peak of reproduction and recruitment of the shrimp and *bycatch* species (3 months)			
4	inc(+10%)	Increasing fishing effort by 10, 25, 50 and 100%	Effort	Stock status based in traditional approaches indicates that the fleet exploits shrimp species close or at maximum exploitation rates	1
5	inc(+25%)				
6	inc(+50%)				
7	inc(+100%)				
8	dec(-10%)	Decreasing fishing effort by 10, 25, 50 and 100% (no fishing)			
9	dec(-25%)				
10	dec(-50%)				
11	no_fishing dec(-100%)				
12	env1	Decreasing primary production (PP) by 2, 5 and 10% respectively	Environnemental	Biomass and catch patterns of shrimp and *bycatch* species are associated to environmental drivers (e.g., chlorophyll-a and rainfall).	4
13	env2				
14	env3				
15	clos + env	Scenarios of best balancing conditions, in terms of catch and conservation indicators, combined with reducing primary productivity.	Minimise or maximize the impacts obtained by environmental change		-
16	inc + env				
17	dec + env				

1-Lopes et al. (2017); Peixoto et al. (2018) and Silva et al. (2016); 2- Normative N°14 MMA/2004; 3-Silva Júnior et al. (2015) and Eduardo et al. (2018); 4-Blanchard et al. (2012); Krumhardt et al. (2017); Lotze et al. (2019); Reay et al. (2007).

2011 to 2012. A full description and sources of information of the input and main output parameters for the fifty trophic groups (Fig. 2) of the baseline Ecopath model are presented in Appendix 2.

The values of the B, P/B, Q/B, EE and landings for all groups and fleets (Table 3) revealed that the invertebrates represented more than

Table 2
Ecological indicators considered to evaluate the changes on the ecosystem over time.

Code	Ecosystems Attributes	Description	Goal	Units	Reference
Total B	Total biomass	Sum of the biomass of all groups in the ecosystem (excluding detritus)	Quantify general changes at the ecosystem level	$t \cdot km^{-2}$	1
Fish B	Biomass (B) of fish	Sum of the biomass of fish species	Evaluate the dynamics of fish group	$t \cdot km^{-2}$	1
Inver.B	Biomass (B) of invertebrate	Sum of the biomass of invertebrate species	Evaluate the dynamics of invertebrates in response to fishing and predation	$t \cdot km^{-2}$	1
Kemp.Q	Kempton's biodiversity index (Q)	Represents the slope of the cumulative species abundance curve	Measure the effects of mortality on species diversity	-	2
Total C	Total Catch (C)	Sum of the catch of all species in the ecosystem	Represent the dynamics of fisheries	$t \cdot km^{-2} y^{-1}$	1
Fish C	Catch (C) of all fish	Sum of the catch of all fish species	Represent the dynamics of fish fisheries	$t \cdot km^{-2} y^{-1}$	1
Inver.C	Catch (C) of all invertebrate	Sum of the catch of all invertebrate species	Represent the dynamics of invertebrate fisheries	$t \cdot km^{-2} y^{-1}$	1
Disc	Total discarded catch	Sum of the catch of all species that are discarded	Assess the impact of fisheries with discards	$t \cdot km^{-2} y^{-1}$	3
mTLc	Tropic level (TL) of the catch	Represents the mean trophic level only of species catch	Evaluate the fishing fleet strategy	-	4
mTLco	Trophic level (TL) of the community (including all organisms)	Represents the mean trophic level weighted by biomasses of all species in the ecosystem	Evaluate the fishing fleet strategy	-	5
MTI	Marine trophic index (including organisms with TL ≥ 3.25)	Represents the mean trophic level only of species catch with a trophic level ≥ 3.25	Evaluate the fishing effect in top food-web	-	6
MLFco	Mean length (ML) of fish community	Represents the mean lenght weighted by biomasses only of fish species	Observe the trends or change of fish size in the ecosystem	cm	7
MLFc	Mean length (ML) of fish catch	Represents the mean length only of fish species	Represent the size dynamics catch species in the ecosystem	cm	7

1: Hilborn and Walters, 1992; 2: Ainsworth and Pitcher (2006); 3: Zeller et al. (2017); 4: Gascuel et al., 2011; 5: Shannon et al., 2014; 6: Pauly and Watson, 2005 7: Ravard et al., 2014 and Rochet and Trenkel, 2003.

half of the total biomass, being 11 % shrimps, while the biomass of the fish represented 14 % of the total biomass. Among the fleets evaluated, gillnet and line represented 35 % of the total landings, while the trawling corresponded to 75 % in BSIR, with the shrimp species totalizing approximately 84 % of the total catch.

Birds (TL = 4.26), Seaturtles (TL = 4.20) and piscivore fish such as *Trichiurus lepturus* - Tri.lep (TL = 4.19), *S. guachancho* - Sph.gua (TL 4.06), *M. ancylodon* - Mac.anc (TL = 3.20) had the highest estimated trophic levels of the food web (Fig. 2) and the larger number of trophic pathways. Compared with the trawling fleet, the target of line and gillnet fleets was mostly the species with higher TL.

The herbivore/detritivore rate (H/D) was 2.21, indicating that the energy flowed in larger proportion mainly from the primary producers to the second trophic level in the BSIR food web (Table 4). The Total System Throughput (TST) was 4060 t km^{-2}·y^{-1}, with 25 % due to consumption and 35 % due to flows into detritus. The mean trophic level of the catch (TLc) was 2.89, and the rates of the TPP/TR and TPP/TB were 3.84 and 49.36 respectively, while the Finn's Cycling Index (FCI) was low (3.76), and the system overhead was 69 %.

3.2. Historical ecosystem state

The catches predicted from the Ecosim baseline model (Stand) were compared to the catch time series for the target shrimp species (*X. kroyeri, P. subtilis* and *P. schmitti*) (Fig. 3). The model was able to recreate the official values and trends in catches for these species (Fig. 3), reproducing the increased catches between 1994 and 1997 and between 2004 and 2007.

Except for the Kempton's biodiversity, which decreased from 1988 to 2014, the ecosystem indicators displayed similar trends over time in the

Table 3

Basic inputs and estimated outputs (in bold) of the groups of the Barra of Sirinhaém Ecopath model (BSIR), Pernambuco, Northeast of Brazil. TL: trophic level; B: biomass; P/B: production–biomass ratio; Q/B: consumption–biomass ratio; EE: ecotrophic efficiency and Landings (t. km^{-2}). See Table S1 to group name details.

Group name (t km^{-2})		TL	B	P/B (year^{-1})	Q/B (year^{-1})	EE	Landings (t km^{-2}) Trawling	Gillnet	Line
1	Macroalgae	**1.00**	7.370	13.25	–	**0.75**	–	–	–
2	Phytoplankton	**1.00**	2.200	682.00	–	**0.32**	–	–	–
3	Zooplankton	**2.05**	3.480	50.21	150.65	**0.69**	–	–	–
4	Polychaeta	**2.13**	**3.596**	3.60	25.52	0.95	–	–	–
5	Amphipoda	**2.23**	**3.607**	6.64	34.51	0.95	–	–	–
6	Blue crabs	**2.92**	**0.880**	2.00	8.00	0.90	–	–	–
7	Crabs	**2.70**	**1.860**	5.23	10.82	0.95	–	–	–
8	Isopoda	**2.05**	**0.706**	13.75	34.51	0.95	–	–	–
9	Pen.sub	**2.79**	0.208	5.25	13.45	**0.94**	0.1075	–	–
10	Pen.sch	**2.30**	0.230	3.75	13.45	**0.88**	0.1770	–	–
11	Stomatopoda	**2.69**	**0.597**	23.68	85.27	0.95	–	–	–
12	Xip.kro	**2.52**	1.533	10.40	26.00	**0.99**	0.5013	–	–
13	Other crustaceans	**2.61**	**1.512**	5.80	19.20	0.95	–	–	–
14	Squids	**3.44**	0.18	6.40	36.50	**0.86**	–	–	–
15	Flatfish	**3.37**	0.087	3.07	11.26	**0.41**	0.0018	<0.0001	–
16	Anc.spi	**3.15**	0.012	2.68	13.30	**0.92**	0.0003	–	–
17	Asp.lun	**2.23**	0.042	2.27	12.50	**0.65**	0.0012	–	–
18	Bag.mar	**3.43**	0.183	2.30	8.49	**0.54**	0.0059	0.0067	0.0554
19	Car.hip	**3.96**	0.0001	0.46	6.66	**0.61**	<0.0001	–	–
20	Cet.ede	**2.00**	0.072	2.29	53.42	**0.63**	0.0022	–	–
21	Chi.ble	**3.06**	0.135	3.05	20.19	**0.99**	0.0045	–	–
22	Con.nob	**3.59**	0.164	3.22	8.78	**0.04**	0.0059	0.0031	0.0009
23	Cyn.vir	**3.82**	0.027	2.53	5.00	**0.86**	0.0010	0.0005	0.0020
24	Dia.sp	**2.91**	0.027	2.90	10.61	**0.47**	0.0005	–	0.0001
25	Euc.sp	**3.11**	0.042	1.33	12.84	**0.36**	0.0008	0.0004	0.0001
26	Ham.cor	**3.54**	0.366	2.48	11.19	**0.11**	0.0140	–	0.0017
27	Hyp.gut	**3.51**	0.015	0.35	2.68	**0.17**	0.0004	–	–
28	Iso.par	**3.72**	0.246	1.93	8.13	**0.35**	0.0082	–	–
29	Lar.bre	**3.50**	0.275	2.49	8.48	**0.47**	0.0100	0.0165	0.0006
30	Snappers	**3.61**	0.006	0.27	6.47	**0.57**	0.0001	–	–
31	Lyc.gro	**3.11**	0.068	3.03	20.69	**0.76**	0.0025	0.0004	0.0006
32	Mac.anc	**3.91**	0.051	1.75	8.20	**0.97**	0.0020	0.0018	0.0786
33	Met.ame	**3.15**	0.140	2.15	7.19	**0.56**	0.0039	0.0002	0.0323
34	Mic.fur	**2.25**	0.162	2.69	6.90	**0.29**	0.0033	0.0051	0.0207
35	Neb.mic	**3.26**	0.037	1.44	8.50	**0.76**	0.0011	–	0.0017
36	Odo.muc	**2.21**	0.257	4.58	17.70	**0.82**	0.0087	–	–
37	Oph.pun	**3.42**	0.077	1.93	10.88	**0.44**	0.0021	–	–
38	Par.bra	**3.12**	0.162	3.89	8.70	**0.87**	0.0060	0.0018	–
39	Pel.har	**2.81**	0.783	2.90	81.00	**0.72**	0.0268	–	0.0004
40	Pol.vir	**3.21**	0.083	3.83	12.05	**0.21**	0.0031	0.0004	
41	Sph.gua	**4.07**	0.028	0.49	4.65	**0.99**	0.0009	0.0001	0.0093
42	Ste.bra	**3.61**	0.047	2.19	12.90	**0.89**	0.0016	–	–
43	Ste.mic	**3.36**	0.396	5.47	11.07	**0.35**	0.0148	–	–
44	Ste.ras	**3.47**	0.148	3.56	8.09	**0.83**	0.0062	<0.0001	0.0002
45	Ste.ste	**3.20**	0.094	2.11	11.60	**0.46**	0.0031	–	–
46	Sym.tes	**3.17**	0.031	1.27	10.51	**0.83**	0.0012	–	–
47	Tri.lep	**4.20**	0.139	1.68	3.62	**0.51**	0.0023	0.0001	0.0687
48	Birds	**4.26**	0.015	5.40	80.00	**0.00**	–	–	–
49	Seaturtles	**4.20**	0.003	0.15	22.00	**0.00**	–	–	–
50	Detritus	**1.00**	–	–	–	**0.17**	–	–	–

Table 4

Ecosystem attributes, ecological and flow indicators of the Barra of Sirinhaém Ecopath model, Pernambuco, Northeast of Brazil.

Parameters	Value	Units
Ecosystem properties		
Sum of all consumption (TC)	1029.88	t $km^{-2} y^{-1}$
Sum of all exports (TE)	1182.09	t $km^{-2} y^{-1}$
Sum of all respiratory flows (TR)	416.14	t $km^{-2} y^{-1}$
Sum of all flows into detritus (TD)	1432.14	t $km^{-2} y^{-1}$
Total system throughput (TST)	4060.26	t $km^{-2} y^{-1}$
Sum of all production (TP)	1886.05	t $km^{-2} y^{-1}$
Mean trophic level of the catch (TLc)	2.89	–
Gross efficiency (catch/net p.p.)	0.00085	–
Calculated total net primary production (TNPP)	1598.09	t $km^{-2} y^{-1}$
Net system production (NSP)	1181.95	t $km^{-2} y^{-1}$
Total biomass (excluding detritus) (TB)	32.38	t km^{-2}
Total catch (Tc)	1.37	t $km^{-2} y^{-1}$
Ecosystem maturity		
Total primary production/total respiration (TPP/TR)	3.84	–
Total primary production/total biomass (TPP/TB)	49.36	–
Total biomass/total throughput (TB/TST)	0.008	y^{-1}
Food web structure		
Connectance Index (CI)	0.26	–
System Omnivory Index (SOI)	0.27	–
Finn's Cycling Index (FCI)	3.76	% TST
Finn's mean path length (FML)	2.54	–
Ascendancy (AS)	30.05	%
System Overhead (SO)	69.95	%
Herbivore/Detritivore rate (H/D)	2.21	–
Model reability		
Ecopath pedigree index	0.65	–
Transfer efficiency total	18.14	%

structure of the BSIR (Fig. S7). The increases were related to different indexes (e.g., Fish B, Total C, MTI, mTLc, and TL catch) from 1994 to 1997 and 2004 to 2007 (Fig. S7).

3.3. Back to the future

After closing the fishing period to the trawling fleet for 4 and 3 months (clo1s and clo2s), the model predicted a similar pattern of biomass and catches. In these scenarios, the bycatch fish, shrimp, birds and turtles increased in biomass compared to the baseline, while the biomass of the lower TL compartments (phytoplankton, zooplankton and other invertebrates) increased for clo1s and decreased for clo2s over time in the 2015–2030 projection (Fig. 4). Simulations of increased or decreased trawling efforts (e.g., inc(+50 %), inc(+100 %), dec(-25 %) and dec(-50 %)) indicated divergent effects, with differences being more evident in scenarios with effort changes above 25 %. By reducing the effort, the biomass of the target species increased, as did the bycatch fish, birds and turtles, but to a lesser extent (Fig. 4). Scenarios with increased trawling effort projected a negative impact on biomass for the target species *P. schmitti* and *P. subtilis* and for the bycatch fish (e.g., *Hypanus guttata, Paralonchurus brasiliensis* and *Trichiurus lepturus*) (Figs. 4 and 5). Similar trends were noted during primary production (PP) scenarios (env1, env2 and env3).

Specifically, for the target species (*P. subtilis* and *P. schmitti*), with the reduction in fishing effort and in considering the closed season to trawling, the simulations projected progressive recoveries, almost doubling the initial biomass over time (Figs. 4 and 5). However, the increased trawling effort and primary production scenarios negatively impacted the biomass of these two shrimp species in comparison to the baseline scenario, with a reduction of 68 % for *P. subtilis* and 86 % for *P. schmitti* in the inc(+100 %) scenario (Figs. 4 and 5). For *X. kroyeri*, there was a slightly positive variation in the biomass, from 0.06 % to 0.28 % when reducing the effort, while in the PP scenario (e.g., env3), the shrimp biomass declined from approximately 12 % (Figs. 4 and 5).

In general, scenarios involving closed fishing periods, decreased trawling efforts and PP reduction led to few changes (e.g., dec(-10 %)) and, in some cases reduced catches (e.g., clo1s, dec(-50 %) and env2) of the shrimp and bycatch species (Fig. 6). Although in general, the increased effort projected an average increase capture of the shrimp species (Fig. 6) (*P. subtilis* for example), only in the short term (2015–2020), these scenarios involving increased effort (e.g., 10–50%)

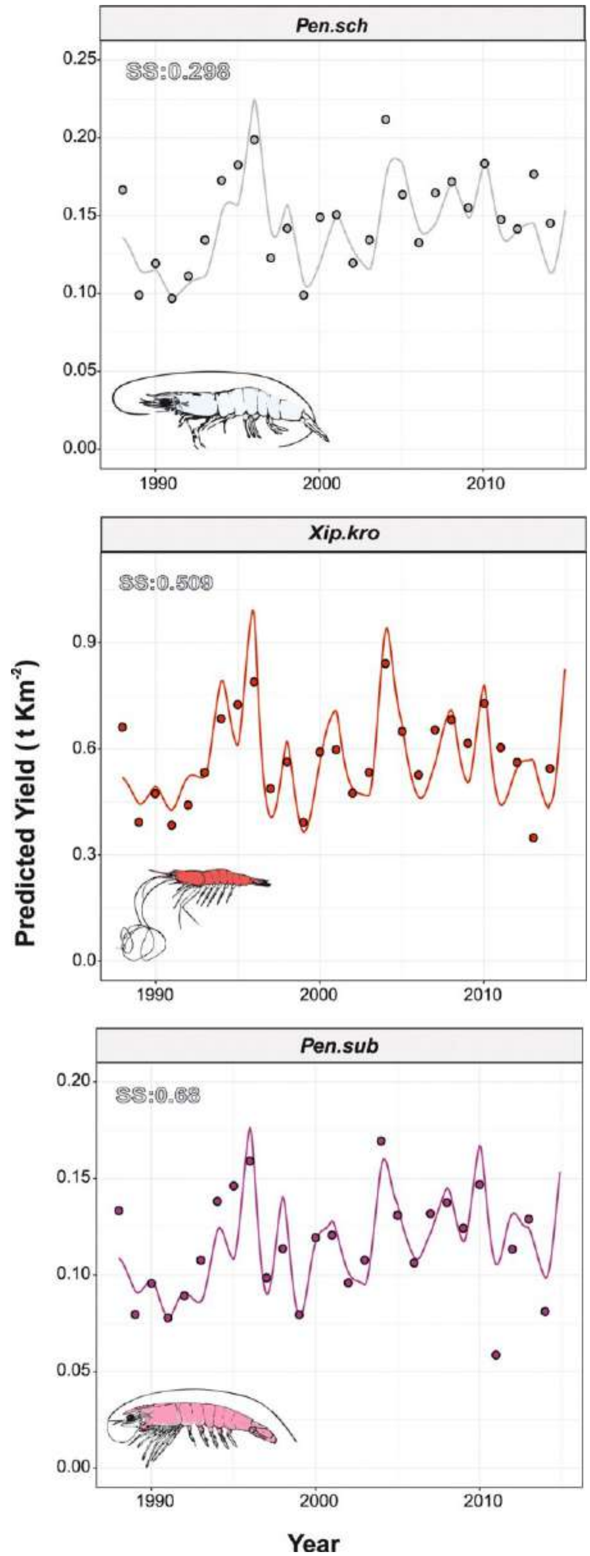

Fig. 3. Comparison between the estimated landing time series from the Ecosim model (lines) and official logbooks of landings (1988-2014) in the Barra of Sirinhaém Ecopath model, Pernambuco, north-eastern Brazil.

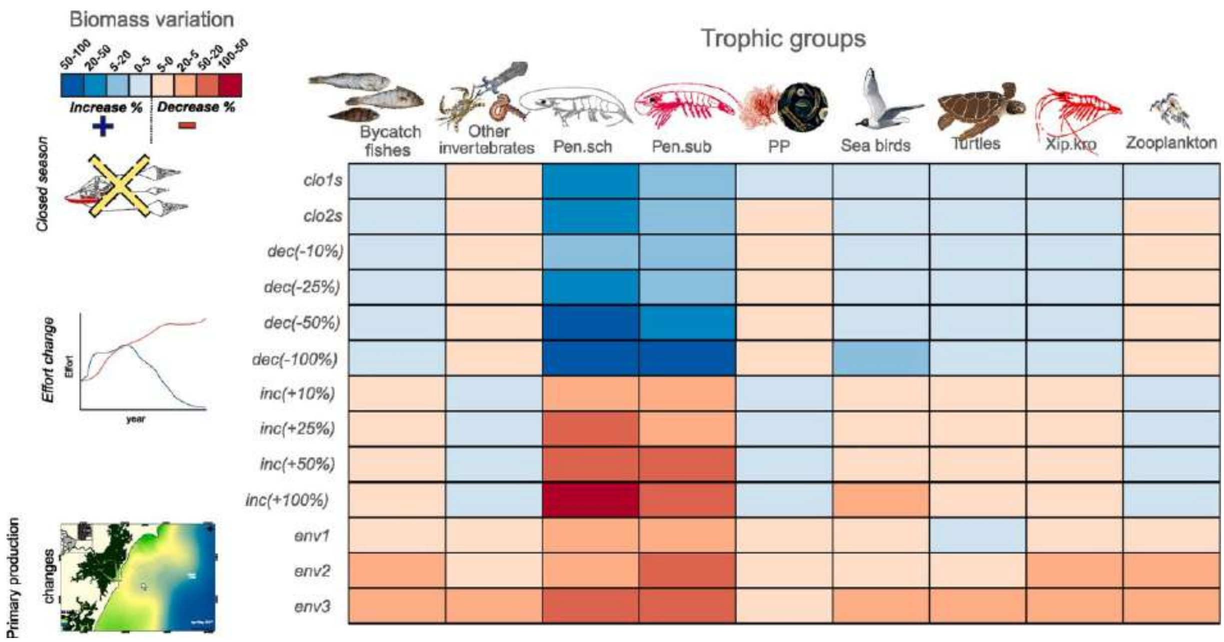

Fig. 4. Average biomass variations for each trophic group obtained by Fishing Management Scenario simulation from 2015–2030 compared to the baseline model (constant effort). Blue and red-coloured gradients indicate increased and decreased biomass, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

has shown a gain of 4–16 % in the catch, being gradually reduced until 2030 (see Table S7). However, for *P. schmitti*, the trend projected a reduction of approximately 27–70% (e.g., inc(100 %)) in catches between 2020 and 2030 (see Table S7). All the biomass and catch ratios for the shrimp species and FMS compared to the baseline scenario are available in the Table S7 and Fig. S8.

The ecosystem indicators calculated from the Ecosim outputs showed similar patterns in the scenarios temporarily closed to trawling. A significant increasing trend (t-test; $p < 0.05$) in biomass-based indicators (Total B, Fish B and Inver B), such as trophic (mTLc and MTI) and size-based (MLFc) indexes (Fig. 7), was projected. In addition, those indicators increased over time with the effort reduction, except for the total and invertebrate catches for dec(-25 %) to dec(-100 %) scenarios (Fig. 7). Under the 10 % increased fishing effort scenarios (inc(+10%)), several indicators associated with the biomass, catch and size, primarily Fish B, Inver C and mTLco, presented a significant increasing pattern (Fig. 7) (t-test; $p < 0.05$), although an increased effort of >50 % (e.g., inc+50 %) and inc(+100 %)) showed negative impacts on the Kempton's biodiversity (Kemp Q) and Inver B (t-test; $p < 0.05$). Strong negative effects (t-test; $p < 0.05$) in all PP reduction scenarios, primarily for those with changes above 2 % (env2 and env3), were reported (Fig. 7). The indicators predicted in the model, with confidence intervals assessed by Monte Carlo routine for each FMS, are presented in Fig. S9.

3.4. Cumulative effects of the PP anomaly and FMS

Among the individually evaluated FMS, the closed fishing periods (clo1s – 4 months) and the scenarios with little changes in effort (increase – 10 % and decrease - 10 %) showed the best balancing conditions, with minimal reduction to even improvement of catches (e.g., invertebrate capture) and conservation indicators (Fig. 8). These scenarios (clo1s; inc(+10 %); dec(-10 %)) were combined to drive environmental changes, in terms of reducing the PP to assess the cumulative effects of the impacts obtained from the PP change and FMS until 2030. Thus, among the climate change scenarios (Blanchard et al., 2012; Krumhardt et al., 2017; Lotze et al., 2019; Reay et al., 2007) and the time of our model (until 2030), the 2 % is the lowest PP reduction rate, hence we have chosen as the most feasible PP scenario. The model projected a reduction of the impact on the biomass caused by the PP decrease with bottom trawl reduction control in 10 % (dec(-10 %)). However, the increased effort scenarios intensified the biomass decrease for shrimp and high TL species, which were already reduced by the decreasing PP (Fig. 8 and Fig. S8).

P. subtilis and *P. schmitti* showed the largest cumulative recovery in terms of biomass for the 4-month closed fishing period (clo1s+env1), followed by 10 % effort reduction dec(-10 %) env1 (Fig. 8). The management measures related to effort control (clo1s, dec(-10 %), inc(+10 %)) led to few changes in the *X. kroyeri* biomass with PP reduction (Fig. 8). In terms of catch, the FMS over time barely changed the trends observed with the reduced PP for shrimp species, except for *X. kroyeri* (Fig. 8). All the biomass trends for each species, including bycatch and FMS compared to the env1 scenario, may be observed in Fig. S8.

3.5. Scenarios as decision support tools

In general, the target and some non-target species biomasses benefit from decreased fishing pressure, but the catches are reduced. However, a controlled increase in trawling up to 10 % led to promising results in terms of catches and biomass level maintenance. Our findings indicated that the effort-reduction conservation measures evaluated here (e.g., clo2s and dec(-50 %)) have positive impacts on ecosystem health indicators (e.g., high TL biomasses and shrimp, mean trophic level of the ecosystem); however, they have a negative effect on catches at different trophic levels (Fig. 9). The opposite trend was noted with increased bottom trawling activity (Fig. 9). Adverse effects on all aspects of conservation and exploitation were reported with the environmental simulations (PP decrease on 2 %) of the near future. These negative conditions resulting from PP were minimized with the implementation of management measures, especially with a 10 % trawling reduction (Fig. 9).

4. Discussion

Although their contribution to global discards are considered small (Zeller et al., 2017), small-scale fisheries, primarily those operating in

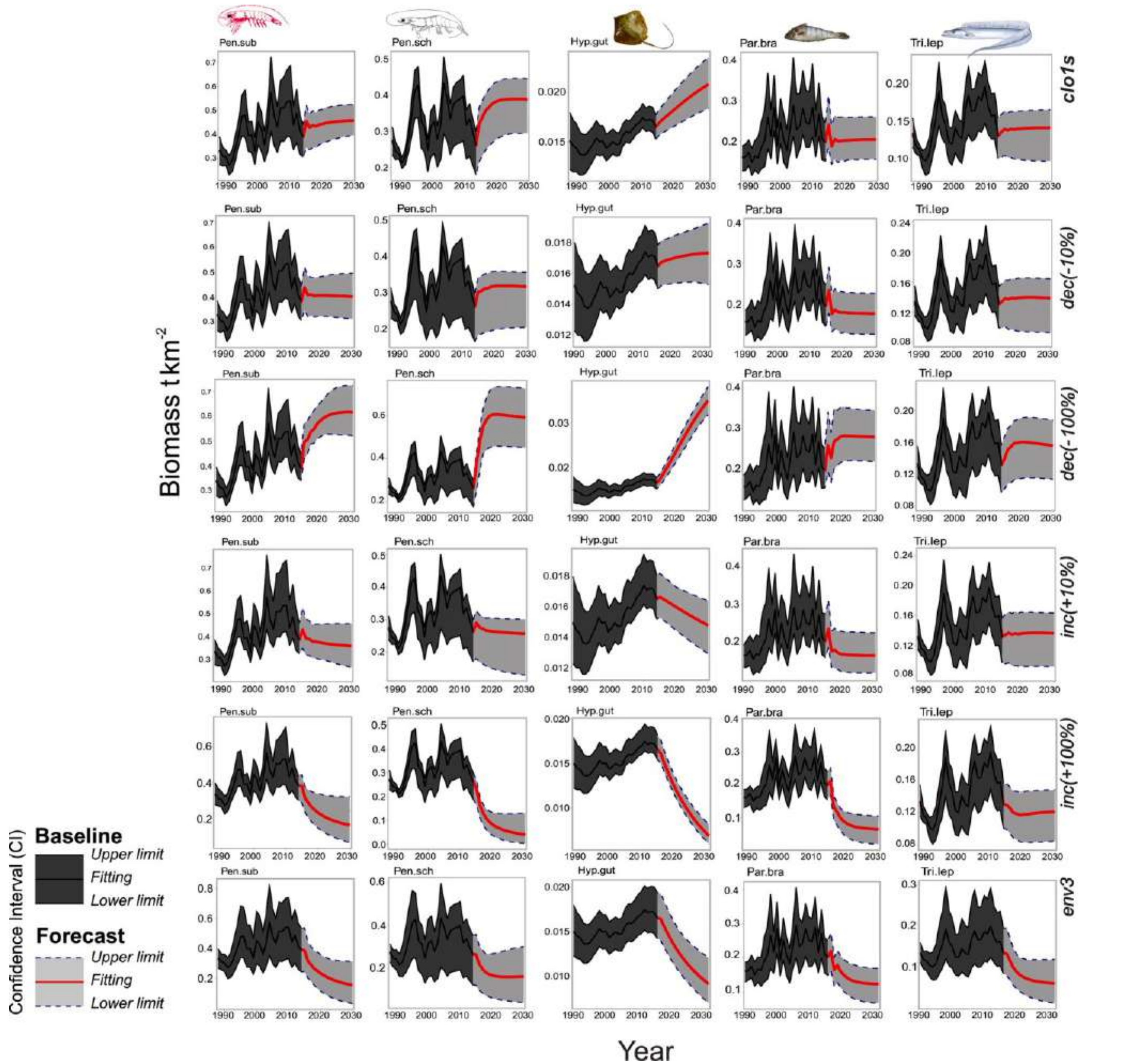

Fig. 5. Biomass predicted (t km^{-2}) in the model with a confidence interval of 95 % by Monte Carlo routine (1000 runs) for some groups in the scenarios clo1s, dec(-10 %), dec(-100 %), inc(+ 10 %), inc(+ 100 %) and env3. Pen.sub: *Penaeus subtilis*; Pen.sch: *Penaeus schmitti*; Xip.kro: *Xiphopenaeus kroyeri*.; Hyp.gut: *Hypanus guttata*; Par.bra: *Paralonchurus brasiliensis* and Tri.lep: *Trichiurus lepturus*.

estuaries and coastal waters, play an important role in traditional communities (Gillett, 2008). On the Brazilian coast, limiting fishing efforts, closed fishing periods, and mesh size regulations (Dias-Neto, 2011; Gillett, 2008; Santos, 2010) are the currently applied management recommendations used to regulate the shrimp fisheries in this country. However, this is not the case for Barra of Sirinhaém (BSIR) in Pernambuco (Northeast Brazil), which is currently unregulated. Although they are applied in most parts of the country, these management strategies may be ineffective primarily due to weak fishery policy associated with limited fisher knowledge about formal norms and also given their traditional approaches to focusing on single species, without accounting for the ecosystem as a whole.

4.1. Ecopath model

The present study provides, to the best of our knowledge, the first attempt to evaluate the potential impact to the shrimp fisheries in Brazil using an ecosystem-based approach with an EwE model. We developed a mass-balanced Ecopath model to describe the trophic interactions and energy fluxes, followed by a temporal dynamic Ecosim model to assess the response of the marine resources and ecosystem conditions under different fishing management scenarios (FMS) for the Barra of Sirinhaém coast as a case study for north-east Brazil.

The evaluation and validation of the structure and the outputs of the model was evaluated through the pre-balance (PREBAL) tool (Link, 2010), which identifies possible inconsistencies in input data (Heymans et al., 2016; Link, 2010). In general, our input data for the Ecopath model followed the general rules/principles of ecosystem ecology,

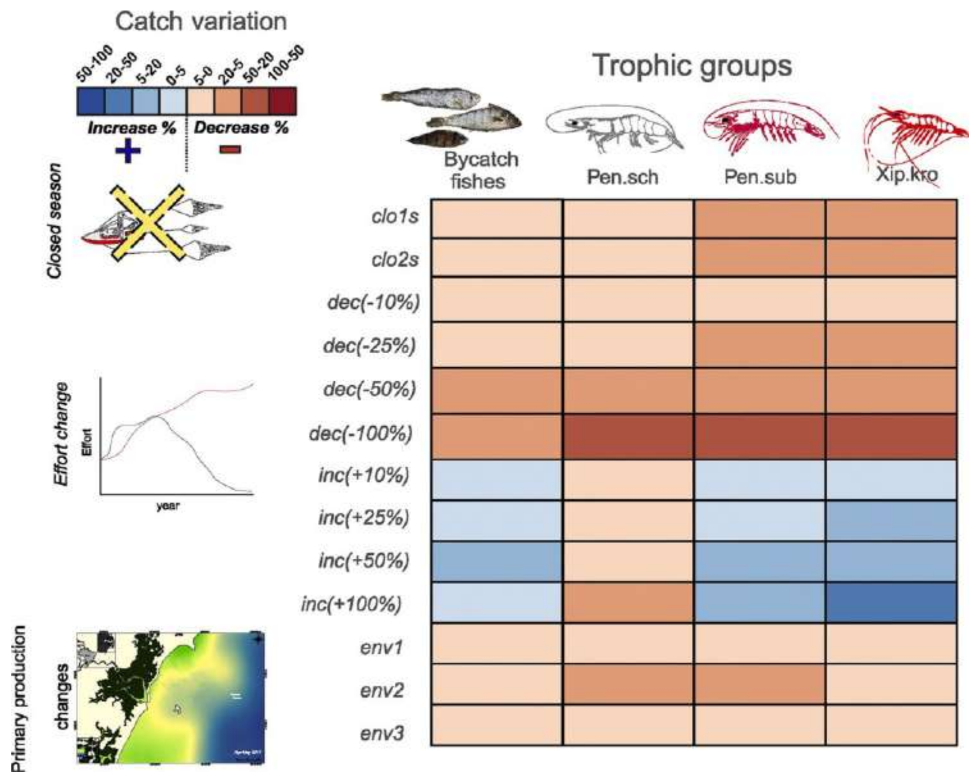

Fig. 6. Average catch variation for shrimp and by fish catch as simulated using the Fishing Management Scenarios from 2015 – 2030 compared to the baseline model (effort constant).The blue and red-coloured gradient indicates increased and decreased catches, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

similar to other studies (Alexander et al., 2014; Bentorcha et al., 2017).

Energy flow in the food web was based mainly from the primary producers, while the indicators of the ecosystem structure in the BSIR model were similar to those of the others coastal models (Geers et al., 2016), with values of respiration and consumption lower than exports and detritus values, and a high value of total primary production/total respiration (TPP/TR). The BSIR model had higher Overhead (SO) than Ascendancy (AC), and low values of connectance index (CI) and Finn's Cycling Index (FCI), similar to the other coastal ecosystems, such as the Isla del Coco, Costa Rica (Fourriére et al., 2019), coral reef Media Luna, Honduras (Cáceres et al., 2016) and the temperate coastal lagoon Ria de Aveiro, Portugal (Bueno-Pardo et al., 2018). In mature systems, the Primary Production rate (TPP) is similar to the respiration flow (close to 1), while the total biomass of the ecosystem is larger than the TPP (Christensen et al., 2005; Odum, 1969), causing an accumulation of biomass within the system compared to the productivity (Corrales et al., 2017). PP-based ecosystems, with relatively low CI and FCI, suggests a low trophic complexity and reduced resilience level (Odum, 1969).

These indicators are considered to be good indexes of the food web complexity, robustness and, indirectly, of the ecosystem maturity and stability (Christensen and Pauly, 1992; Saint-Béat et al., 2015). However, due to the dependence of this indexes to model structure (number of trophic compartments), they often do not reflect the structure of the ecosystem with accuracy (Bueno-Pardo et al., 2018; Christensen et al., 2005; Finn, 1976).

The high system overhead value in the BSIR, and the results reported for other indicators (TPP/TR, TPP/TB, AC, CI and FCI), suggest that the BSIR is an ecosystem in development with a low degree of resilience and low trophic complexity, similar to other coastal systems explored by fishing (Gulf of Mexico, Zetina-Rejón et al., 2015; Tunisia, Hattab et al., 2013; Israeli, Corrales et al., 2017; and China, Rahman et al., 2019)). Although different models presented similar patterns, given the high dynamics, as in the case of coastal ecosystems (e.g, bays, reefs, lagoons and shelfs), it is not possible to set a reference level for all systems, regardless of size, depth, or type of ecosystems (Heymans et al., 2014). The shallow coastal zone, as the present study area, is influenced by different anthropogenic stressors (e.g., tourism, fishing, pollution, etc.), which can affect the ecosystem, providing barriers to evolution towards a more stable state, complex and mature of ecological succession (Bueno-Pardo et al., 2018). Therefore, these ecosystems require particular strategies to maintain the equilibrium state, such as ecosystem-based management integrating the different coastal and marine areas (Dell' Apa et al., 2015; Lazzari et al., 2019), considering the functional limits and the different stressors of each systems.

4.2. Ecosystem historical state

The Ecosim model was able to reproduce the catches and their trends for shrimp species (*P. subtilis, P. schmitti* and *X. kroyeri*) given our available time series data. The trends in our model showed the bottom- up role provided by environmental variability in the function and structure of the ecosystem. Similar results were obtained from other studies in the Mediterranean Sea (Coll et al., 2016; Macias et al., 2014), west coast of Scotland (Serpetti et al., 2017), West Florida, USA (Cha- garis et al., 2015) and Barra del Chuy, Uruguay (Lercari et al., 2018). The nutrient availability, and consequently the primary production, is considered a key controller of biological processes, driving bottom-up processes in the food web (Piroddi et al., 2017). In the BSIR region, the species abundance is strongly associated with environmental drivers (Silva Júnior et al., 2019), for example, the highest chlorophyll con- centration in the rainy season in shallow waters near the mouth of river, where the primary fisheries operate, and the sea surface temperature (SST) impact on shrimp abundance and consequently the fishing productivity (Lopes et al., 2018).

The historical reconstruction from the fitted model for the BSIR reported increases in indicators associated with the biomass, catch, size,

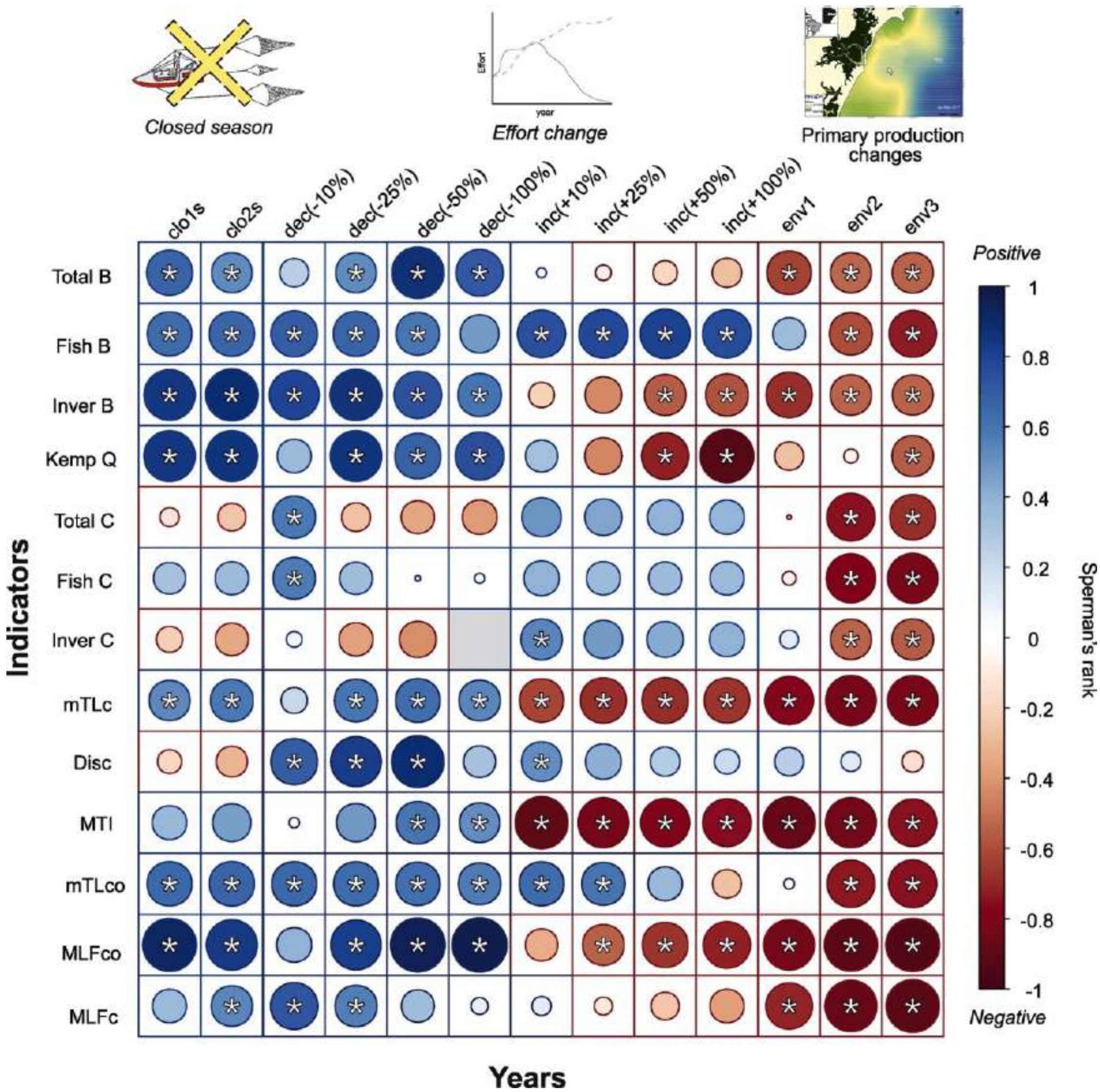

Fig. 7. Spearman's rank correlation between ecological indicators (see Appendix Table 2 for detail) and the temporal scale for the future scenarios (2015 – 2030, see Table 1 for detail) in the Barra of Sirinha´em, Pernambuco, north-eastern Brazil. The blue to red coloured gradients indicate positive and negative correlations, respectively. The colour intensity and size of the circles are proportional to the correlation coefficients Rho. The significant correlation between the indicators and over time (t-test, $p < 0.05$) are represented with a white * symbol. Total B: Total biomass, Fish B: Biomass of fish, Inver.B: Biomass of invertebrate, Kemp.Q: Kempton's biodiversity index, Total C: Total Catch, Fish C: Catch of all fish, Inver.C: Catch of all invertebrate, Disc: Total discarded catch, mTLc:Tropic level of the catch, mTLco: Trophic level of the community (including all organisms), MTI: Marine trophic index (including organisms with TL ≥ 3.25), MLFco: Mean length of fish community, MLFc: Mean length of fish catch. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

trophic level and biodiversity between 1994 and 1997 and 2004 and 2009, given the increase in primary productivity. This pattern could have been caused by climate anomalies (e.g., El Niño and La Niña), which directly influences the changes in terrestrial and marine environmental conditions at both global and regional scales. There are changes in the environmental variables over time, and the SST, precipitation, salinity and chlorophyll concentration are essential for under- standing the effects of the ecosystem dynamics on marine populations (Cloern et al., 2014; Falkowski et al., 1998; Hughes et al., 2017) and consequently affecting the productivity, fisheries, pollution, ecosystem health, socioeconomics, and governance in coastal oceans (Sherman, 2014a, 2014b). Anomalous climate events have been observed since 1950 and have been intensified with the effects of climate change, particularly during the 1997–1998, 2015–2016 (El Niño) and 2007–2008 (La Niña) (Trenberth, 2019) events, leading to profound impacts on biodiversity and humans, since floods, droughts, heat waves, and other environmental changes have modified the ecosystem dy- namics of the region (Marrari et al., 2017; Rossi and Soares, 2017). Although a growing trend in biomass-based indicators (Total B, Fish B and Inver B) has been observed over time, a decline in the mean trophic level of the catch and the mean length of the fish community at the end of the analysis period was reported, which reflected the increased dis- cards and invertebrate catches in the system. It is important to indicate that the historical model calibration and adjust was performed considering only shrimp groups fitted by time-series. Although, no time series were available for the bycatch (e.g., squid, fishes, turtles and etc.) requiring caution when interpreting the results (Piroddi et al., 2017), in general, the historical reconstruction and predictions to future of our model were satisfactory. Often, due to absence of biomass or capture data of the non- target organisms, the studies with EwE approaches mainly focus on exploited species (Abdou et al., 2016; Bornatowski et al., 2017; Coll et al., 2013; Niiranen et al., 2012).

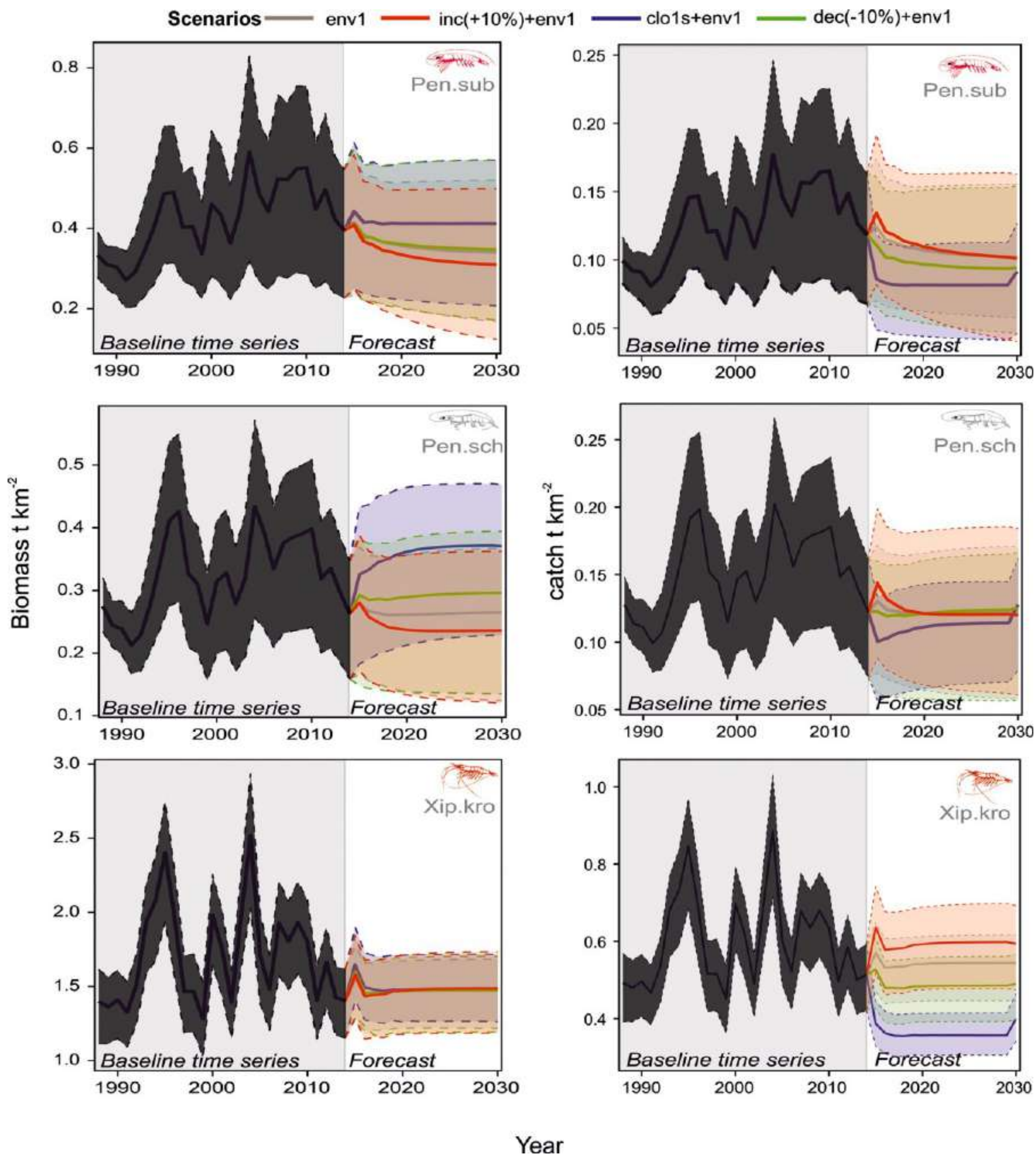

Fig. 8. Comparison between the predicted biomass (t km–2) and catch (t km–2. year—1) for shrimp species from cumulative scenarios for PP anomalies and simulated fisheries management from 2015 to 2030 (see plot legend for details). The black line represents historical model predictions and the coloured lines represent different scenarios. Shadows represent the 5 and 95 % percentiles obtained using the Monte Carlo routine with 1000 runs. Pen.sub: *Penaeus subtilis*; Pen.sch: *Penaeus schmitti*; and Xip.kro: Xiphopenaeus kroyeri.

4.3. Fishing management scenarios (FMS) for the future

Banning trawling fishing as a management measure, whether for a time or an area, has promoted improvements in the ecosystem, with shrimp population recovery, reduced bycatch and benefits for birds, mammals and most fish stocks (Heath et al., 2014; Joseph John et al., 2018). These positive effects through the food web are not always directly related to decreases in anthropic activities, but could also cause indirect consequences to prey-predator relationships (Kempf et al., 2010; Meekan et al., 2018). Conversely, increased fishing efforts may cause significant negative impacts over time on the target species biomass (Ngor et al., 2018; Szuwalski et al., 2017), also indirectly affecting other groups in the food web (Gasche and Gascuel, 2013). In our long-term analysis, when considering the closed fishing period and effort reduction, the model predicted the increased abundance of several bycatch species as well as that of *P. subtilis* and *P. schmitti*. However, the fishing increase caused a decline in biomass for these groups, in the more intense fishing scenarios. For example, a slight decrease in bycatch biomass, primarily in predators of invertebrates, engendered a cascade effect in the food web, increasing the biomass of benthic invertebrates (except for *P. subtilis*, *P. schmitti* and *X. kroyeri*), zooplankton and primary producers (phytoplankton and macroalgae). In addition, the target species catches declined during the simulated season that was closed to bottom trawling. Shifts in fishing effort and catchability, fluctuations in population abundance, market-related factors and environmental change influence catch rates and may confound the potential effects of the management measures (Kerwath et al., 2013; Stefansson and Rosenberg, 2005). Nevertheless, an important step to investigating the

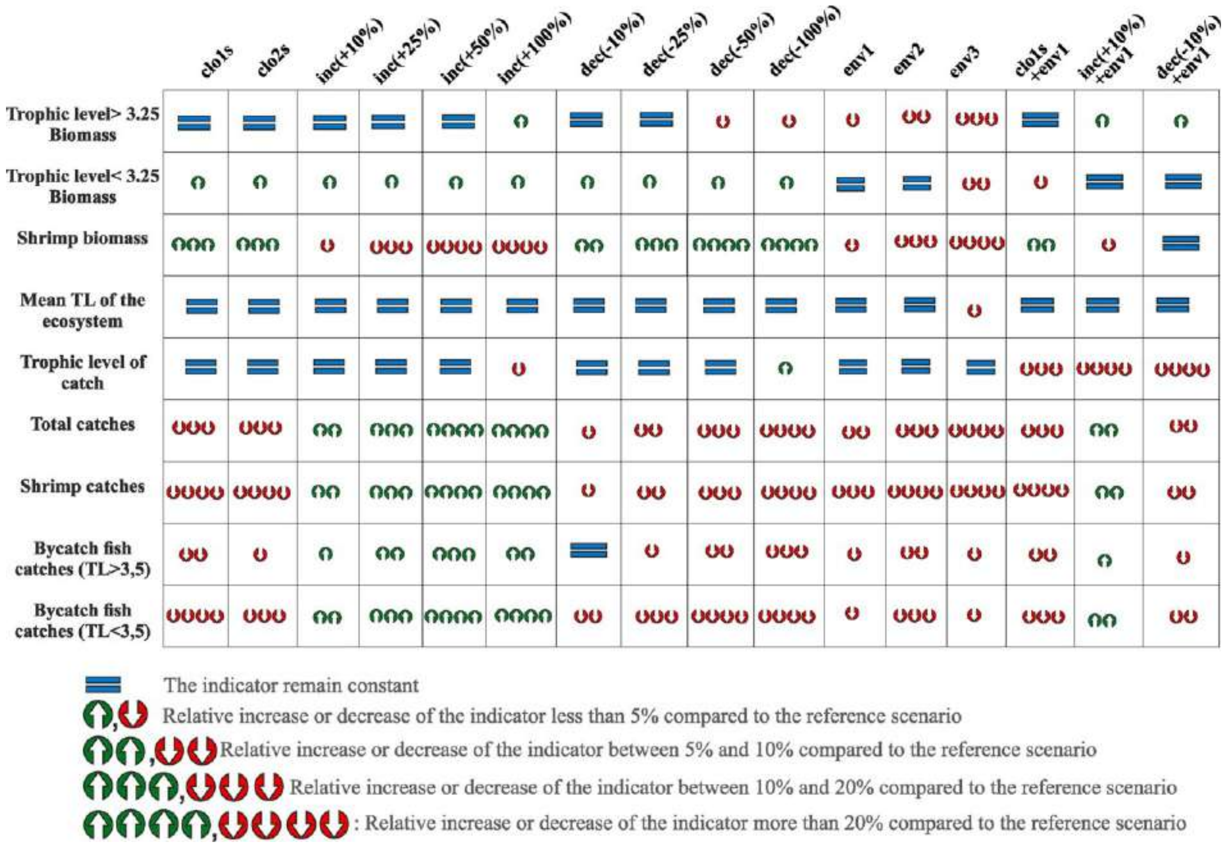

Fig. 9. Summary of the projected responses in fishing management scenarios and environmentally driven previsions in terms of conservation and exploitation indicators. For more detail about each scenario, see *Table 1.*

impact of management strategies on conservation or environmental recovery includes the insertion and evaluation of multiple species at several trophic levels and their trophic interactions (Baudron et al., 2019; Christensen and Walters, 2005).

Intense negative effects on biomass, catch and biodiversity indicators (e.g., Kempton' s biodiversity - Kemp Q) were reported in decreasing scenarios from 1 % PP, reinforcing the need to simulate and project the possible impacts caused by climate change. Although PP is critical in maintaining biodiversity and supporting fishery catches, predicting the responses of populations associated with primary production changes is complex (Brown et al., 2010). Climate change will impact the food web. Ocean warming, for example, has the capacity to drive an energetic collapse at the base of marine food webs, and this effect can propagate to higher trophic levels, subsequently leading to significant biomass decline within the entire food web (Ullah et al., 2018).

Temperature change simulations are most often reported, indicating the reduction in both the number of species and the trophic interactions in the ecosystem (Gibert, 2019; Petchey et al., 2010; Régnier et al., 2019). Doubleday et al. (2019) observed that the enrichment of CO_2 responsible for ocean acidification intensified the bottom-up and top-down control. The effects of warming and acidification is noted in Goldenberg et al. (2018) as a driver of changes in consumer assemblages in future oceans. Moreover, Nagelkerken et al. (2020) indicate cumulative and adverse changes in the whole trophic structure, emphasizing that the adaptive capacity of ecosystems with unbalanced food web to global change is weak and ecosystem degradation is likely. Specifically, in the BSIR, the environment and shrimp fishery dynamics are influenced by primary production fluctuation as controlled by precipitation patterns, which directly affect the fishing activity. The major importance of the temperature and precipitation in shrimp productivity is also re- ported by Lopes et al. (2018), highlighting that these fisheries could collapse in a warmer and drier future.

Our projections highlighted some evidences that the control of bot- tom trawling activity helped to reduce, even at low levels, the highly adverse effects due to primary production reduction. The impacts of climate change on organisms and ecosystems is an imminent reality, and therefore the search for measures for mitigating and even minimizing these impacts is crucial. Historically, less developed regions in terms of fishery governance, as in our case study those primarily associated with small-scale fisheries, are more vulnerable to climate change (Johnson and Welch, 2010) due to the greater difficulty of adapting to productivity loss scenarios (McIlgorm et al., 2010). Some climate change con- sequences might be locally positive for some areas and targeted populations with efficient management measures, but for many fisheries and species, the effects will be undesirable (Quentin Grafton, 2010), for example, the catch decrease in the BSIR.

At the ecosystem level, the increased effort scenarios and PP reduction did not reflect an overall improvement in marine resources. Thus, several ecological indicators displayed a downward trend, such as the Kempton' s Q biodiversity Index, MTI, mTLc, and mTLco. An increase in the bycatch biomass has also been reported. Monitoring these ecosystemic indicators (Cury and Christensen, 2005; Fulton et al., 2004; Heymans et al., 2014) may help researchers to detect food web changes and ecosystem sensitivity to fishing (Coll and Steenbeek, 2017; Halouani et al., 2019; Shin et al., 2018). For example, significant decreases in Kempton' s Q and MTI indices over time indicate negative effects on the ecosystem due to the decline of high trophic level species (Ainsworth and Pitcher, 2006; Piroddi et al., 2010), while the reduction of the mTLco is attributable to the reduction of the biomass for most ecosystem components, primarily the predators TL > 3.25 (Coll et al., 2008; Cor- rales et al., 2018). The improvement of some of these indicators during the closed fishing period represented a rebuilding of the total biomass, including high trophic level species as well as discard reduction. However, the reduced capture of target species by bottom trawling must be better evaluated from a social-economic viewpoint.

4.4. Uncertainty and limitations in BSIR

The integration of ecosystem models, such as the trophic models in fisheries management process, is appreciated because it can address fisheries policy questions (Baudron et al., 2019; Bauer et al., 2019; Christensen and Walters, 2005; Coll and Libralato, 2012). However, it depends on the ability of the ecosystem model to reproduce, in detail, the observed trends and patterns in nature (Christensen and Walters, 2005; Cury and Christensen, 2005; Steenbeek et al., 2018), usually including the environmental effects, uncertainty estimates and confidence limits (Ehrnsten et al., 2019; Guesnet et al., 2015). Recently, several data based gaps have been described in previous studies using EwE models (Ecopath, Ecosim and Ecospace) (Chagaris et al., 2015; Corrales et al., 2018; Geers et al., 2016), especially those related to the lack of trophic information with a temporal dimension, reliable histor- ical catch data and fishing efforts, limited information on biomass (Piroddi et al., 2017) and migration among habitats for different species (Halouani et al., 2016).

Thus, developing this ecosystem approach, particularly on the north- east coast of Brazil, is a challenging task, primarily due to the difficulties involved in gathering and integrating good-quality local data (e.g., dietary information, fishing data, environmental features, etc.) as reported by Lira et al. (2018). Despite this concern, the BSIR model was built on the basis of local studies and specific sampling in the area to estimate the biomass of several groups (all fish and shrimp species), and the diets and stable carbon and nitrogen isotope compositions of the primary consumers (see Supplementary Information). However, the absence of time series data for a large number of groups (e.g., catches, biomass and fishing effort) is considered as our primary weakness. Alternatively, to minimize the limitations cited above, we performed a sensitivity analysis (Monte Carlo routine) to evaluate the uncertainty around model parameters and to assess, in our case, the biomass and ecological indicators (Christensen and Walters, 2004; Niiranen et al., 2012; Steen- beek et al., 2016). In addition, although we recognize the importance of incorporating specific periods of the closing season within scenarios, some major data, as for example the spawning parameters (egg pro- duction, egg-laying timing etc.), are lacking, hampering this analysis within the model.

We are confident that our study presents a satisfactory representation of the ecosystem structure and the fishing impact on the ecosystem and may be replicable to other small scale shrimp fisheries. In addition, incorporating additional tools to the current model, such as Ecospace, to investigate the potential impacts of spatial management plans (e.g., area closed to fishery), and tools to assess the cumulative effect of future climate change (e.g., sea temperature, species distribution change, and phenological changes) on small-scale fisheries would enable useful in- sights into the effects of various management policies and possible trade-offs at the ecosystem level.

4.5. Management support tool

Multiple indicators were considered in the context of Ecosystem-Based Fishery Management to evaluate the potential effects of different FMS with the aim of providing a straightforward set of decision parameters to small-scale fisheries managers, specifically to bottom trawlers, to fulfil both fisheries and conservation management objec- tives in the near future. In general terms, the decreased trawling efforts were promising, with better fishing management performance than the closed fishing periods of 3 and 4 months, primarily due to significant losses in the catches of high market-value target species (e.g., the white shrimp *P. schmitti* and the pink shrimp *P. subtilis*) and bycatch fishes considered as byproduct in these scenarios.

Some aspects of the BSIR that may be shared with other locations should be considered within the management framework. The shrimp fishing dynamics are well-defined yearly. Shrimp and bycatch are abundant and are mainly caught during the periods of highest primary production as a consequence of the rainfall (Silva Júnior et al., 2019). At the opposite, the lowest shrimp and bycatch abundances and catches are related to dry periods, which correspond to the peak of reproduction of these species (Eduardo et al., 2018; Lira et al., 2019; Lopes et al., 2017; Peixoto et al., 2018; Silva et al., 2016; Silva Júnior et al., 2015). Consequently, during the dry season, the trawling activities are basically inactive due to the decline in production (Eduardo et al., 2016; Silva Júnior et al., 2019; Tischer and Santos, 2003), barely covering the operating costs of the fishery. This phenomenon could be considered as a "*natural closed season*", or the economically unprofitable due to low shrimp and bycatch abundance that regulates the fishing activities. In addition to the importance of the target species, knowledge of the bycatch destination is crucial during the management process. In the BSIR, the incidental catch primarily removes juveniles (Eduardo et al., 2018; Lira et al., 2019; Silva Júnior et al., 2015), which are often consumed by the fishermen and local community as additional sources of food and income as a byproduct(Silva Júnior et al., 2019). Thus, a major decline in the capture of bycatch with the implementation of a management measure may cause negative effects from nutritional, economic and social viewpoints. In this way, the impact of the fishing activities on the ecosystems appears to be counter-balanced by the beneficial role of the bycatch in the local community. Although we are aware of the importance of this fishery bycatch for the local food security, we cannot disregard the fact of several fish species of bycatch (e. g., croaker, weakfish, jacks, snappers) has the longer life history, low spawning potential, and high commercial value when adults, and therefore need to be considered in future evaluations, including new information incorporating the socio-economic aspect.

Considering the particularities of our case study and without ac- counting for the effect of environmental changes, not adopting effort control measures for the current trawling conditions (baseline scenario) do not appear to cause major losses in terms of biomass and catches. However, it is clear that in the near future (2030), with the uncontrolled increase >50 % in trawling combined with environmental changes, for example, in the rainfall or in primary production, significant adverse impacts will affect the ecosystem functioning. In these cases, bottom trawling control efforts can help to mitigate, even at low levels, these highly negative effects.

Our findings indicate that it is possible to maintain the same level of landings with a controlled reduction of bottom trawlers activities, for example, close to 10 %, without compromising the ecosystem structure. However, other management measures could be incorporated into the model and better evaluated in the future, such as the application of Bycatch Reduction Devices (e.g., fisheye, grid and square mesh) used to exclude small fish, juveniles of species of high commercial value (e.g., croaker, weakfish, jacks, snappers) and other non-target species from the trawlers (Broadhurst, 2000; Eayrs, 2007; Larsen et al., 2017); an increase in the area and/or improvement in enforcing the existing Ma- rine Protected Areas (e.g., MPA Guadalupe) as well as including other environmental drivers from the IPCC predictions (e.g., RPC4.5 and RPC8.5) (Reay et al., 2007). These measures would enable important and useful insights on the direct and indirect effects of climate changes, other management policies, and possible trade-offs at the ecosystem level. However, any management measures to be considered as successful to mitigate the fishing impacts depend on interactions among highly heterogeneous social, political, economic and conservation fac- tors, which are especially relevant in small-scale fisheries such as our case study fishery.

Credit author statement

Alex Lira: Sampling procedures, laboratorial analysis, data analysis and manuscript preparation.

Flávia Lucena-Frédou: Sampling procedures, data analysis and

manuscript preparation.
François Le Loc'h: Data analysis and manuscript preparation.

Declaration of Competing Interest

The authors report no declarations of interest.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.fishres.2020.105824.

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Chapter main findings and Thesis outlook

In this Chapter, to the best of our knowledge, this was the first attempt to evaluate the potential impact to the shrimp fisheries in Brazil using an ecosystem-based approach with an EwE model (Figure 10). The trends of ecosystem indicators (e.g., biomass-based, trophic and size-based indexes) revealed the bottom-up role provided by the environmental variability over the functioning and structure of the ecosystem. As already highlighted in the Chapter 1 (sec. *Characterization of the abiotic condition of the shrimp fishing sites*), the species abundance is strongly associated with environmental drivers In this Chapter, by modelling, we have demonstrated that the highest chlorophyll concentration in the rainy season, can impact the shrimp abundance and consequently the fishery productivity. This effect is more decisive over the ecosystem and fishing balance than management measures as closed season and variations of the fishing effort in ±10%. However, it is evident that in the near future (2030), with the uncontrolled increase of trawling combined with environmental global changes, significant adverse impacts will affect the ecosystem functioning (Figure 10). Yet, a controlled reduction of bottom trawlers activities, may help to reduce, even at low levels, these highly adverse effects and to maintain the similar level of landings, without compromising the ecosystem structure.

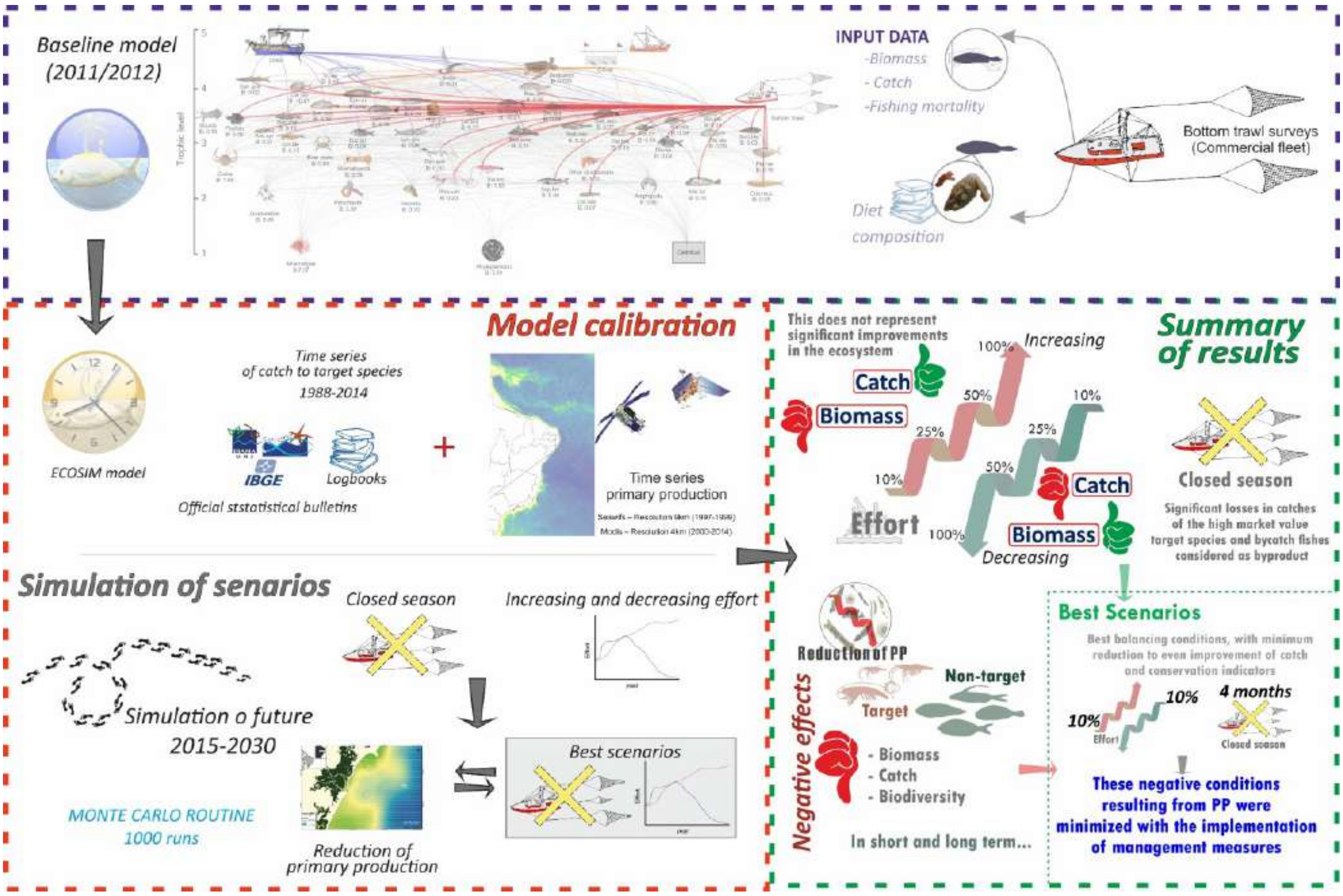

Figure 10. Schematic of the process framework used to build and calibrate the Ecopath with Ecosim (EwE) model of Barra of Sirinhaém, Pernambuco, north-eastern Brazil.

The limited amount of information available mainly to fish bycatch, has restricted our conclusions in order to identify, at the species level, species the most vulnerable to trawling, which deserve special attention by managers. Hence, in the next Chapter, we apply and adapt another ecosystem approach, and information on the species and fishery were obtained through the Chapters 1, 2 and 3 in order to assess the vulnerability of the species caught by the shrimps fishery (target and bycatch). Considering the world relevance of the small-scale fisheries, and their bycatch in particular, which are usually neglected by assessment approaches and hence by the decision makers, we evaluate the vulnerability and the potential risk on a specific level of the target and non-target species exploited by the shrimp fishery. For this, in Chapter 4, we apply a semi-quantitative Ecological Risk Assessment (ERA), the PSA (Productivity and Susceptibility Analysis). Within the family of data-limit model, PSA is function of species biological productivity and the susceptibility to be captured by a specific fishing gear. In addition, we bring an adapted approach to regional conditions, incorporating uncertainties to allow for a better confidence of the results. We expect that our model could be replicated in other tropical fisheries where the limitation of information hampers the identification of priority species for the management and conservation actions by decision makers.

Chapter 4

Vulnerability of marine resources affected by a small-scale fishery in Northeast Brazil

CHAPTER 4. *Vulnerability of marine resources affected by a small-scale tropical shrimp fishery in Northeast Brazil*

Introduction

Bottom trawls are a major type of fishing gear used worldwide (Hintzen *et al.*, 2020), responsible for almost a quarter of marine landings (Watson and Tidd, 2018). Although economically important, bottom trawling causes significant adverse impacts on seabed habitats and biota (Jones, 1992; Kaiser *et al.*, 2002; Johnson *et al.*, 2015; Davies *et al.*, 2018; Ortega *et al.*, 2018b), including a high quantity of bycatch and discards (Zeller *et al.*, 2017). Such effects also lead to losses of protein sources, affecting food security and the fishery sustainability (Srinivasan *et al.*, 2010; Belton and Thilsted, 2014). Impacts vary in intensity depending on the size and technology of the fleet concerned (Amoroso *et al.*, 2018).

In the southwestern Atlantic Ocean, along the Brazilian continental shelf, shrimp trawling is a very common fishery activity, operating at three scales: (i) industrial, present in the North (Amazon river estuarine system), Southeast and South Regions of Brazil; (ii) semi-industrial, with an intermediate technology and fishing power, and (iii) artisanal, operating along the entire coast and involving a larger number of people but lower levels of technology, capture and profit (Dias-Neto, 2011). Management measures for bottom trawlers are mainly based on closed seasons (Dias-Neto, 2011; Nakamura and Hazin, 2020) and, particularly for the industrial fleet, the Turtle Excluder Device (TED). Apart from the TED, all other recommendations available in the country focus only on target species, thus neglecting the bycatch.

Brazilian fisheries were officially monitored up to 2010. At that time, the bottom trawling fleet was one of the largest and most productive in Northeast Brazil, involving more than 100 000 persons, about 1700 motorized and 20 000 non-motorized boats (Santos, 2010), representing approximately 10% of the total marine landings in the country (IBAMA, 2008). Within this region, the shrimp fishery in Barra de Sirinhaém (BSIR), south of Pernambuco, is predominantly small-scale, and accounted for 50% of the state shrimp production (Tischer and Santos, 2003) in the decade 2000–2010, representing an important source of income and food for the local population (Lira *et al.*, 2010).

The incidental catch of the shrimp trawl fisheries in the region of BSIR represents about 26% of total landings, primarily removing juveniles, which are often consumed by the fishermen and local community as an additional source of food, or sold as a by-product (Tischer and Santos, 2003; Silva Júnior *et al.*, 2019; Lira *et al.*, 2021b). In this case, the impact of the fishery on the ecosystems appears to be counterbalanced by the beneficial role of the bycatch for local communities (Carvalho *et al.*, 2020; Lira *et al.*, 2021b). However, despite its high relevance, the shrimp trawl fishery in Pernambuco currently has no regulations (Santos, 2010; Silva Júnior *et al.*, 2019; Lira *et al.*, 2021b), mainly due to a lack of knowledge about the bycatch. This hampers the inclusion of the incidental catch in assessment models (Yonvitner *et al.*, 2020), increasing the 'risk' (here defined as the probability of something undesirable happening to stocks; Francis and Shotton, 1997; Sethi, 2010) to these non-target species.

Tropical fisheries, including those in Brazilian waters, are multispecific (Frédou *et al.*, 2006, 2009a, 2009b; Chrysafi and Kuparinen, 2016; Zhou *et al.*, 2019). The scenario is one of great diversity of species and limited data. In the last two decades, a wide range of qualitative and quantitative assessment approaches for data-limited fisheries has been developed to support fisheries management, including quantitative life-history (Le Quesne and Jennings, 2012), catch and length-based models (Hordyk *et al.*, 2015b), and qualitative and semi-quantitative methods, such as risk analysis (Hobday *et al.*, 2007, 2011). Productivity and Susceptibility Analysis (PSA) is a semi-quantitative risk analysis method that relies on the relationship between the biological productivity related to the life history characteristics (Stobutzki *et al.*, 2001; Hobday *et al.*, 2007) and the susceptibility of the stock to fishing (Patrick *et al.*, 2010; Lucena-Frédou *et al.*, 2017).

The PSA approach is a well-accepted framework for estimating the vulnerability of species to fishing, having already been used in several fisheries around the world, e.g. in the Australian northern prawn fishery, Stobutzki *et al.* (2001); the tuna longline fleets in the South Atlantic and Indian Ocean, Lucena-Frédou *et al.* (2017); Alaska groundfish, Ormseth and Spencer (2011); gillnet fishing in the Bangladesh, Faruque and Matsuda (2021); and multiple gears in the Skagerrak–Kattegat (eastern North Sea), Hornborg *et al.* (2020). However, risk analysis has been little used in tropical fisheries (Clarke *et al.*, 2018) and only three studies have been made on the Brazilian coast. Two of these used the susceptibility method that existed prior to PSA (described by Stobutzki *et al.*, 2001)) to assess the sustainability of ornamental fish caught in a trap fishery (Feitosa *et al.*, 2008) and fish bycatch by shrimp trawling (da Silva *et al.*, 2013) in the Northeast Region. In Brazil, PSA has only been applied to large scale fisheries, such as the gillnet fishery in southeast Brazil (Visintin and Perez, 2016). PSA has rarely been applied to any small-scale fisheries worldwide (Micheli *et al.*, 2014; Martínez-Candelas *et al.*, 2020; Yonvitner *et al.*, 2020), and has never been reported in Brazil. This approach is quite a promising member of the family of data-poor models, but its minimal data requirements and relatively subjective nature are weaknesses. Few studies address these uncertainties regarding input parameters and calculation procedures (Lucena-Frédou *et al.*, 2016; Duffy and Griffiths, 2019; Altuna-Etxabe *et al.*, 2020; Baillargeon *et al.*, 2020), which has led PSA to be strongly criticized (Hordyk and Carruthers, 2018). In addition, given the particularities of different fishing gears and ecosystems, the approach must be adapted to the particular circumstances of each case study, taking into account appropriate attributes and scores.

Despite the world relevance of small-scale fisheries, particularly their bycatch, they are usually neglected by assessment approaches and hence by decision makers. Our study evaluates, for the first time, the vulnerability and potential risk of the target and non-target species exploited by the shrimp fishery in Sirinhaém coast as a case study of a small-scale fishery in Northeast Brazil. For this, we used a PSA adapted to regional conditions, while also assessing any effects of the intrinsic subjectivity of the method. We believe that this approach could also be used to assess other tropical fisheries where uncertainties and limited information hampers management and conservation action by decision makers.

Material and methods

Study area and gear description

Barra of Sirinhaém (BSIR), located on the southern coast of Pernambuco, in Northeast Brazil (Figure 1), has a tropical climate, with precipitation ranging from 20 to 450 mm·month^{-1} and a rainy season between May and October. The mean surface water temperature is 29°C, pH varies between 8.0 and 8.7, and salinity between 23 and 37 (Mello, 2009; APAC, 2015). Fishing, the sugar cane industry and other farming industries are the main anthropic activities in the area (CPRH, 2011). The fishing zones are inside or close to the marine protected areas around Santo Aleixo Island (MPAS of Guadalupe and Costa dos Corais) (Figure 1). The fleet operates from 1.5 to 3.0 miles off the coast, mainly between 10 and 20 m depth. Hauls last from 4 to 8 hours and boat velocity varies between 2 and 4 knots. Boats measure 8–10 m in length, nets have horizontal opening of 6.1 m, and mesh sizes of the body and codend are 30 mm and 25 mm, respectively.

Target and non-target species

Fish and shrimp captures were first assessed monthly (August 2011 to July 2012) and then quarterly (October 2012 to July 2014) by accompanying the local trawling fishers (for details see Silva Júnior *et al.*, 2019; Lira *et al.,* 2021). Penaeid shrimps are the main targets, particularly seabob shrimp (*Xiphopenaeus kroyeri*), which is the most abundant, and pink shrimp (*Penaeus subtilis*) and white shrimp (*Penaeus schmitti*), which have higher market values (Santos, 2010). The amount of fish bycatch is 0.39 kg of fish captured for each 1 kg of shrimp (Silva Júnior *et al.*, 2019). Non-target fishes (bycatch species) are composed of 87 species, 21 orders and 35 families, including teleosts and elasmobranchs (Tischer and Santos, 2003; Silva Júnior *et al.*, 2019). The five families most highly represented in the bycatch (in number and weight), Pristigasteridae, Scianidae, Haemulidae, Ariidae and Trichiuridae, represented, on average, 82% of the total catch (Tischer and Santos, 2003; Silva Júnior *et al.*, 2019) (Fig. 1). Thus, ninety species (87 non-target fish and 3 main target shrimp species) caught by trawling fishing in the region were considered in the PSA approach.

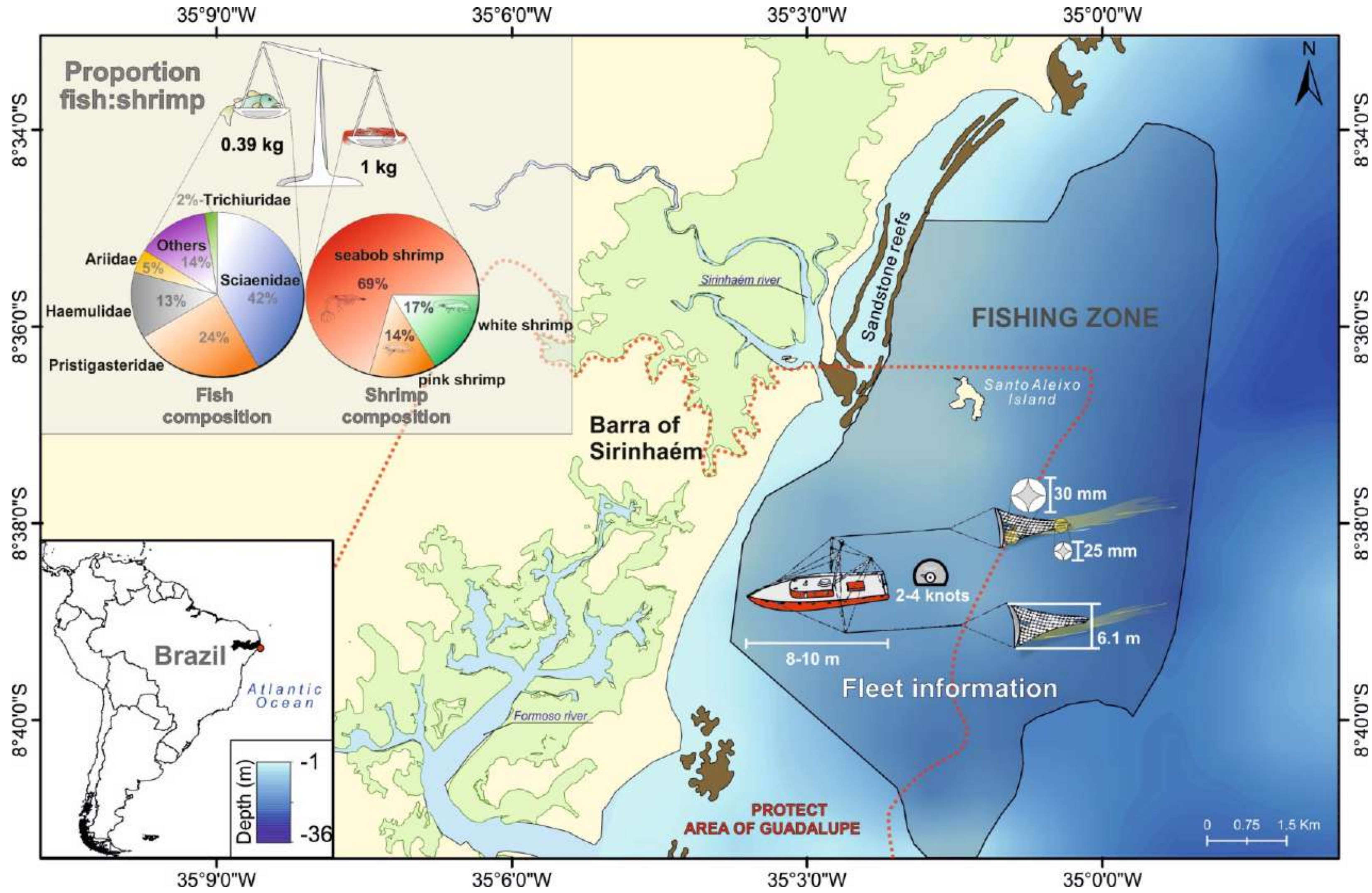

Figure 1. Study area, gear description and catch composition by bottom trawl fishing in Barra of Sirinhaém (BSIR), south of Pernambuco, Northeast Brazil (sources: Silva Júnior et al. (2019); Lira et al. (2021)).

Vulnerability approach

The vulnerability assessed by PSA refers to the risk potential of a stock with regard to a specific fishing gear (Patrick *et al.*, 2009). It is defined as a function of productivity and susceptibility attributes (Stobutzki *et al.*, 2001; Hobday *et al.*, 2007; Patrick *et al.*, 2009). Originally, Stobutzki *et al.* (2001) carried out an analysis where the exposure of a species to capture and mortality was taken as the 'Susceptibility', and the capacity of the population to recover after depletion was 'Recovery'. However, Recovery was replaced by the concept of 'Productivity' by Hobday et al. (2007) and Patrick et al. (2009). In the approach of these latter authors, the vulnerability score (v) is obtained by the calculation of Euclidean distance of the weighted productivity (P) and susceptibility (S) scores (see section *Measuring uncertainties* for details):

$$v = \sqrt{[(P - X_0)^2 + (S - Y_0)^2]}$$

where X_0 and Y_0 are the (x, y) origin coordinates of the biplot, respectively.

The species most vulnerable to fishing have low productivity and high susceptibility scores, while the least vulnerable have high productivity and low susceptibility scores (Patrick *et al.*, 2010). Productivity and susceptibility scores are calculated assigning attributes and scores. Each of the productivity (P) and susceptibility (S) attributes (defined below) are scored on a scale of three levels: indicating low (1), medium (2), and high (3) values. When information on attributes are missing, they are not used in the computation of the final P or S scores (Lucena-Frédou *et al.*, 2017).

Productivity

Eight life-history traits correlated with productivity were selected (Table 1) following Patrick *et al* .(2010), Lucena-Frédou *et al.* (2016) and Lucena-Frédou *et al.* (2017). Summaries of these traits are given in the following list, while equations and data details can be found in Supplementary Table S1 and the Supplementary material, respectively.

(i) Von Bertalanffy growth coefficient (k; $cm.y^{-1}$) reflects the speed at which the growth curve reaches the asymptotic length. This attribute is positively correlated with productivity, so species of high and low k value are more and less productive, respectively (Patrick *et al.*, 2010). The k parameter was obtained from the literature or by using the empirical equation of Le Quesne and Jennings (2012).

(ii) Maximum length (L_{max}; cm) is the maximum reported total length of each shrimp and fish species, obtained from our database or from the literature (whichever was larger). In general, species with large L_{max} values have a long life expectancy and, consequently, lower productivity (Roberts and Hawkins, 1999).

(iii) Size at first maturity (L_{50}; cm) is the total length at which 50% of individuals first attain sexual maturity and are capable of reproduction. As L_{50} is negatively correlated with productivity, species with late maturity (high L_{50}) often have slow growth and tend to live longer, resulting in lower rates of population recovery and low productivity. When not available in the literature, the size at first maturity was estimated by the relationship proposed by Froese and Binohlan (2000).

(iv) Intrinsic growth rate (r) represents the intrinsic rate of population growth or maximum population growth that would occur in the absence of fishing at a small size (Gedamke *et al.*, 2007). It was estimated from life history parameters for each species using the approach of Fortuna *et al.* (2014). This parameter is inversely correlated with productivity (see details in Supplementary material).

(v) Trophic level (TL) indicates the trophic position of the species and the potential role in the food-web. Considering the trophic pyramid theory (Lindeman, 1942), TL is often inversely proportional to productivity. TL values were obtained from the EwE model developed in the same region (Lira *et al.*, 2021b), and when unavailable, from the literature.

(vi) L_{50}/L_{max} reflects the ratio of the relative investment in somatic and reproductive growth. Small-sized species are usually more productive and tend to reach sexual maturity at relatively larger sizes compared with their maximum size, whereas large-sized species reach maturity at relatively smaller sizes (Juan-Jordá *et al.*, 2013).

(vii) Maximum age (A_{max}; y^{-1}) is the maximum reported age for each species, which is inversely correlated with the productivity. When not available in the literature, this parameter was estimated according to the empirical equation proposed by Taylor (1960).

(viii) Breeding strategy is the only non-quantitative attribute and indicates the level of mortality that may be expected for offspring in the early stages of life (Patrick *et al.*, 2010). It is quantified by the index of parental investment described by Winemiller (1989) and modified by King and McFarlane (2003), according to which score values are attributed for i) placement of zygotes or larvae (e.g. no placement or maintained in a nest; score ranges from 0 to 2); ii) parental protection of zygotes or larvae (score ranges from 0 to 4); and iii) nutritive contribution (score ranges from 0 to 8). The sum of these values ranges from 0 (species without placement of zygotes or larvae, parental protection and nutritive contribution) to 14 (species with all these characteristics) (Table S1). Following King and McFarlane (2003) and Patrick *et al.* (2010), species that presented values of 0 were considered as having high productivity and those with values 4 ≥ as having low productivity.

Susceptibility

Three susceptibility attributes related to abundance, distribution and fishery were adapted from Patrick *et al.* (2010) and Lucena-Frédou *et al.* (2017). Given the specificities of the case study, another three attributes are proposed also here (Table 2). See supplementary Table S2 and Supplementary material for details.

(i) Frequency of occurrence and abundance (FOA). We estimated the frequency of occurrence (number of occurrences of a species divided by the total number of trawls x100, %F) and abundance, initially obtained in g m^{-2} by the sum of weight (W; g) caught of each species divided by the estimated swept area (a; m^2): CPUAb = W/a; and then converted into relative values (catch per unit area; %CPUA). The covered area was estimated as: a = D.H.X; where, D is the distance covered (km) obtained by GPS tracking; H is the head-rope length (0.012 km) and X is the fraction of the head rope length = 0.5 (Pauly, 1980). Species showing %FO> average %FO were considered as frequent, whereas those with %FO< average %FO were considered rare (Garcia and Vieira, 2001). A similar method was applied to %CPUA, resulting in Highly Abundant (%CPUA>average %CPUA) and Scarce (%CPUE<average %CPUA) categories. Finally, based on both criteria, species were classified according to Garcia and Vieira (2001) into three groups of differing relative importance (relative importance index): i) abundant and frequent; ii) frequent but less abundant; and iii) less abundant and less frequent (Table S2). In our approach, species with high abundance and frequency were classified as highly susceptible (3) while the less abundant and frequent species were classified as having low susceptibility (1) (Table 2). The more frequently and abundantly a species is caught, therefore, the more susceptible it is considered.

(ii) Percentage of individuals > L_{50} (% > L_{50}) corresponds to the proportion of individuals larger than the length at first maturity (L_{50}), obtained from the length distributions (Lucena-Frédou *et al.*, 2017), calculated only for species with samples that included most of the size spectra of the species (including both juveniles and adults) (Supplementary Figure S1). Species with high percentage of individual with less than L_{50} are more susceptible to fishing.

(iii) Ratio between fishing mortality and natural mortality (F/M) provides an indication of the relative impact of fishing pressure, because the relative values provide a better description of the magnitude of exploitation than the absolute value (Zhou *et al.*, 2012; Huynh *et al.*, 2018). A conservative rule of thumb is that M should be an upper limit of F (Thompson, 1993), hence we considered that F/M should not exceed 1, and values above 1.0 and below 0.5 were defined as high and low susceptibility, respectively (Table 2). We used the 'natural mortality' routine (https://github.com/shcaba/Natural-Mortality-Tool) in the Barefoot Ecologist's Toolbox (Prince, 2003) to estimate the M (see Supplementary material for details). Fishing mortality was obtained as the difference between M and Z, estimated by a Catch curve (Pauly, 1983; Wetherall, 1986) from the FSA package (Ogle *et al.*, 2020), but only for species with representative length frequency distribution as described above (juveniles and adults included) (Supplementary Figure S2).

(iv) Overlap area (OA) is an indicator that aggregates and adapts two susceptibility attributes (based on Patrick *et al.,* 2010) related to the overlap between the fishing gear and the geographic distribution and position of the species in the water column. We considered the behaviour of the species as demersal (DE), pelagic (PE) or reef-associated (RE). We also considered the functional guilds proposed by Elliott *et al.* (2007): marine stragglers (MS), marine migrants (MM) or estuarine (ES) species, which represent the use of the environment by a species over its life cycle. Hence, considering that a bottom trawl (the case in this study) mainly acts on demersal species of shallow marine areas with unconsolidated substrate (e.g. mud and sand), there is a higher overlap of species distribution and fishing effort, even if we recognize that the area of study is only part of the species distribution. Species (DE + MM or MS) and (PE + MM or MS) were considered to be of high and moderate susceptibility to the fishing, respectively. Conversely, species with pelagic (PE) or demersal (DE) behaviour (PE) and estuarine (ES) and reef-associated distribution (RE + MS or MM) (with marginal overlap of the fishing sites) were classified with low susceptibility (Table 2). Information on vertical distribution and functional guild was assessed by an extensive literature review, including articles, books and reports, as well as the FishBase repository (Froese and Pauly, 2019).

(v) Mixed Trophic Impact (MTI) is an index proposed by Christensen *et al.* (2008) and obtained by Lira et al. (2021) from the EwE model developed for the shrimp trawling fishery in the same area of the present study (BSIR). This index represents the positive (increase) and negative (decrease) impacts of one species/group of species over the biomass of another species or group of species (Ulanowicz and Puccia, 1990), considering the natural mortality (M) by the predation and the mortality caused by the fishery (F). A high negative MTI of a fleet (in our case, the shrimp trawling fleet) over a species indicates a high impact due to fishing, and consequently, higher susceptibility (See Christensen *et al.*, 2005, for more detail).

(vi) Length-Based Spawning Potential Ratio (SPR) is a model developed by Hordyk *et al.* (2015) for data-limited fisheries that calculates the proportion of the unfished reproductive potential at any given level of fishing pressure (Walters and Martell, 2004; Patrick *et al.*, 2010). This method requires basic knowledge of the life history parameters (natural mortality rate, M; the von Bertalanffy growth parameters, L_∞ and k; and the length at first maturity, L_{50}), a representative size distribution and the shape of a population's size structure (Hordyk *et al.*, 2015a; Prince *et al.*, 2015). SPR can be used as an alternative reference point to biomass at maximum sustainable yield (B_{MSY}) (Pons *et al.*, 2019), representing a proxy of the biomass of spawners (SSB). An SPR of 100% ($SPR_{100\%}$) indicates an unexploited stock, while $SPR_{0\%}$ represents a stock with no spawning, where all mature fish have been removed, or all female fish have been caught (Hordyk *et al.*, 2015b). An SPR equal to or above 0.4 ($SPR_{40\%}$) is a conservative proxy of the for B_{MSY} (Clark, 2002), here considered as the less susceptible and used by several international fisheries commissions (e.g. International Commission for the Conservation of Atlantic Tunas, ICCAT). An SPR smaller than 0.2 ($SPR_{20\%}$) is a proxy for impaired recruitment rates of a stock (Goodyear, 1993), here the value considered the most susceptible. Similar to F/M attribute, only species with length distributions considered representative of most life history stages (juveniles and adults) were included.

Defining boundaries

The values of productivity and sustainability attributes are classified according to a ranking of three levels (low = 1, moderate = 2, high = 3) (Tables 1 and 2). Given the intrinsic subjectivity of the model, two methods were used to calculate the boundaries of scoring. The first method was the tercile approach, as already used in some previous studies (Lucena-Frédou *et al.*, 2017; Duffy and Griffiths, 2019; Faruque and Matsuda, 2021). A multivariate analysis based on the clustering k-means method (Altuna-Etxabe *et al.*, 2020) (Supplementary Figure S3) was also employed to calculate the bounds. k-means is an iterative method that minimizes the within-class sum of squares for a given number of clusters (MacQueen, 1967; Hartigan and Wong, 1979). In this approach, all attribute scores were considered together and were partitioned into three clusters from minimum distance between each observation to the cluster centres. For the productivity attributes (except breeding strategy) and susceptibility, specifically the MTI and % > L_{50} that do not have boundaries defined in the literature, the categories (high: 3; moderate: 2 and low:1) were defined by using the two approaches described above.

Table 1. Productivity attributes and rankings used to determine the vulnerability of species caught by bottom trawl fishing in BSIR, south of Pernambuco, Northeast Brazil. Boundaries of scoring defined by quantile and k-means methods (for more details see section *Defining boundaries*).*classification from Patrick et al. (2010).

	Attribute	Ranking			Sources
		High (3)	Moderate (2)	Low (1)	
Quantile method	Von Bertalanffy Growth coefficient (k, cm.year^{-1})	> 0.47	0.34 – 0.47	< 0.34	(1,2)
	Maximum length (L_{max}, cm)	< 25.0	25.00 – 42.80	> 42.80	(1,2)
	Size at first maturity (L_{50}, cm)	< 12.0	12.00 – 18.90	> 18.90	(1,2)
	Intrinsic growth rate (r)	> 0.74	0.54 – 0.74	< 0.54	(1,2)
	Trophic level (TL)	< 3.10	3.10 – 3.42	> 3.42	(1,2)
	L_{50}/L_{max}	< 0.50	0.50 – 0.54	> 0.54	(2)
	Maximum age (A_{max}; year^{-1})	< 5.92	5.92 – 8.24	> 8.24	(1,2)
K-means method	Von Bertalanffy Growth coefficient (k, cm.year^{-1})	> 0.93	0.24 – 0.93	< 0.24	(1,2)
	Maximum length (L_{max}, cm)	< 41.68	41.68 – 112.00	> 112.00	(1,2)
	Size at first maturity (L_{50}, cm)	< 19.36	19.36 – 58.40	> 58.40	(1,2)
	Intrinsic growth rate (r)	> 1.52	0.51 – 1.52	< 0.50	(1,2)
	Trophic level (TL)	< 3.15	3.15 – 3.93	> 3.93	(1,2)
	L_{50}/L_{max}	< 0.51	0.51 – 0.53	> 0.53	(2)
	Maximum age (A_{max}, year^{-1})	< 8.48	8.48 – 15.04	> 15.04	(1,2)
	Breeding strategy*	0.00	1.00 – 3.00	≥ 4.00	(1)

(1) Patrick *et al.* (2010); (2) Lucena-Frédou et al. (2017)

Table 2. Susceptibility attributes and rankings used to determine the vulnerability of species caught by bottom trawl fishing in BSIR, south of Pernambuco, Northeast Brazil. FOA, Frequency of occurrence and abundance; OA, Overlap area; F/M, Ratio between fishing mortality and natural mortality; MTI, Mixed Trophic Impact; SPR, Spawning Potential Ratio; % > L_{50}, Percentage of individuals > L_{50}. The classifications of the species for overlap area are demersal (DE), pelagic (PE), reef-associated (RE), marine stragglers (MS), marine migrants (MM), estuarine (ES). Attributes that had the boundaries of scoring defined by quantile (*) and k-means (**) methods (for more details see section *Defining boundaries*).

Attributes	Ranking			Sources
	Low (1)	Moderate (2)	High (3)	
FOA	Rare and less abundant	Frequent and less abundant	Frequent and Higher Abundant	Present study
OA	(ES + PE or DE) (MS or MM + RE)	(PE + MM or MS)	(DE + MM or MS)	Present study
F/M	< 0.5	0.5 – 1	> 1	(1)
SPR	> 0.4	0.2 – 0.4	< 0.2	(1)
MTI*	> -0.005*	(-0.022) – (-0.005)*	< -0.022*	Present study
	> -0.014**	(-0.014) – (-0.036)**	< -0.036**	
% > L50	> 0.6*	0.198 – 0.6*	< 0.198*	(2)
	> 0.687**	0.039 – 0.687**	< 0.039**	

(1) Patrick *et al.* (2010); (2) Lucena-Frédou et al. (2017)

Measuring uncertainties

In this study we evaluated the effect of subjectivities that could lead to uncertainties in the results, considering the following aspects: (a) definition of the boundaries of the scores (as previous described); (b) assessing the potential redundancy between attributes and (c) attributing random weights.

Weights from 0 to 3 were set for each attribute (default weight of 2) (Stobutzki *et al.*, 2002; Hobday *et al.*, 2007; Lucena-Frédou *et al.*, 2017). A baseline scenario was set up based on Lucena-Frédou *et al.* (2017): weight 3 was assigned to the productivity attributes L_{max} and k (which are decisive to in explaining productivity), and (r) (a key to resilience of the species), while a default weight of 2 was given to all other productivity and susceptibility attributes.

Assessing the potential redundancy between attributes

Additionally, to avoid potential redundancy of some of the PSA attributes (Duffy and Griffiths, 2019), we evaluated relationships between pairs of productivity attributes using a scatterplot matrix and linear regressions. Some redundancies had already been indicated by Lucena-Frédou *et al.* (2016), concerning the parameters L_{50} and L_{max} with k. The correlations between TL, intrinsic growth rate (r) and the other attributes had not been previously evaluated and were investigated in this study. We found weak linear correlations of TL and r with the majority of the productivity attributes (R-Squared – R^2 value less than 0.25; $p < 0.05$), indicating that these attributes can be retained in estimates of vulnerability scores (Supplementary Figure S4). The exception was the positive correlation between r and k ($R^2 = 0.89$; $p < 0.05$) (Supplementary Figure S4). Hence, we tested the removal of attributes with a significant level of correlation both in this study and in Lucena-Frédou *et al.* (2017), zeroing their weight to explore the redundancy effect. However, no changes in the scores or, consequentially, the vulnerability categories were observed so we decided to retain all productivity attributes in the analysis.

Attributing random weights

Weight assignment is subjective. Hence, from the baseline scenario, a total of 10,000 simulations were performed, assigning a random sample of integer weights between 1 and 3 to all productivity and susceptibility attributes to evaluate the sensitivity of the vulnerability scores and ranks with the different weights. Standard deviations of the vulnerability values and the empirical probabilities of being classified as low, medium or highly vulnerable were calculated for each species.

All analyses were performed using the R environment (Core Team, 2020), with packages *vegan* (Oksanen *et al.*, 2017), *cluster* (Maechler *et al.*, 2019), *NbClust* (Charrad *et al.*, 2014), *ggplot2* (Wickham, 2009) and *gplots* (Warnes *et al.*, 2016).

Results

Vulnerability index

Considering the quantile method to define the boundaries of the attribute, all target species of the bottom trawl were considered as being at moderate risk (Table 3). Twenty-three species were classified as being at high risk ($v > 1.72$), with the top ten all being non-target species: (*Bagre marinus, Pseudobatos percellens, Micropogonias furnieri, Menticirrhus americanus*, *Hypanus guttatus, Bagre Bagre*, *Macrodon ancylodon, Rhizoprionodon porosus, Polydactylus virginicus, Cynoscion virescens*), while the majority (44 species) were categorized as being at moderate risk and 22 as being at low risk ($v < 1.15$) (Table 3, Figure 2a). Considering the k-means method, two of the target species (*P. subtilis* and *X. kroyeri*) were considered as being at high risk, while *P. schmitti* was assigned as moderate (Table 3), showing a mean vulnerability score similar to several bycatch species. Similarly, 23 species were classified as high risk ($v > 1.60$). Eight among the top ten of these (excluding *Paralonchurus brasiliensis* and *Larimus breviceps*) were the same as for the quantile method, forty-four as moderate risk and twenty-two as low risk ($v < 1.15$) (Table 3, Figure 2b).

Table 3. Productivity, susceptibility and vulnerability scores (v) defined by quantile and k-means methods (for more details see section *Defining boundaries*), rank and risk rating of the target and non-target species by caught by bottom trawl fishing in BSIR, south of Pernambuco, Northeast Brazil. Vulnerability risk (quantile method): High (H) v > 1.72; Moderate (M) 1.72 < v > 1.15; Low (L) v < 1.15. Vulnerability risk (k-means method): High (H) v > 1.60; Moderate (M) 1.60 < v > 0.85; Low (L) v < 0.85. IUCN ratings: Critically Endangered (CR), Endangered (EN), Vulnerable (VU), Near Threatened (NT), Least Concern (LC), Data Deficient (DD). Families: Achiridae (ACH), Albulidae (ALB), Ariidae (ARI), Atherinopsidae (ATH), Carangidae (CAR), Carcharhinidae (CARC), Clupeidae (CLU), Cynoglossidae (CYN), Dactylopteridae (DAC), Dasyatidae (DAS), Echeneidae (ECH), Engraulidae (ENG), Ephippidae (EPH), Gerreidae (GER), Haemulidae (HAE), Hemiramphidae (HEM), Lutjanidae (LUT), Mullidae (MUL), Ophichthidae (OPH), Ophidiidae (OPH), Ostraciidae (OST), Paralichthyidae (PAR), Pempheridae (PEM), Peneidae (PEN), Polynemidae (POL), Pristigasteridae (PRI), Rhinobatidae (RHI), Sciaenidae (SCI), Serranidae (SER), Sphyraenidae (SPH), Stromateidae (STR), Tetraodontidae (TET), Trichiuridae (TRIC), Triglidae (TRI), Urotrygonidae (URO).

Quantile method									K-means method								
					Vulnerability									Vulnerability			
Family	Species	Code	P	S	Score	Rank	Risk	IUCN	Family	Species	Code	P	S	Score	Rank	Risk	IUCN
ARI	*Bagre marinus*	bag.mar	1.42	2.60	2.24	1	high	DD	POL	*Polydactylus virginicus*	pol.vir	2.47	3.00	2.06	1	high	LC
RHI	*Pseudobatos percellens*	pse.per	1.00	2.00	2.23	2	high	DD	CARC	*Rhizoprionodon porosus*	rhi.por	1.00	1.00	2.00	2	high	DD
SCI	*Micropogonias furnieri*	mic.fur	1.42	2.50	2.17	3	high	LC	SCI	*Micropogonias furnieri*	mic.fur	1.68	2.50	1.99	3	high	LC
SCI	*Menticirrhus americanus*	men.ame	1.63	2.60	2.10	4	high	DD	DAS	*Hypanus guttatus*	hyp.gut	1.31	2.00	1.95	4	high	LC
DAS	*Hypanus guttatus*	hyp.gut	1.21	2.00	2.05	5	high	LC	ARI	*Bagre marinus*	bag.mar	1.89	2.60	1.94	5	high	DD
ARI	*Bagre bagre*	bag.bag	1.47	2.33	2.02	6	high	NT	RHI	*Pseudobatos percellens*	pse.per	1.36	2.00	1.91	6	high	DD
SCI	*Macrodon ancylodon*	mac.anc	2.00	2.75	2.01	7	high	LC	SCI	*Cynoscion virescens*	cyn.vir	1.63	2.33	1.91	7	high	LC
CARC	*Rhizoprionodon porosus*	rhi.por	1.00	1.00	2.00	8	high	DD	SCI	*Macrodon ancylodon*	mac.anc	2.31	2.75	1.87	8	high	LC
POL	*Polydactylus virginicus*	pol.vir	2.10	2.75	1.96	9	high	LC	SCI	*Paralonchurus brasiliensis*	par.bra	2.68	2.83	1.86	9	high	LC
SCI	*Cynoscion virescens*	cyn.vir	1.31	2.00	1.95	10	high	LC	SCI	*Larimus breviceps*	lar.bre	2.57	2.83	1.84	10	high	LC
PAR	*Paralichthys brasiliensis*	para.bra	1.31	2.00	1.95	11	high	LC	PEN	*Penaeus subtilis*	pen.sub	2.78	2.83	1.84	11	high	LC
TRI	*Prionotus punctatus*	pri.pun	1.31	2.00	1.95	12	high	LC	ACH	*Trinectes paulistanus*	tri.pau	2.21	2.60	1.78	12	high	LC
GER	*Diapterus rhombeus*	dia.rho	1.89	2.50	1.86	13	high	LC	TRIC	*Trichiurus lepturus*	tri.lep	2.05	2.50	1.77	13	high	LC
SCI	*Larimus breviceps*	lar.bre	2.10	2.60	1.83	14	high	LC	SCI	*Stellifer rastrifer*	ste.ras	2.42	2.66	1.76	14	high	LC
ALB	*Albula nemoptera*	alb.nem	1.47	2.00	1.82	15	high	LC	SCI	*Menticirrhus americanus*	men.ame	2.26	2.60	1.76	15	high	DD
TRIC	*Trichiurus lepturus*	tri.lep	2.00	2.50	1.80	16	high	LC	PRI	*Pellona harroweri*	pel.har	2.47	2.66	1.74	16	high	LC
SCI	*Paralonchurus brasiliensis*	par.bra	2.31	2.66	1.80	17	high	LC	GER	*Diapterus rhombeus*	dia.rho	2.10	2.50	1.74	17	high	LC
DAC	*Dactylopterus volitans*	dac.vol	1.21	1.00	1.78	18	high	LC	TRI	*Prionotus punctatus*	pri.pun	1.57	2.00	1.73	18	high	LC
ARI	*Aspistor luniscutis*	asp.lun	1.52	2.00	1.78	19	high	LC	CAR	*Caranx hippos*	car.hip	1.31	1.00	1.68	19	high	LC
ARI	*Aspistor quadriscutis*	asp.qua	1.52	2.00	1.78	20	high	LC	PEN	*Xiphopenaeus kroyeri*	xip.kro	2.78	2.66	1.68	20	high	DD
SCI	*Stellifer rastrifer*	ste.ras	2.05	2.50	1.77	21	high	LC	ARI	*Bagre bagre*	bag.bag	2.00	2.33	1.66	21	high	NT
HAE	*Conodon nobilis*	con.nob	2.10	2.50	1.74	22	high	LC	SCI	*Stellifer microps*	ste.mic	2.31	2.50	1.64	22	high	LC
CAR	*Selene brownii*	sel.bro	1.57	2.00	1.73	23	high	LC	ARI	*Aspistor luniscutis*	asp.lun	1.73	2.00	1.61	23	high	LC
PRI	*Odontognathus mucronatus*	odo.muc	2.21	2.50	1.69	24	moderate	LC	PRI	*Odontognathus mucronatus*	odo.muc	2.47	2.50	1.59	24	moderate	LC
CAR	*Selene vomer*	sel.vom	1.63	2.00	1.69	25	moderate	LC	HAE	*Conodon nobilis*	con.nob	2.57	2.50	1.55	25	moderate	LC
SCI	*Umbrina coroides*	umb.cor	1.63	2.00	1.69	26	moderate	LC	HAE	*Haemulopsis corvinaeformis*	hae.cor	2.57	2.50	1.55	26	moderate	DD
CAR	*Caranx hippos*	car.hip	1.31	1.00	1.68	27	moderate	LC	SCI	*Ophioscion punctatissimus*	oph.pun	2.57	2.50	1.55	27	moderate	DD
OPH	*Myrichthys ocellatus*	myr.oce	1.31	1.00	1.68	28	moderate	LC	SCI	*Isopisthus parvipinnis*	iso.par	2.36	2.40	1.53	28	moderate	LC
PEN	*Penaeus subtilis*	pen.sub	2.78	2.66	1.68	29	moderate	LC	PEN	*Penaeus schmitti*	pen.sch	2.78	2.50	1.51	29	moderate	DD
PRI	*Pellona harroweri*	pel.har	2.78	2.66	1.68	30	moderate	LC	SPH	*Sphyraena guachancho*	sph.gua	1.52	1.33	1.51	30	moderate	DD
SCI	*Nebris microps*	neb.mic	1.89	2.25	1.66	31	moderate	LC	CAR	*Selene brownii*	sel.bro	1.94	2.00	1.45	31	moderate	LC
TET	*Lagocephalus laevigatus*	lag.lae	1.42	1.50	1.65	32	moderate	LC	SCI	*Nebris microps*	neb.mic	2.26	2.25	1.45	32	moderate	LC
SCI	*Stellifer microps*	ste.mic	2.05	2.33	1.63	33	moderate	LC	ENG	*Cetengraulis edentulus*	cet.ede	2.47	2.33	1.43	33	moderate	LC
SPH	*Sphyraena guachancho*	sph.gua	1.42	1.33	1.61	34	moderate	DD	ARI	*Aspistor quadriscutis*	asp.qua	2.00	2.00	1.41	34	moderate	LC
SCI	*Ophioscion punctatissimus*	oph.pun	2.47	2.50	1.59	35	moderate	DD	SCI	*Umbrina coroides*	umb.cor	2.00	2.00	1.41	35	moderate	LC
CAR	*Carangoides bartholomaei*	car.bar	1.42	1.00	1.57	36	moderate	LC	PRI	*Chirocentrodon bleekerianus*	chi.ble	2.68	2.33	1.37	36	moderate	LC
ECH	*Echeneis naucrates*	ech.nau	1.42	1.00	1.57	37	moderate	LC	CYN	*Symphurus tessellatus*	sym.tes	2.47	2.25	1.35	37	moderate	LC
LUT	*Lutjanus analis*	lut.ana	1.42	1.00	1.57	38	moderate	NT	ALB	*Albula nemoptera*	alb.nem	2.10	2.00	1.34	38	moderate	LC
SCI	*Isopisthus parvipinnis*	iso.par	2.00	2.20	1.56	39	moderate	LC	PAR	*Paralichthys brasiliensis*	para.bra	2.10	2.00	1.34	39	moderate	LC

ACH	*Trinectes paulistanus*	tri.pau	2.36	2.40	1.53	40	moderate	LC
PEN	*Penaeus schmitti*	pen.sch	2.68	2.50	1.53	41	moderate	DD
HAE	*Haemulon steindachneri*	hae.ste	1.47	1.00	1.52	42	moderate	LC
ARI	*Sciades herzbergii*	sci.her	1.47	1.00	1.52	43	moderate	LC
PEN	*Xiphopenaeus kroyeri*	xip.kro	2.78	2.50	1.51	44	moderate	DD
OPH	*Lepophidium brevibarbe*	lep.bre	1.89	2.00	1.49	45	moderate	DD
MUL	*Upeneus parvus*	upe.par	1.89	2.00	1.49	46	moderate	LC
HAE	*Haemulon plumierii*	hae.plu	1.52	1.00	1.47	47	moderate	LC
LUT	*Lutjanus synagris*	lut.syn	1.52	1.00	1.47	48	moderate	NT
CAR	*Chloroscombrus chrysurus*	chl.chr	1.63	1.50	1.45	49	moderate	LC
SCI	*Menticirrhus littoralis*	men.lit	1.94	2.00	1.45	50	moderate	DD
HAE	*Haemulopsis corvinaeformis*	hae.cor	2.47	2.33	1.43	51	moderate	DD
URO	*Urotrygon microphthalmum*	uro.mic	2.00	2.00	1.41	52	moderate	DD
GER	*Eucinostomus gula*	euc.gul	1.89	1.80	1.36	53	moderate	LC
CYN	*Symphurus plagusia*	sym.pla	2.10	2.00	1.34	54	moderate	LC
PRI	*Chirocentrodon bleekerianus*	chi.ble	3.00	2.33	1.33	55	moderate	LC
OST	*Acanthostracion polygonius*	aca.pol	1.68	1.00	1.31	56	moderate	LC
GER	*Diapterus auratus*	dia.aur	2.15	2.00	1.30	57	moderate	LC
CYN	*Symphurus tessellatus*	sym.tes	2.68	2.25	1.28	58	moderate	LC
SCI	*Bairdiella ronchus*	bai.ron	2.21	2.00	1.27	59	moderate	LC
ENG	*Lycengraulis grossidens*	lyc.gro	2.26	2.00	1.24	60	moderate	LC
PAR	*Cyclopsetta chittendeni*	cyc.chi	2.316	2.00	1.21	61	moderate	LC
ARI	*Cathorops spixii*	cat.spi	1.78	1.00	1.21	62	moderate	LC
HAE	*Haemulon aurolineatum*	hae.aur	1.78	1.00	1.21	63	moderate	LC
ENG	*Cetengraulis edentulus*	cet.ede	2.78	2.16	1.18	64	moderate	LC
SCI	*Stellifer brasiliensis*	ste.bra	2.78	2.16	1.18	65	moderate	LC
HAE	*Anisotremus moricandi*	ani.mor	2.36	2.00	1.18	66	moderate	LC
EPH	*Chaetodipterus faber*	cha.fab	1.84	1.00	1.15	67	moderate	LC
ENG	*Anchoa spinifer*	anc.spi	2.42	2.00	1.15	68	low	LC
PAR	*Citharichthys spilopterus*	cit.spi	2.68	2.00	1.04	69	low	LC
PAR	*Etropus crossotus*	etr.cro	2.68	2.00	1.04	70	low	LC
PAR	*Citharichthys macrops*	cit.mac	2.78	2.00	1.02	71	low	LC
HAE	*Genyatremus luteus*	gen.lut	2.00	1.00	1.00	72	low	LC
OPH	*Ogcocephalus vespertilio*	ogc.ves	2.00	1.00	1.00	73	low	LC
STR	*Peprilus paru*	pep.par	2.21	1.50	0.93	74	low	LC
CLU	*Harengula clupeola*	har.clu	2.36	1.66	0.91	75	low	LC
ACH	*Achirus lineatus*	ach.lin	2.10	1.00	0.89	76	low	LC
ACH	*Achirus declivis*	ach.dec	2.68	1.60	0.67	77	low	LC
ENG	*Anchoviella lepidentostole*	anc.lep	2.68	1.50	0.59	78	low	LC
HEM	*Hyporhamphus unifasciatus*	hyp.uni	2.42	1.00	0.57	79	low	NT
ENG	*Anchoa januaria*	ach.jan	2.78	1.50	0.54	80	low	LC
ENG	*Anchoa tricolor*	anc.tri	2.78	1.50	0.54	81	low	LC
SCI	*Stellifer stellifer*	ste.ste	2.78	1.50	0.54	82	low	LC
TET	*Sphoeroides greeleyi*	sph.gre	3.00	1.50	0.50	83	low	LC
SER	*Diplectrum formosum*	dip.for	2.52	1.00	0.47	84	low	LC
CLU	*Opisthonema oglinum*	opi.ogl	2.73	1.33	0.42	85	low	LC
ATH	*Atherinella brasiliensis*	ath.bra	2.68	1.00	0.31	86	low	LC
PEM	*Pempheris schomburgkii*	pem.sch	2.68	1.00	0.31	87	low	LC
CLU	*Rhinosardinia bahiensis*	rhi.bah	2.68	1.00	0.31	88	low	LC
TET	*Sphoeroides testudineus*	sph.tes	2.84	1.00	0.15	89	low	LC
GER	*Eucinostomus argenteus*	euc.arg	2.89	1.00	0.10	90	low	LC

CAR	*Selene vomer*	sel.vom	2.10	2.00	1.34	40	moderate	LC
URO	*Urotrygon microphthalmum*	uro.mic	2.21	2.00	1.27	41	moderate	DD
SCI	*Stellifer brasiliensis*	ste.bra	2.57	2.16	1.24	42	moderate	LC
SCI	*Menticirrhus littoralis*	men.lit	2.31	2.00	1.21	43	moderate	DD
LUT	*Lutjanus analis*	lut.ana	1.78	1.00	1.21	44	moderate	NT
PAR	*Citharichthys spilopterus*	cit.spi	2.36	2.00	1.18	45	moderate	LC
PAR	*Cyclopsetta chittendeni*	cyc.chi	2.36	2.00	1.18	46	moderate	LC
OPH	*Lepophidium brevibarbe*	lep.bre	2.36	2.00	1.18	47	moderate	DD
CYN	*Symphurus plagusia*	sym.pla	2.36	2.00	1.18	48	moderate	LC
MUL	*Upeneus parvus*	upe.par	2.36	2.00	1.18	49	moderate	LC
DAC	*Dactylopterus volitans*	dac.vol	1.84	1.00	1.15	50	moderate	LC
ENG	*Anchoa spinifer*	anc.spi	2.47	2.00	1.13	51	moderate	LC
HAE	*Anisotremus moricandi*	ani.mor	2.47	2.00	1.13	52	moderate	LC
PAR	*Citharichthys macrops*	cit.mac	2.47	2.00	1.13	53	moderate	LC
GER	*Diapterus auratus*	dia.aur	2.52	2.00	1.10	54	moderate	LC
HAE	*Haemulon plumierii*	hae.plu	1.89	1.000	1.10	55	moderate	LC
LUT	*Lutjanus synagris*	lut.syn	1.89	1.00	1.10	56	moderate	NT
SCI	*Bairdiella ronchus*	bai.ron	2.57	2.00	1.08	57	moderate	LC
ECH	*Echeneis naucrates*	ech.nau	1.94	1.00	1.05	58	moderate	LC
HAE	*Haemulon steindachneri*	hae.ste	1.94	1.00	1.05	59	moderate	LC
ENG	*Lycengraulis grossidens*	lyc.gro	2.68	2.00	1.04	60	moderate	LC
TET	*Lagocephalus laevigatus*	lag.lae	2.10	1.50	1.02	61	moderate	LC
PAR	*Etropus crossotus*	etr.cro	2.78	2.00	1.02	62	moderate	LC
ARI	*Sciades herzbergii*	sci.her	2.00	1.00	1.00	63	moderate	LC
CAR	*Chloroscombrus chrysurus*	chl.chr	2.15	1.50	0.97	64	moderate	LC
EPH	*Chaetodipterus faber*	cha.fab	2.05	1.00	0.94	65	moderate	LC
CAR	*Carangoides bartholomaei*	car.bar	2.10	1.00	0.89	66	moderate	LC
ACH	*Achirus declivis*	ach.dec	2.36	1.60	0.87	67	moderate	LC
CLU	*Harengula clupeola*	har.clu	2.47	1.66	0.84	68	low	LC
SCI	*Stellifer stellifer*	ste.ste	2.47	1.66	0.84	69	low	LC
ARI	*Cathorops spixii*	cat.spi	2.15	1.00	0.84	70	low	LC
HAE	*Haemulon aurolineatum*	hae.aur	2.15	1.00	0.84	71	low	LC
GER	*Eucinostomus gula*	euc.gul	2.42	1.60	0.83	72	low	LC
OST	*Acanthostracion polygonius*	aca.pol	2.21	1.00	0.78	73	low	LC
OPH	*Myrichthys ocellatus*	myr.oce	2.21	1.00	0.78	74	low	LC
ENG	*Anchoa januaria*	ach.jan	2.47	1.50	0.72	75	low	LC
ENG	*Anchoviella lepidentostole*	anc.lep	2.47	1.50	0.72	76	low	LC
STR	*Peprilus paru*	pep.par	2.57	1.50	0.65	77	low	LC
ATH	*Atherinella brasiliensis*	ath.bra	2.36	1.00	0.63	78	low	LC
HAE	*Genyatremus luteus*	gen.lut	2.36	1.00	0.63	79	low	LC
OPH	*Ogcocephalus vespertilio*	ogc.ves	2.36	1.00	0.63	80	low	LC
TET	*Sphoeroides greeleyi*	sph.gre	2.68	1.50	0.59	81	low	LC
ENG	*Anchoa tricolor*	anc.tri	2.78	1.50	0.54	82	low	LC
ACH	*Achirus lineatus*	ach.lin	2.47	1.00	0.52	83	low	LC
SER	*Diplectrum formosum*	dip.for	2.47	1.00	0.52	84	low	LC
PEM	*Pempheris schomburgkii*	pem.sch	2.47	1.00	0.52	85	low	LC
CLU	*Rhinosardinia bahiensis*	rhi.bah	2.47	1.00	0.52	86	low	LC
CLU	*Opisthonema oglinum*	opi.ogl	3.00	1.33	0.33	87	low	LC
GER	*Eucinostomus argenteus*	euc.arg	2.68	1.00	0.31	88	low	LC
TET	*Sphoeroides testudineus*	sph.tes	2.68	1.00	0.31	89	low	LC
HEM	*Hyporhamphus unifasciatus*	hyp.uni	2.78	1.00	0.21	90	low	NT

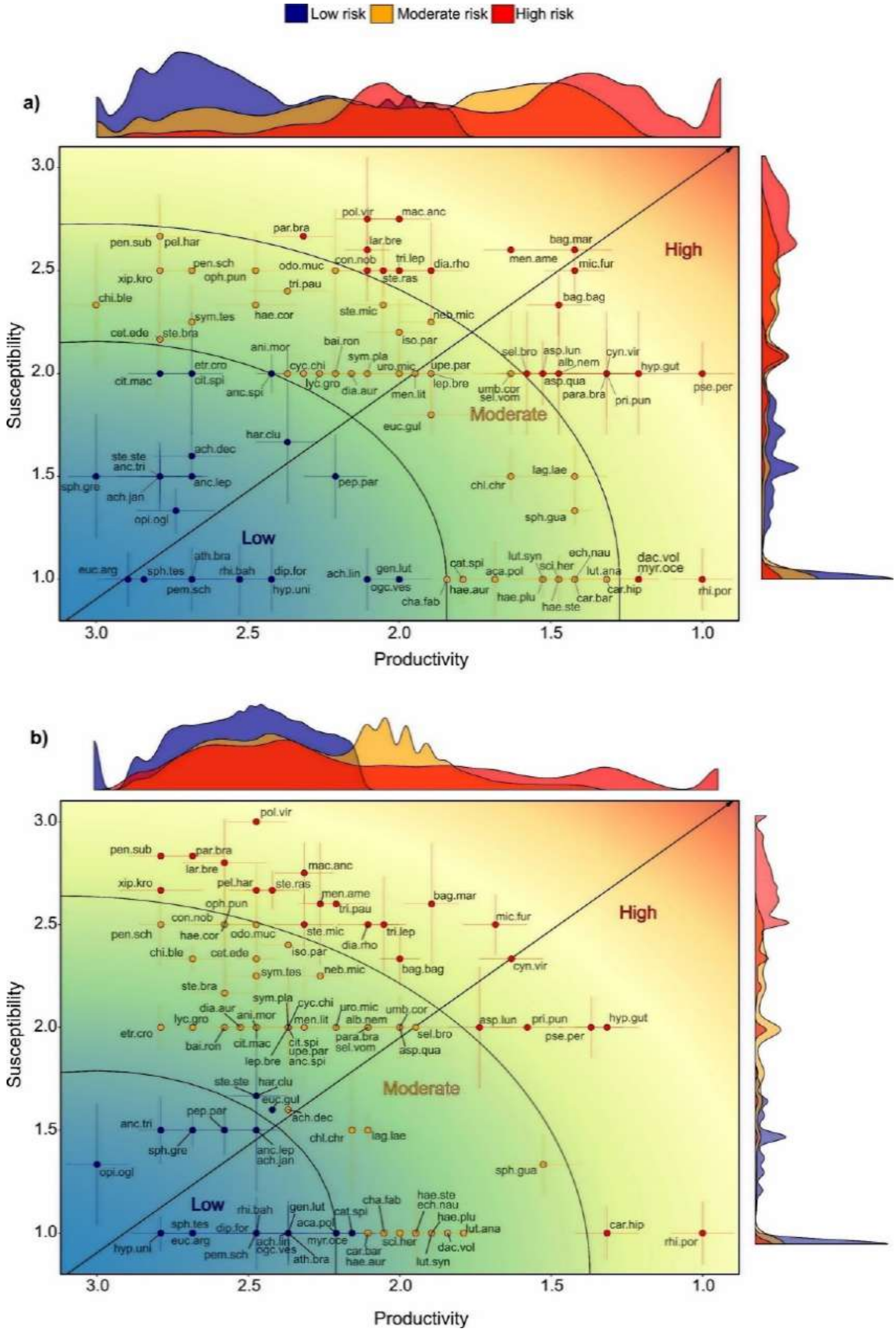

Figure 2. Scores of productivity (P), susceptibility (S) and vulnerability (v) of species caught by bottom trawl fishing in Barra of Sirinhaém (BSIR), south of Pernambuco, Northeast Brazil estimated by quantile (a) and k-means (b) methods (Species codes are given in Table 3). The colour scale represents the lowest v (blue) and highest v (red) values. The range lines for each point show the standard deviation obtained from uncertainty simulations (10,000 runs). The density plots represents the total variation of the P and S scores, for each risk category (a) quantile (High v > 1.72; Moderate 1.72 > v > 1.15; Low v < 1.15) and (b) k-means (High v > 1.60; Moderate 1.60 > v > 0.85; Low v < 0.85).

Assessing uncertainties

In general, most species (76%; 68 species) did not change their risk category (low, moderate or high) according to the methods used to define of the boundaries of the attribute scores (Figure 3). From these, seventeen species of high vulnerability were always classified as high risk (e.g. *Bagre marinus*, *Hyphanus guttatus*, *Macrodon ancylodon*, *Larimus breviceps*), thirty-three as moderate (e.g. *P. schmitti*, *Odontognathus mucronatus*, *Haemulopsis corvinaeformis*, *Isopisthus parvipinnis*) and eighteen as low (e.g. *Atherinella brasiliensis*, *Harengula clupeola*, *Hyporhamphus unifasciatus*, *Opisthonema oglinum*). However, given the changes in productivity and susceptibility attribute values (Supplementary Figure S5), for 22 species (24%) a decrease in risk status was found (Figure 3), between high and moderate or moderate and low risk categories. Six species (e.g. *Albula nemoptera*, *Dactylopterus volitans*, *Paralichthys brasiliensis*) changed from high (quantile method) to moderate risk (k-means method) and five (e.g. *Acanthostracion polygonius*, *Haemulon aurolineatum*, *Myrichthys ocellatus*) from moderate (quantile) to low risk (k-means) (Table 3 and Figure 3). The risk status also increased for 11 species, six from moderate (quantile) to high (k-means) (e.g. *P. subtilis*, *X. kroyeri*), and five from low (quantile) to moderate risk categories (e.g. *Anchoa spinifer, Etropus crossotus*, *Citharichthys spilopterus*) (Table 3 and Figure 3).

For 94% of the species, the position in the vulnerability ranking changed, but within same risk category, such as the *B. marinus* (High risk; rank: 1 on quantile and 5 on k-means), *Chirocentrodon bleekerianus* (Moderate risk; rank 56 on quantile, rank 36 on k-means) and *Stellifer stellifer* (Low risk; rank 82 on quantile, rank 69 on k-means) (Figure 3).

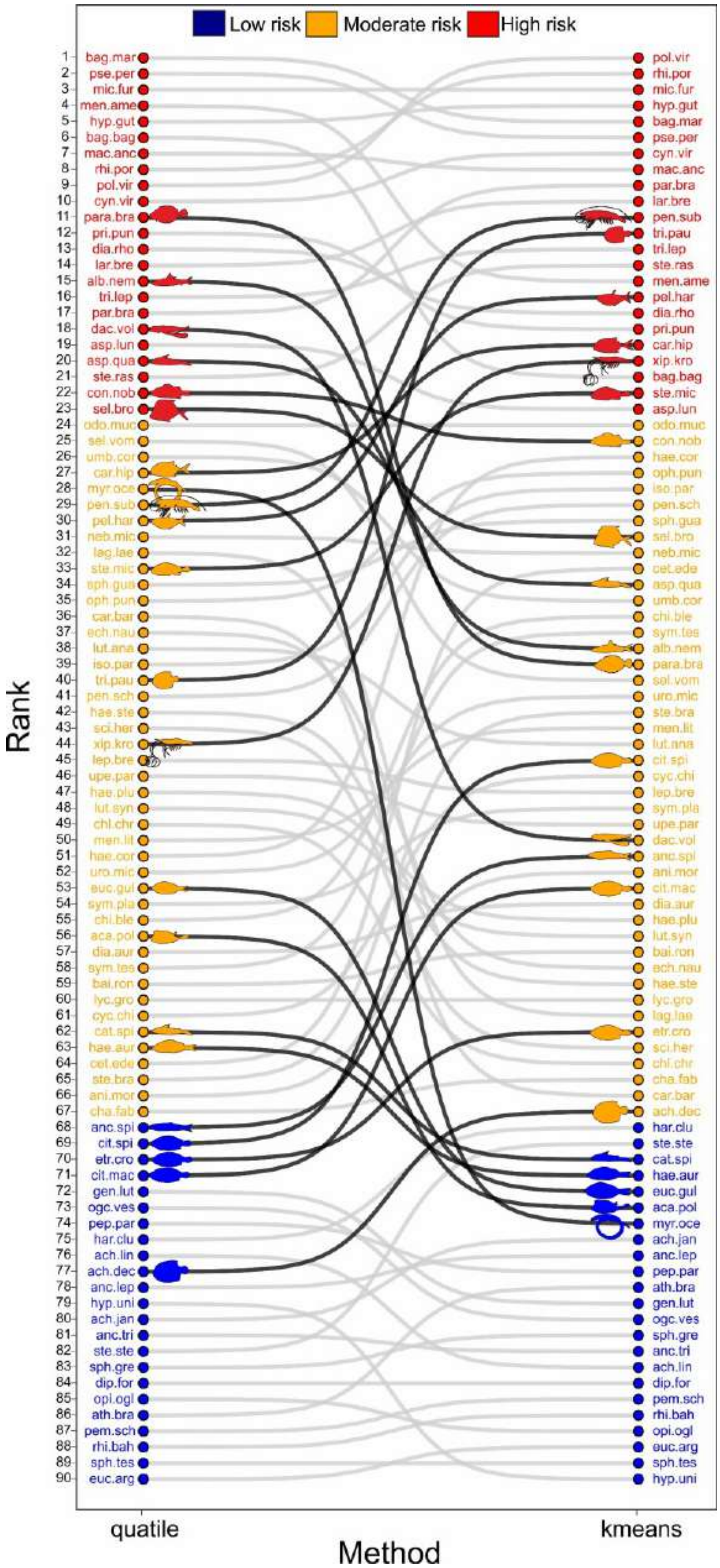

Figure 3. Difference in rank and risk categories of target and non-target species caught by bottom trawl fishing in Barra of Sirinhaém (BSIR) south of Pernambuco, Northeast Brazil. The lines show changes in rank between the methods (quantile and k-means) to define the boundaries of attribute scores. Black lines indicate that the species changed risk category and grey lines indicate that they did not. Species codes are given in Table 3.

Regardless of the weight assignments, including zeroing redundant attributes of productivity and susceptibility, most species did not show alterations in their classification of risk (Figure 2 and 3). For both methods (quantile and k-means), the top twelve species at risk, including *B. marinus, P. percellens, M. furnieri, M. americanus, H. guttatus, M. ancylodon, R. porosus, P. virginicus, C. virescens*, *L. breviceps*, *B. bagre* and *P. brasiliensis* (Table 3), had a probability larger than 0.8 of being classified as at high risk (Figure 4a and 4b). Conversely, sardines, (e.g. *H. clupeola, O. oglinum, Anchoa tricolor, Rhinosardinia bahiensis*), estuarine fishes (e.g. *S. greeleyi, H. unifasciatus, A. brasiliensis*) and reef fishes (e.g. *Diplectrum formosum, Haemulon aurolineatum*) had a high probability (> 0.6) of being at low risk from bottom trawling fishing (Figure 4a and 4b).

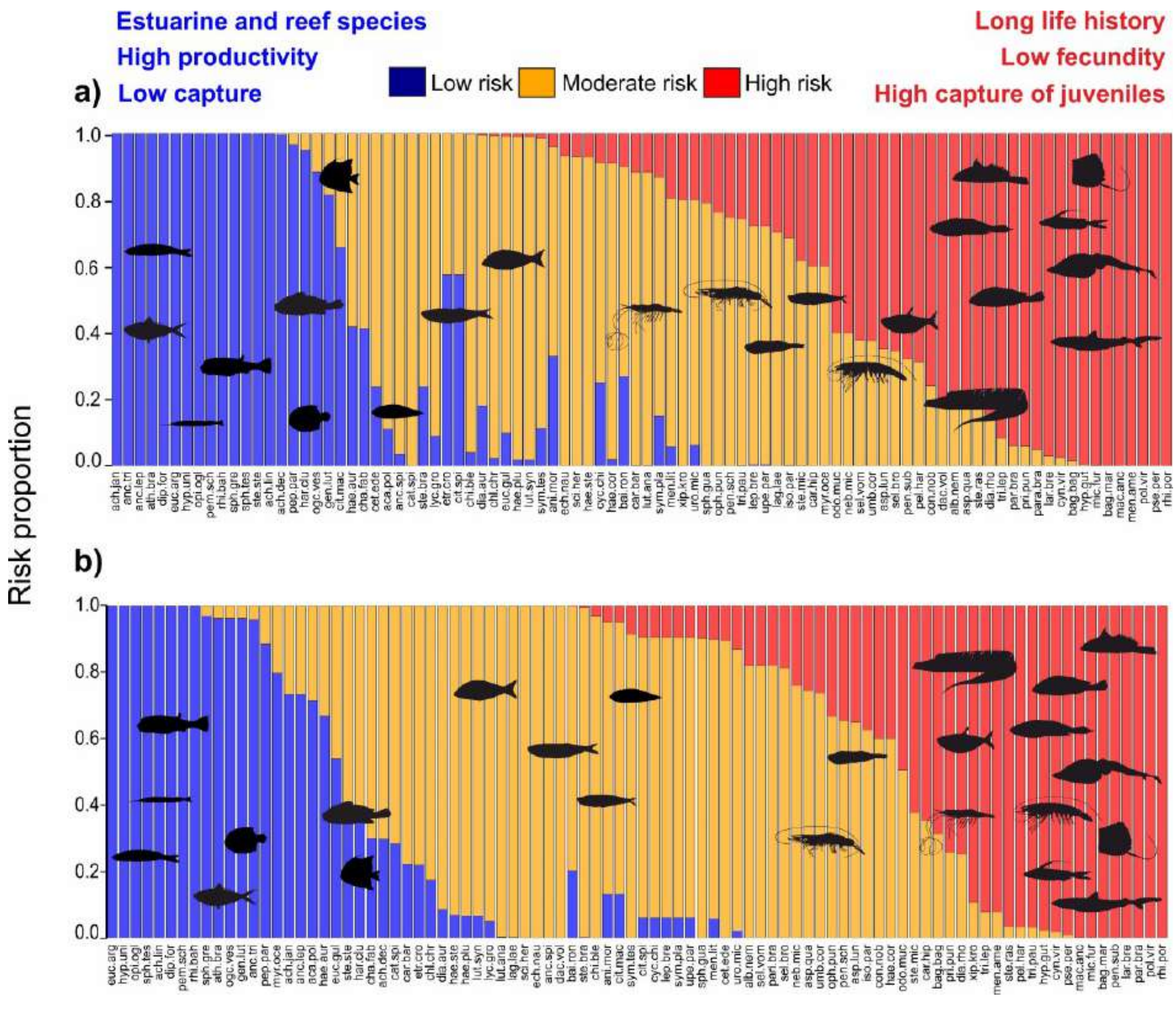

Figure 4. Probability of risk from uncertainty simulations by the methods: a) quantile and b) k-means for each species caught (species codes are given in Table 3) by bottom trawl fishing in Barra of Sirinhaém (BSIR), south of Pernambuco, Northeast Brazil. Species are ordered (left to right) according to vulnerability rank: low (blue), moderate (yellow) and high (red).

Discussion

Although the Productivity and Susceptibility Analysis approach does not provide traditional fishery management reference points (Fujita *et al.*, 2014), it allows policy makers and stakeholders to focus on monitoring, assessment and management of the stocks and species shown to be at the highest risk from fishing (Hobday *et al.*, 2011). PSA is particularly useful in data-poor cases, where the catches or biological data (e.g. biomass and size) are not comprehensive, are aggregated across species or are insufficient to run a quantitative stock assessment (Lucena-Frédou *et al.*, 2017), as is the case in many tropical multispecies fisheries including small-scale Brazilian fisheries. Given the lack of stock assessment analysis in these cases, particularly in shrimp fisheries, fishery regulations currently available are restricted to target species and do not take into account non-target species or the ecosystem as a whole (Gillett, 2008; Santos, 2010; Dias-Neto, 2011). The PSA approach has been gaining strength in defining fishery regulation, such as for regional fisheries management organizations like the Northwest Atlantic Fisheries Organization, NAFO; International Commission for the Conservation of Atlantic Tunas, ICCAT; Western and Central Pacific Fisheries Commission, WCPFC; Commission for the Conservation of Southern Bluefin Tuna, CCSBT and expert groups of the International Council for the Exploration of the Sea, ICES. However, for small-scale fisheries, that usually have a data-limited status, this approach has rarely been used.

The region and fishery in our case study has no monitoring data enough. Thus, quantitative assessments of the stocks and how much they are affected by fishing are not available and data-limited analysis approaches, including PSA, are highly recommended. However, given its nature, PSA should be used with caution, its results applied prudently, and a comparation with other assessment approaches strongly recommended (Osio *et al.*, 2015). For example, Zhou *et al.* (2016), comparing stock assessments in Australia using Ecological Risk Assessment tools, found that half of the species classified as high risk by PSA were also considered as overfished by stock assessment models. Lucena-Frédou *et al.* (2017), comparing the risk obtained by PSA with the IUCN (International Union of Conservation of Nature) extinction risk categories and the status of stock as assessed by the RFMOs (Regional Fisheries Management Organizations), reported that vulnerability ranks were comparable, and several species at high risk were overfished and/or subjected to overfishing, as well as being in IUCN extinction risk categories (CR, Critically Endangered; EN, Endangered; or VU, Vulnerable). The results presented here should, therefore, be considered with some caution and may refer, either for the target or non-target species, to one specific part of the population exploited by small-scale shrimp trawling in Sirinhaém, Northeast Brazil. We believe, however, that the method is important in highlighting the species that should be prioritized, either for urgent assessment and/or data collection.

Seventeen among the 90 species caught by bottom trawling in the region were considered exclusively of high vulnerability, independently of the method (quantile and k-means) used to define the boundaries of the attribute scores. Among these, we reported Elasmobranchii (e.g. *H. guttatus P. percellens, R. porosus*) and catfishes (e.g. *B. marinus*, *B. bagre*), which are often discarded or consumed, and hake species (e.g. *M. ancylodon, Cynoscion virescens*) and croaker (*M. furnieri*), which are usually sold. The high vulnerability scores mainly resulted from the combination of very low productivity due to medium to long lifespans (Simpfendorfer *et al.*, 2011; Caltabellotta *et al.*, 2019) and low spawning/potential reproduction (Pinheiro *et al.*, 2006; da Silva *et al.*, 2018) (in the case of Elasmobranchii and catfishes); or very high susceptibility to the bottom trawling due to high capture rates of young individuals (Silva Június *et al.*, 2015) and overlap of feeding and breeding grounds with fishing areas (Silva Június *et al.*, 2019) (in the case of Sciaenidae).

Hake species, croakers, catfishes and elasmobranchs, mainly as adults, are important fishery resources on the Brazilian coastline (MPA, 2011), but the high amount of juveniles captured can negatively affect the recruitment process (Biju Kumar and Deepthi, 2006). Given their life history characteristics (large maximum size, late maturity, slow growth rate and low intrinsic population growth rate), elasmobranchs are less resilient to fishing impacts than other groups (García *et al.*, 2008; Hutchings *et al.*, 2012; Duffy and Griffiths, 2019). Elasmobranch species are often reported as being highly vulnerable to multi-gear fisheries throughout the world, including shrimp trawl fishery, such as in Costa Rica, Eastern Tropical Pacific (Clarke *et al.*, 2018); U.S. coast (Patrick *et al.*, 2010); Gulf of Mexico (Martínez-Candelas *et al.*, 2020) and Australia (Zhou *et al.*, 2011). In south Brazil, the trawl fishery has already contributed to the depletion of some Elasmobranchs and Sciaenidae populations (Vasconcellos and Haimovici, 2006; Barreto *et al.*, 2016; Dias and Perez, 2016; Haimovici and Cardoso, 2017; Mendonça *et al.*, 2020). For the Brazilian sciaenids, Chao *et al.* (2015) identified habitat degradation and high bycatch capture rates as the main threats. Even taking into account the different nature of the trawl fisheries (artisanal in this case study and industrial in south Brazil), we must be careful with these species, which are extremely common in trawling fisheries, included trawls targeting shrimps. Moreover, some of these exploited species are categorized as Data Deficient (DD) (e.g. *Bagre marinus*, *Pseudobatos percellens*, *Rhizoprionodon porosus*) at the regional level according to IUCN Red List criteria, indicating data is inadequate to assess the risk of extinction, recognizing the possibility of being endangered (ICMbio, 2018). *Bagre bagre* was considered as the sixth most vulnerable species (quantile method) and is also classified as Near Threatened (NT) (ICMbio, 2018). In Northeast Brazil, hake species, croakers, catfishes and elasmobranchs do not have adequate stock assessments, or have not been evaluated due to lack of information, although they deserve attention given the history of overexploitation and depletion already reported in the country. Thus, these species must be prioritized in research and formal stock assessment and possibly regulation are urgently required.

Most species (33) were classified, regardless of the method used, as being at moderate risk, but two groups of species were differently affected by trawling. The first, including species of the main bycatch families, Pristigasteridae, Scianidae and Haemulidae (e.g. *H. corvinaeformis, I. parvipinnis, C.*

bleekerianus), have reproduction and feeding sites that largely overlap the fishing area (Silva Júnior *et al.*, 2015; Eduardo *et al.*, 2018a; Lira *et al.*, 2019) and are also consumed by fishermen and local communities. Although most of these species were considered Least Concern (LC) (e.g. *Stellifer rastrifer*, *I. parvipinnis*, *Odontognathus mucronatus*), some were categorized as DD (e.g. *Ophioscion punctatissimus*, *H. corvinaeformis*) (ICMbio, 2018). Moreover, Verba *et al.* (2020) recently classified many of these Sciaenidae and Haeumilidae species as fully or overexploited within the Brazilian Exclusive Economic Zone, in response to synergistic interaction between the warming of the sea, fishery exploitation and specific life-history traits. Our findings, as well as those reported by other authors (Chao *et al.*, 2015) using different approaches, confirm the acceptable level of risk for these species. However, they should be considered a monitoring and research priority in coming years.

Another group, composed of reef-associated and sand bottom fish species (grunts *Haemulon* spp., Jacks *Caranx* spp, snappers *Lutjanus* spp and barracuda *Sphyraena guachancho*), are at moderate risk. They have long lifespans and low growth rates (Lessa *et al.*, 2004; Vasconcelos-Filho *et al.*, 2018). However, they suffer little incidental capture (Silva Júnior *et al.*, 2019) compared with the first group of species, and fishing has a lower overlap with their reproduction zones (Cardoso de Melo *et al.*, 2020). Although these species are not particularly threated by shrimp trawling, they are heavily exploited in Northeast Brazil by multiple gears (Resende *et al.*, 2003; Frédou *et al.*, 2006; Lessa *et al.*, 2009), and some has been already considered as fully or overexploited during the 2000´s (Frédou *et al.*, 2009b) and are classed as NT (Near Threatened) (ICMbio, 2018) (*Lutjanus analis* and *L. synagris*). Particular attention should therefore be paid to the additive effect of the artisanal shrimp fishery, especially because this fishing activity mainly targets juveniles.

Estuarine and pelagic species with high productivity, including sardines, puffer and some flatfishes, were shown to be at low risk (lowest vulnerability scores) from bottom trawling. These species, such as *A. brasiliensis*, *H. clupeola*, and *O. oglinum,* inhabit estuarine areas or are migrating between the estuarine and surf zones, occasionally using the deepest areas of the coastal zone (Félix *et al.*, 2007; Santana *et al.*, 2013; Ferreira *et al.*, 2019). These species have high growth rates and natural mortality ratios, and great spawning potential (Lessa *et al.*, 2004, 2008; Chaves *et al.*, 2017). They are categorized LC at the regional level (ICMbio, 2018), except *H. unifasciatus*, which is considered NT and was evaluated as overexploited during the early 2000s (Lessa *et al.*, 2009).

Considering the target shrimps, all three species were classified as being at moderate risk by the quantile method or at high risk, in the cases of *X. kroyeri* and *P. subtilis*, by k-means method. They were not, however, in the top ten of the vulnerability rankings. In general, *P. subtilis* showed higher vulnerability values and rank. This species spawns in the open sea, with larvae and post-larvae migrating to nursery grounds in estuaries and other wetlands, and juveniles living in shallow zones and migrating to offshore waters when they become adults (Dall *et al.*, 1990; Silva *et al.*, 2016). Hence, in our study fishery, which operates near the coast (Tischer and Santos, 2003), many young individuals are caught (Lopes *et al.*, 2014; Silva *et al.*, 2015, 2018), increasing the susceptibility of the species. Lira *et al.* (2021) reported that *P. subtilis* is more affected by increasing effort than the other two species because it causes significant biomass reductions. However, the current stock status does not indicate overexploitation in the region (Silva *et al.*, 2015).

Xiphopenaeus kroyeri and *P. schmitti* are the main targets of trawl fishing in the region in terms of catch volume and market value, respectively (Santos, 2010). Traditional stock assessments carried out in the region do not indicate overexploitation, which is supported by the species' short life cycle, rapid growth and high natural mortality (Lopes *et al.*, 2014; Silva *et al.*, 2015, 2018). Also, according to Lira *et al.* (2021), these two shrimp species are more resilient to changes in fishing effort. The main factors affecting these species have instead been environmental, in terms of rainfall or primary production, underlining the importance of the climate change effects on these stocks and, therefore, on fishing activity. Both shrimp species were recently classified as DD (ICMbio, 2018) and present evidence of overexploitation on the southern coast of Brazil, with strong decreases in stock biomass and size of individual catches (Fernandes *et al.*, 2011; Almeida *et al.*, 2012; Davanso *et al.*, 2017; Musiello-Fernandes *et al.*, 2018; Carvalho *et al.*, 2021).

Uncertainty measures

The subjective nature of PSA may lessen the reliability of the results and consequently the management measures adopted. In our study, we addressed some of these obstacles, such as the choice of method to select attribute boundaries, the potential redundancy between attributes and the consequence of differential weights applied to productivity and susceptibility attributes. Recently, some studies have addressed the fragilities of PSA. McCully Phillips *et al.* (2015) applied the adapted confidence scores and beta probability distributions for susceptibility attributes, and Brown *et al.* (2015) tested the attribute combinations resulting in standard deviation as measure of the effect of subjectivity on the scores generated for each species. Lucena-Frédou *et al.* (2017) incorporated the standard errors of the parameter with the highest correlation with productivity (intrinsic growth rate-r) and its effect on the estimates of the vulnerability ranks. Duffy and Griffiths (2019) evaluated the impacts of weightings and removal of correlated attributes and, as for our study, did not observe any notable changes of the vulnerability status of the species. A new method to classify the vulnerability outputs into sustainability categories using a Gaussian mixture model (GMM) was applied by Baillargeon *et al.* (2020), who observed a more effectively grouped species with similar productivity and susceptibility scores together.

In our case, we assessed the potential redundancy between pairs of attributes not previously evaluated for other studies, compared two methods (quantile and clustering k-means) to select boundaries of scoring the productivity and susceptibility attributes, and evaluated their impact on the estimates of vulnerability values and the risk rank of the species. Finally, we performed simulations assigning random weights (1 to 3) to the attributes, thus obtaining standard deviations for the productivity, susceptibility and vulnerability scores of each species, which allowed an estimation of the probability of a species being classified as being at low, moderate or high risk.

High correlations between attributes suggest that two or more of them convey similar information, which would lead to overemphasis of their effect. To counter such misleading effects, one of the correlated attributes should be removed. Conversely, low correlations suggest that both attributes should be considered because each of them conveys unique biological information to define the vulnerability of a species (Stobutzki *et al.*, 2001). Removal of one of the correlated attributes did not, however, change the scores or, consequentially, the risk category of the species, hence they were all considered in the analysis. When considering the different methods for defining boundaries, most species changed their vulnerability rank, but did not change their risk category (low, moderate or high). The clustering method has been successfully used in the PSA, mainly to identify similar groupings of species for different factors (Cortés *et al.*, 2010; Cope *et al.*, 2011; Furlong-Estrada *et al.*, 2017). More recently, Altuna-Etxabe *et al.* (2020) applied, for the first time, a criterion for defining the boundaries of attribute scores, but did not evaluate its effects in the estimation of the vulnerability risk of the species. These authors concluded that, due to the narrow range of attribute values for most of the species they studied (from cephalopods to sharks), the k-means method created thresholds that were too coarse and so considered only the quantile method to define attribute boundaries. Although, in our study, no significant differences were observed in the overall PSA results when comparing the two methods, some species changed risk category. For example, the target species *X. kroyeri* and *P. subtilis*, classified as being at moderate risk, changed to the high risk category with the k-mean method. This happened mainly through changes in the vulnerability ranking due to differences in boundaries defined for some attributes in the k-mean method compared with the quantile method, e.g. Percentage of individuals > L_{50} in the catches (% > L_{50}).

Extreme values of the PSA vulnerability score are often well correlated with the risk of overexploitation, while intermediate values have high uncertainty concerning the risk posed by exploitation of the species (Hordyk and Carruthers, 2018). Extreme values may also be related to many false positives or negatives (Hobday *et al.*, 2011; Zhou *et al.*, 2016; Lucena-Frédou *et al.*, 2017) obtained when the attribute scores overestimate or underestimate the level of risk of a species relative to an assessment based on a larger dataset. Hence, the performed simulations were important for two reasons: first, to minimize the uncertainties of the results associated with the attribution of weights, mainly for the species at higher (high vulnerability) and lower (low vulnerability) risk; and second, through a probability estimation, to reinforce the risk status associated with each species.

Finally, PSA summarizes the complex biological and ecosystem processes involved in determining the potential risk to one part of a population exploited or subjected to exploitation by a specific fishery. Nevertheless, it is necessary to consider the regional circumstances, assessing the potential vulnerability of species to the fisheries operating in the area (Hornborg *et al.*, 2020). Attributes and scores should, therefore, be chosen to reflect the specificities of study cases. Recently, new attributes concerning the characteristics of local fisheries have been considered. Lucena-Frédou *et al.* (2017) added new productivity and susceptibility attributes and Martínez-Candelas *et al.* (2020) incorporated the type of fishing vessel and technology into the analysis.

Management support conclusions

The shrimp fishery at Pernambuco is multispecific in nature and is currently unregulated, contradicting the Code of Conduct for Responsible Fisheries (CCRF) that recommends that entire catches should be managed in an ecologically sustainable manner considering the main species involved (target and bycatch) (FAO, 1995). Our findings suggest that some non-target species can be more vulnerable to bottom trawling fishing than the target species in the region, thus underlining that vulnerability of bycatch populations should be taken into account when making management decisions as part of an ecosystem approach.

Catches of elasmobranchs and catfish by bottom trawling are not high in the region (Tischer and Santos, 2003; Silva Júnior *et al.*, 2019). Trawl fishing therefore does not appear to be the main threat to these species. However, given the inherent high vulnerability of elasmobranchs and catfish based on their biological traits, these are high risk species. In contrast, hakes and croakers are economically important and a large proportion of catches are juveniles, which could pose a threat to the sustainability of their stocks and other associated fisheries that capture them as adults. Thus, for these species, we suggest a more effective complementary assessment by quantitative approaches (e.g. traditional stock assessment) to improve understanding of the potential risk, followed by management recommendations in appropriate cases. The use of bycatch reduction devices (e.g. fisheye, grid or square mesh) to exclude juveniles may be a potential solution (Broadhurst, 2000; Eayrs, 2007; Larsen *et al.*, 2017). Although the effects of reducing this bycatch on food security and income generation in the region should be better evaluated. International initiatives have been developed that are important for encouraging effective management of bycatch, such as the FAO project Sustainable Management of Bycatch in Latin America and Caribbean Trawl Fisheries (REBYC-II LAC http://www.fao.org/in-action/rebyc-2/overview/en/), which has four pilot sites along the Brazilian coast (at Pará, Pernambuco, Paraná/Santa Catarina and Rio Grande do Sul).

Less data for potential assessments are available for nearshore species, so evaluating their vulnerability needs to be a key management priority (Patrick *et al.*, 2009). In general, catch rates of the most abundant species of bycatch are high (e.g. *Pellona harroweri*, *Isopisthus parvipinnis*, *Chirocentrodon bleekerianus*) and, considering the particularities of our case study, were classified as being at moderate risk, mainly due to their high resilience. These species deserve priority for research given the lack of information about their population structure and life history traits and considering the beneficial role of the bycatch for the local communities (Carvalho *et al.*, 2020) as well as the potential negative nutritional, economic and social effects in a scenario of declining catches (Lira *et al.*, 2021b).

The risks to two of the main target species, *X. kroyeri* and *P. subtilis*, although considered high by one of the methods used to define the boundaries, are not in the top ten of the vulnerability ranking. Traditional stock assessment developed in the region indicates that these species are caught at close to maximum levels, but within an acceptable level of exploitation (Lopes *et al.*, 2014; Silva *et al.*, 2015, 2018). However, the periodical monitoring of these two species is crucial for the sustainability of the ecosystem and fishery at a regional level, given (i) their target nature, (ii) their ecological and socioeconomic importance, (iii) the absence of current regulation, and (iv) their historically overexploited status within fisheries in other regions of Brazil.

As previously reported, subjectivity and, consequently, uncertainty are intrinsic to PSA and some choices related to the attributes used. The definition of its decision making in the analysis, therefore, directly affects its results. The approaches applied in the present study (see section *Measuring uncertainties* for details) were efficient in weighing the effect of different subjective choices within the analysis, resulting in more comprehensive results that are more useful for management in such data-poor frameworks.

Considering the previous studies on shrimp trawling activity in the region (Tischer and Santos, 2003; Lopes *et al.*, 2014; Eduardo *et al.*, 2018b; Silva *et al.*, 2018; Lira *et al.*, 2019, 2021b; Silva Júnior *et al.*, 2019), the target species are not currently those principally at risk from this fishery. Some species of the bycatch, however, should be carefully assessed and considered as priorities for management. The combined effect of the fishery and ongoing environmental changes, in terms of rainfall or in primary production, should also be considered because their interaction could have significant adverse impacts on ecosystem functioning (Lira *et al.*, 2021b).

Chapter main findings and Thesis outlook

In this Chapter, using a Productivity and Susceptibility Analysis (PSA) approach adapted to regional conditions, we evaluate, for the first time, the vulnerability and the potential risk of the target and non-target species exploited by the shrimp fishery in Sirinhaém coast as a case study of Northeast Brazil (Figure 5). Although the results presented should be considered with some caution and may refer, either for the target or non-target species, to one specific part of the population exploited or subjected to exploitation by small-scale shrimp trawling in the region, we believe that PSA method is important in highlighting the species that should be prioritized, either in urgent assessment and/or data collection.

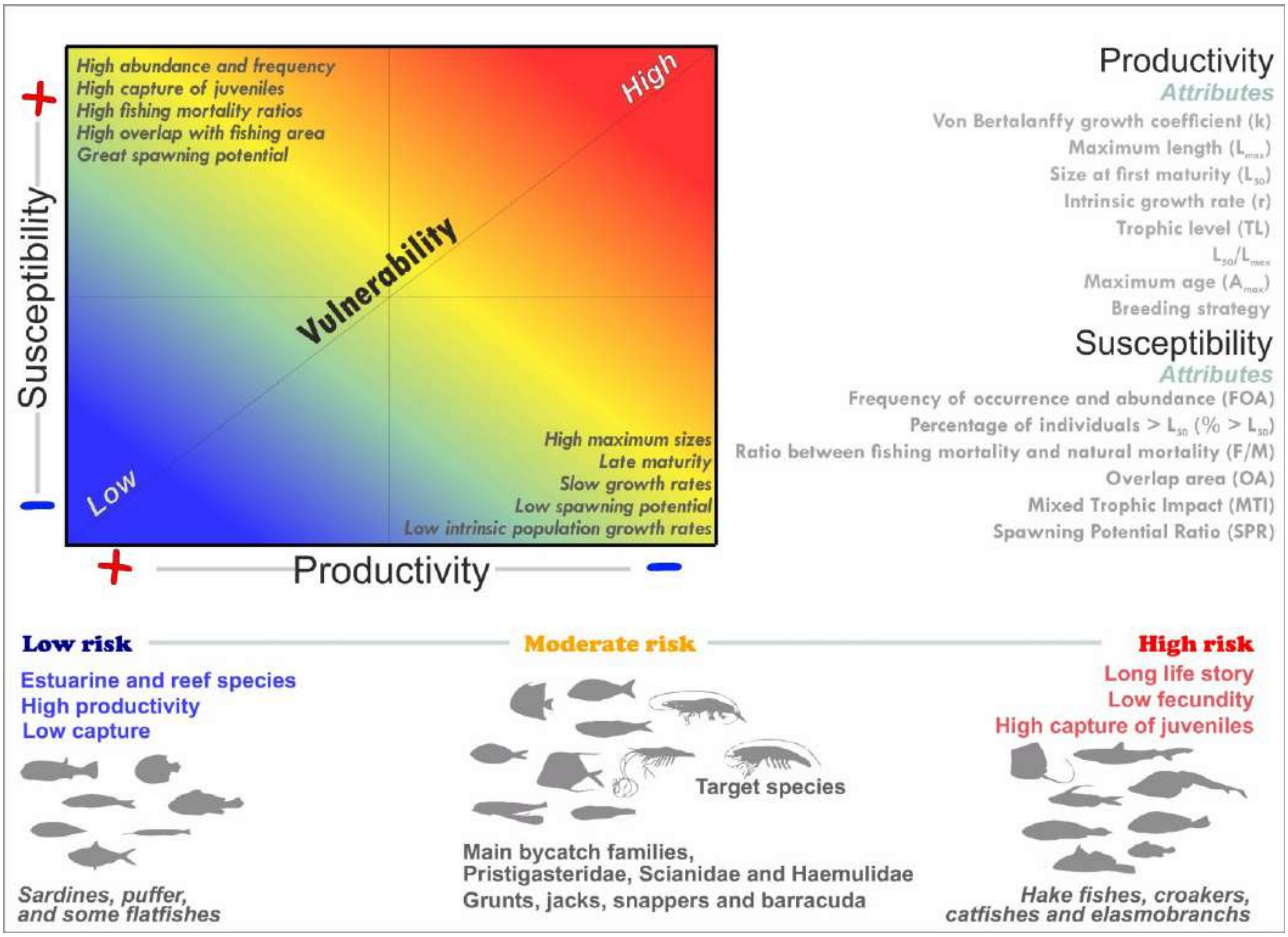

Figure 5. Diagram to represent the productivity and susceptibility considered for Barra of Sirinhaém, Pernambuco, north-eastern Brazil

Some species of the bycatch such as elasmobranchs, catfishes and some Scianidae, should be prioritized, either in urgent assessment and/or data collection (Figure 5). Elasmobranchs, catfish often discarded or consumed; hakes and croakers' fishes, usually commercialized were considered bycatch species of high vulnerability. The most abundant species of the bycatch (e.g., *Pellona harroweri*, *Isopisthus parvipinnis*, *Chirocentrodon bleekerianus*) were classified as the moderate risk (Figure 5), mainly due to their high resilience. Given the lack of information regarding population structure and life history traits and considering the beneficial role of the bycatch for the local communities as well as the potential negative effects from nutritional, economic and social viewpoints in a scenario of bycatch declining. These species deserve priority for research. The risks of two of the main target species (*X. kroyeri* and *P. subtilis*), although were considered high by one of the methods used to define the boundaries, the traditional stock assessment developed in the region indicates that these species are caught close to maximum, but within an acceptable level of exploration.

The results of the present Chapter, complemented by the findings of the previous Chapters (1, 2 and 3), allowed to conclude that currently, the target species are not the main threat of the small-scale shrimp trawling in the region. In addition, we identified several non-target species, often not considered in management measures that, given the high socio-economic importance in the region, need to be better assessed under the EAF, taking into the effect in whole trophic dynamic and the bycatch sustainability, essential for the food security.

Final considerations for management support

FINAL CONSIDERATIONS FOR MANAGEMENT SUPPORT

Fishery is considered as one of the main anthropogenic impact to the marine ecosystems, often associated to the depletion of stocks and degradation of habitats to levels, sometimes irreversible to the sustainable use (Worm *et al.*, 2009; Halpern *et al.*, 2015). This scenario requires decision makers to provide regulatory measures in order to mitigate the potential impacts for the fishery activity and overall ecosystem.

However, the ecosystem and fishery resources are still extremely poor in information. The exception applies mainly to target species of fisheries in developed countries, which have a larger research effort (Aksnes and Browman, 2016). However, there are still gaps of knowledge mainly related to the often-neglected non-target species. Moreover, in those cases, which are the case of most tropical fisheries, assessment do not take into account the climate, and the social, economic and cultural roles of the fishery.

The current paradigm and useful management framework that has been minimizing this gap is the Ecosystem Approach to Fishery (EAF). EAF considers "the knowledge and uncertainties about biotic, abiotic, and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries" (Garcia *et al.*, 2003). Although the EAF are extremely promising, to fulfil both the management objectives for fisheries development and ecosystem conservation (Figure 1), its applicability is often restricted by the limited or absence of an integrative view of the dynamic of ecosystem, fishery, economy, ecology and biology of the target and no-target species (Garcia *et al.*, 2003)

Small-scale fisheries are deeply linked to the history and culture of local fishing communities, providing to millions of persons livelihood, generating employment, income and food, having a strong influence on the regional economy in coastal communities worldwide (Chollett *et al.*, 2014; de Oliveira Leis *et al.*, 2019). Although occasionally, small-scale fishery uses specific gear and is focused on a few target species, in general, it is majoritary multi-gear and multi-species, with a high diversity of species caught, fleets and boats of different types and sizes that make integrated use of the ecosystem, sharing usually fishing areas with the industrial fleet, often resulting in conflicts (Figure 1). In many countries, it faces social difficulties, such as the lack of alternative occupations for fishermen (Cinner *et al.*, 2009), inadequate technical and financial support and weak governance (de Oliveira Leis *et al.*, 2019). In addition, it serious confronts environmental problems, such as pollution (Marín and Berkes, 2010), habitat degradation (Rogers *et al.*, 2018) and the collapse of fish stocks (Plank *et al.*, 2017). In developing countries (e.g., some Latin American nations), the ineffective implementation or the lack of public policies may have serious economic, ecological and food-security adverse consequences for the sector (Mattos and Wojciechowski, 2019), affecting mainly the local communities.

In this study, we applied multiple models under the scope of EAF for the shrimp small-scale fishery in Sirinhaém. Methods adapted to data-limit situations following the EAF principles were useful for assessing the fishery and the ecosystem. We are confident that our results may significantly contribute to the regional fishery management, and the methods here applied, can be replicated for another multispecies tropical fishery of the world, especially those of small-scale nature.

In the last decades, studies have been developed to evaluate the effect and feasibility of implementing fisheries management measures around the world (De Young *et al.*, 2009), in North (e.g., United States - Townsend *et al.* (2019); Canada - Davis (2008)) and South America (e.g., Peru – Aranda (2009); Brazil - Reis and D'Incao (2000), Isaac and Ferrari (2017)), Europe (e.g., several countries - Arlinghaus *et al.* (2002), Berghöfer *et al.* (2008), Bellido *et al.* (2011), Marchal *et al.* (2016)), Africa (e.g., Tunisia - Halouani *et al.* (2016); Senegal - Diedhiou and Yang (2018); South Africa - Shannon *et al.* (2010)), Asia (e.g., China - Shen and Heino (2014); Vietnam - Pomeroy *et al.* (2009), Anh *et al.* (2014); Thailand - Nasuchon and Charles (2010)) and Oceania (e.g., Australia - Smith *et al.* (1999)). Specifically for tropical shrimp fishing, only a few studies are available (Macfadyen *et al.*, 2013), mainly in Caribbean (Criales-Hernandez *et al.*, 2006) and Mexico (Foster and Vincent, 2010), however, this type of assessment is lacking for small-scale shrimp fisheries in Northeastern Brazil.

In general, management measures applied in tropical shrimp fishing may be divided on dimensions (e.g., temporal and spatial) that require different levels of information (Garcia *et al.*, 2003; Walters and Martell, 2004; King, 2007) (Figure 1):

i. Size dimension (mesh or species size control) – these measures are applied to gear, target or non-target species by determining minimum and maximum captured size or changes in gear and mesh size. Information about length/weight information by species, functioning of gear, reproduction and fishery information should be taken into account for the application of these measures.
ii. Effort control (gear limit and fleet licenses) – They are applied seasonally or annually, for a specific type of fleet or gear. These measures require estimates optimum sustainable effort, although the use of ecosystem models like EwE can compensate when catch and effort long time series are absent.
iii. Temporal dimension (closed season) – Regulatory measure more adopted in tropical and subtropical shrimp fisheries to protect a part of the population. It is defined according to the dynamics of the fishery and the bio-ecological patterns of target or non-target species in the ecosystem. It requires multiples information considering the biological, ecological, economic and social aspects.

iv. Catch dimension (quota) - They can be daily, seasonal or annual limits that apply to specific target, areas or vessels. These measures require precise stock assessment, catches and effort data, being more often available in data-rich situation.

v. Spatial dimension (closed areas) - Management tool suggested to protect spawning areas, juvenile aggregating areas, sensitive habitats, and also to separate different types of fishing gear. They are often defined for wide geographic area or zones considering depth, habitats and use of environment. Beyond the spatial detail of the habitats and their relationship to the distribution of the species, this measure needs a data integrated view of the multiple facets within the ecosystem.

vi. Gear modification - Adaptations incorporated into the gear, especially the net, with the aim of improving the productivity of the fishery or reducing the catch of non-target species. It needs detailed information on gear selectivity and productivity, including target and bycatch species. The most popular among them is the Turtle Excluder Device (TED) applied in the tropical shrimp industrial fleet.

According to the results of this thesis, considering the traditional approaches for fisheries regulation, we conclude, for the study case, the following (Figure 1):

i and ii) Although the controlled reduction of the current effort close to 10% was promising, the high decrease the effort or the definition of size and gear limits did not appear to be a necessary measure, considering that, according to the traditional stock assessment, the target species are being exploited at biologically accepted levels.

iii) Regarding the closed season for target species, we did not observe important improvement to the ecosystem and fishery given the seasonal pattern of the species ("natural closed season"). The low shrimp abundance is related to dry season which correspond to the peak of reproduction of these species, basically inactivating the trawling activities or making them economically unprofitable due to the decline in production that barely covers the operational costs of fishing.

v) In relation to the spatial dimension, we identified that the main fishing grounds were small and restricted to muddy beds close to coast. Thus, given its extension, spatial management approaches (e.g., Marine Protect Area – MPA or no-fishing zones) may be not very effective in a possible fisheries management in the region.

In contrast, the non-target species, not yet considered in management measures, given the high socio-economic importance of these for local community in the region, need to be better assessed under the Ecosystem Approach to Fishery (EAF) taking into account the effect in whole trophic dynamic and the bycatch sustainability. Several species were considered potentially vulnerable to bottom trawling, given its biology, ecology and importance to other fleets. For example, Elasmobranchs and catfish, often discarded or consumed had high vulnerability, mainly given the very low productivity, needs to be priority for research. In addition, hakes and croakers' fishes are economically important and a large proportion of catches are juveniles, which can be a threat to the sustainability of their stocks. Moreover, other fisheries in the region capture those groups as adults.

Thus, the use of Bycatch Reduction Devices (e.g., fisheye, grid and square mesh) to exclude bycatch may be one alternative. However, more information to evaluate its viability and especially what would be the socio-economic effect of the potential bycatch reduction on small-scale fisheries is needed. In this way, some initiatives in the world, such as the project Sustainable Management of Bycatch in Latin America and Caribbean Trawl Fisheries (REBYC-II LAC - http://www.fao.org/in-action/rebyc-2/overview/en/) of FAO with 4 pilot sites along of Brazilian coast (e.g. Pará, Pernambuco, Paraná/Santa Catarina and Rio Grande do Sul), are in course.

Regardless the measures that may be applied in the management of small-scale shrimp fisheries in Sirinhaém, Northeastern Brazil, we have found clear evidence that environmental changes (e.g., rainfall, primary productivity), resulting of the climate changes, cause significant adverse impacts in the ecosystem and should be considered in any eventual regulatory measures, since the cumulative effect of these changes and fishing, considerably threat the sustainability of the ecosystem, consequently of the fishery. Thus, considering everything exposed in the present thesis, some recommendations step by step for ecosystem managers are presented below:

i) Collect data on biology and ecology and assessment of high-risk species for which there is not enough information;
ii) Collect data on bottom recovery in the region, to access the option of rotative space management;
iii) Assess economic impacts of applying the BRD;
iv) Invest in public or private policies that encourage the increase of fishermen's income;
v) Promote communication of the results obtained in this work so that there is a better understanding of the problems and solutions;
vi) Reduction of effort by 10% or control effort in the future.

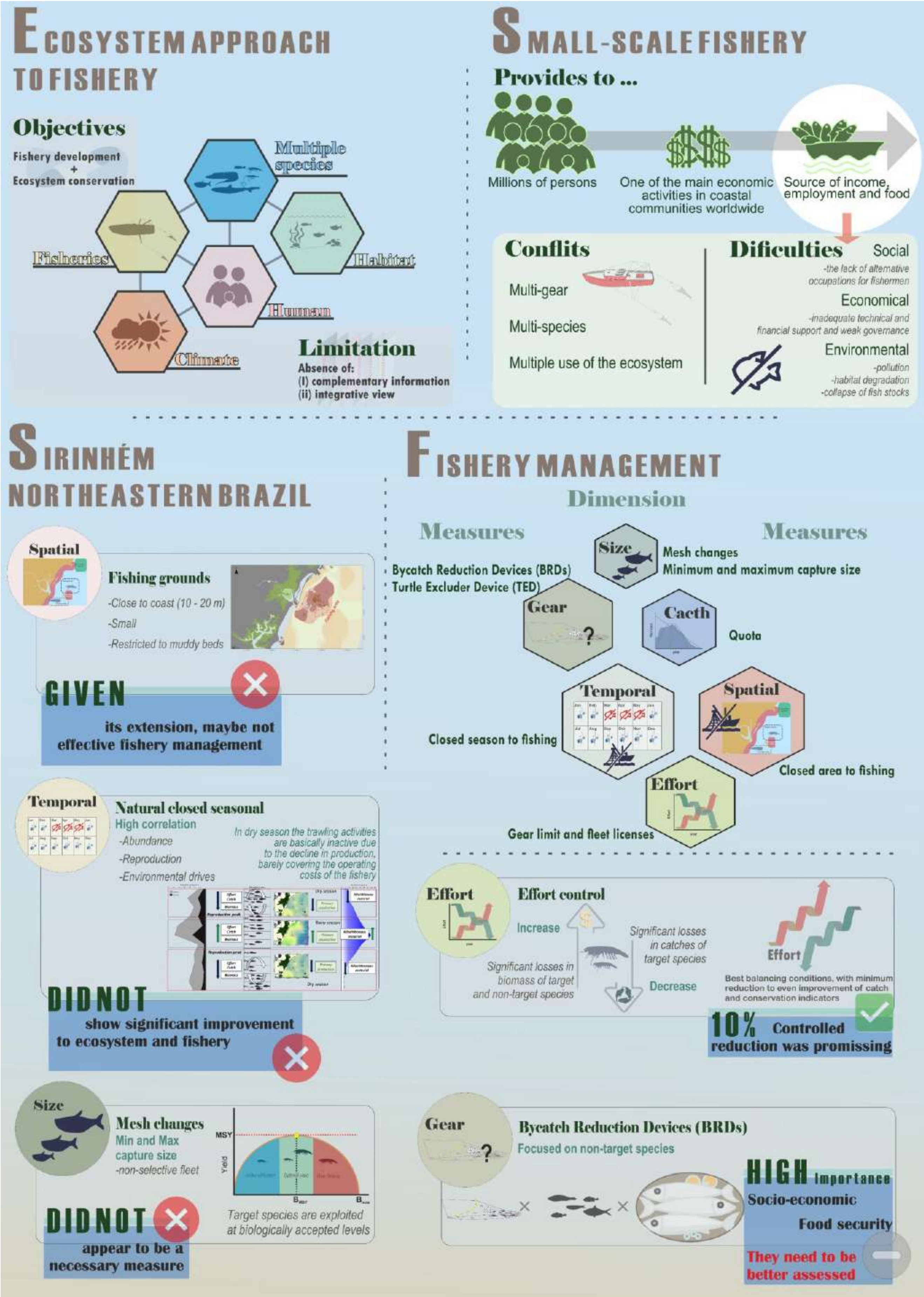

Figure1. Final considerations for Management support in Sirinhaém, Northeastern Brazil.

PERSPECTIVES AND WEAKNESSES TO IMPROVE

In the present thesis, considering the ecosystem approach to fisheries, we have focused mainly on the fisheries, habitat and species dimensions, however the human dimension, including social and economic aspects were hardly explored. As in any modeling approach, or in information-poor situations, the studies developed here required several assumptions and approximations to describe and predict ecosystem functioning, as well as to evaluate the effect of fishing on the species. The approaches and the quality of the results presented can and should be improved by including more accurate and detailed information as soon as it becomes available. Considering that the studies in the present thesis are developed around and with a focus on ecosystem management issues, these adjustments are crucial. For optimal ecosystem management it is decisive to consider all these dimensions, as absence of some of them can lead to a weakening of possible regulatory measures by not providing a holistic effect on the ecosystem.

Knowledge of the ecosystem structure

One of the first step to to provide management regulation is the knowledge of aspects that define the ecosystem structure, and how the dynamic of species and fishing interact between each other. Although we have defined the bottom morphology (sediment types and depth) and the basic climate pattern (rainfall and primary productivity), other important variables were neglected due to the lack of information, such as temperature patterns, salinity, river discharge flow, and organic matter in the sediment, limiting broader conclusions. These variables, if incorporated into the current ones, could help considerably for a more accurate characterization of the ecosystem and identification of the spatio-temporal patterns of the species. This more accurate description could be used as a basis for building spatial models of species distribution not only in the region, but in a wider geographic area.

Fishing monitoring

Fishery statitsics is another information that needs significant improvements. Who and how much ? These are simple questions, currently, with no answers for most fisheries along the Brazilian coast. In Brazil, the fisheries monitoring programs are outdated, compling the assumptions of large uncertainties when using the available information. Morever, most of the official statistical bulletins do not present detailed geographic (cities), resource (species) and scale (industrial and artisanal) information. The use of logbooks by the different sectores of the fishery (e.g. fishermen, vessel owners and intermediaries of the fishery) appears to be a promising alternative. In the present thesis, we benefited of logbooks from the local trawl fishery, with daily landing information by species and cost estimates. Hence, in the absence of what should always be the first option, the fishery monitoring programs at local, regional and national levels, such as the application of logbooks, may be a key element to obtain a more accurate information to use in validating traditional stock assessment or ecosystem models.

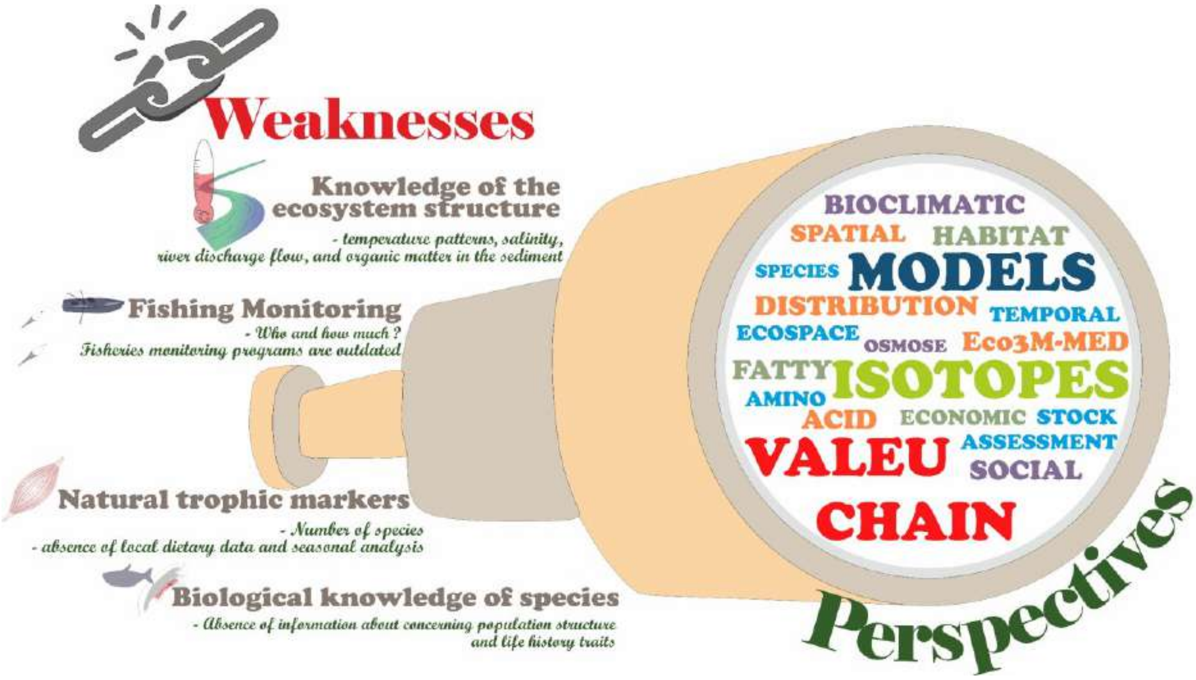

Figure 1. Perspectives and weaknesses to improve the approaches applied in the present thesis.

Natural trophic markers

The possibilities for improving the trophic structure study described in the present thesis can be divided into three parts: (i) increase in the number of functional groups and species evaluated with local diet information; (ii) inclusion of seasonal isotopic evaluation, given the influence of rainfall patterns in the coastal region; and (iii) the use of new natural markers for amino acid and fatty acid isotope analysis. These two markers have advantages over stable isotopes (SI) of carbon and nitrogen due to the potential to evaluate feeding on a finer scale compared to SI analysis. The fatty acids (FA) in marine food webs, may pass from prey to carnivore consumers relatively unchanged, allowing an accurate reflection of the prey's FA profile to be represented in the tissues of the consumer (Hoopes *et al.*, 2020; Twining *et al.*, 2020).The isotope analysis of amino acids can provides greater resolution to the estimation of trophic position (TP), since it considers predictable trophic increases in the $\delta^{15}N$ values between amino acids along food chain (McMahon and McCarthy, 2016). Thus, it would be possible to estimate trophic level from the $\delta^{15}N$ of amino acids within a single organism, avoiding the problems with bulk isotope analysis caused for example by variabilities in isotopic fractionation or by variabilities and mixing of potential isotopic baselines (Xing *et al.*, 2020).

Biological knowledge of species

Taking the bycatch into account, we found many gaps of information concerning population structure and life history traits (e.g., growth, mortality, breeding season, and feeding behavior) hampering the aplication of models with greater reliability. To overcome this, several empirical relationships to estimate lacking parameters (such as asymptotic length, growth coefficient and length at first maturity) were used to overcome the absence of data, however, there is a considerable amount of uncertainty in those empirical formulae, thus they should be used with caution. Population parameters are crucial for fishery management since they are required to the application of assessment and ecosystem approaches. Obtaining the population parameters of bycatch species should be priority for most multispecies fishery, such as the small-scale shrimp fishery here evaluated.

In addition, although we recognize the importance of incorporating specific periods of the closing season within scenarios simulated in the Ecosim model, some major data, as for example the target species spawning parameters (egg production, egg-laying timing etc.), are lacking, hampering the achievement of more accurate results within the model. The absence of these data does not allow a proper evaluation of the seasonal patterns of reproduction and abundance observed in the region.

Advances in ecosystem models

Methods that evaluate the spatial distribution of the species and the potential effect of climate change on the ecosystem, such as the bioclimatic envelope models (BEMs) and habitat models (HMs) (Le Marchand *et al.*, 2020), and Ecospace (Christensen *et al.*, 2009; Abdou *et al.*, 2016) allow for a better characterization of the environment and its relationship with the species providing significant improvements for ecosystem based management. The improvement in the description of the life cycle of the species, may also promote important adjustments in the Ecosim model with the inclusion of stanza groups to better represent life history stages (Christensen *et al.*, 2008; Walters *et al.*, 2008). The implementation of other models based on different hypothesis could be very relevant, such as OSMOSE, which assumes opportunistic predation based on spatial co-occurrence and size adequacy between a predator and its prey (size-based opportunistic predation) (Shin and Cury, 2001; Halouani *et al.*, 2016b).

The economic effects of fishing are also an important issue that can be addressed in the future by incorporating the "Value-Chain" (an economical "chain" of the resources, Christensen *et al*. 2011; Bevilacqua *et al.* 2019), that would enable useful insights on the effects of various management policies (e.g., quota or area closures) and possible trade-offs at the economic, social and ecosystem level. Specifically in our study case, the impact of the trawl fishing activities on the ecosystems appears to be counter-balanced by the beneficial role of the bycatch in the local community (Lira *et al.*, 2021b), hence, a major decline in the capture of bycatch may cause negative effects from nutritional, economic and social viewpoints. In Sirinhaém, our results indicated that only a small part of the bycatch is effectively discarded. This reinforces need to be consider the human dimension in future evaluations, not only in the region, but in other tropical small-scale fisheries, where this activity plays an essential role for nutrition, food security, employment and income. It also represents an important tool for achieving the Sustainable Development Goals (SDGs) proposed by United Nations (United Nations, 2015) specifically those of number 1- no poverty; 2- no hungry and 14- life bellow water; also taking into account The Voluntary Guidelines for ensuring Sustainable Small-Scale fisheries (FAO, 2018).

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APPENDICES

CHAPTER 1. The Ecosystem Approach to Fisheries in Action: a study case of the shrimps small-scale fishery in tropical Brazil

The supplementary material follows the order according to the manuscript presented in the Chapter 1:

Table S1 Review of biological traits L_∞ (asymptotic total length; cm), L_{50} (length at first maturity; cm) and K (growth coefficient; year^{-1}) for fish species caught as bycatch in Barra de Sirinhaém, Pernambuco Northeast Brazil. *L_∞ and k were estimated considering the maximum size of the literature (L_{max}) by empirical relationships of Froese and Binohlan (2000) ($\log_{10}(L_\infty) = 0.444 + 0.9841 \times \log_{10}(L_{max})$ and Le Quesne and Jennings (2012) ($K = 2.15 \times L_\infty{}^{-0.46}$), while the **red values** of L_{50} by ($\log L_{50} = -0.1189 + 0.9157 \times \log L_{max}$) according Binohlan and Froese (2009)

Espécies	Cod	L_∞ (cm)	L_{50} (cm)	K (ano^{-1})	Sources
*Acanthostracion polygonius**	aca.pol	51.99	**27.3**	0.349	(Menezes and Figueiredo, 1980)
*Achirus declivis**	ach.dec	19.75	**11.1**	0.545	(Joyeux *et al.*, 2009)
*Achirus lineatus**	ach.lin	34.64	**18.7**	0.4209	(Joyeux *et al.*, 2009)
*Albula nemoptera**	alb.nem	53.02	**27.8**	0.346	(Robins and Ray, 1986)
*Anchoa januaria**	anc.jan	10.88	6.5	0.717	(Esper, 1982; Franco *et al.*, 2014)
*Anchoa spinifer**	anc.spi	25.25	**14**	0.486	(Cervigón *et al.*, 1992)
Anchoa tricolor	anc.tri	10.4	7.6	1.650	(Silva Júnior *et al.*, 2013; Carvalho, 2014)
Anchoviella lepidentostole	anc.lep	14.3	9.4	0.830	(Giamas *et al.*, 1985; Camara *et al.*, 2001)
*Anisotremus moricandi**	ani.mor	31.5	**9.1**	0.439	(Moura *et al.*, 1999)
*Aspistor luniscutis**	asp.lun	123.06	18	0.235	(Mishima and Tanji, 1983; Burgess, 2004)
*Aspistor quadriscutis**	asp.qua	51.99	**27.3**	0.349	(Carpenter, 2002b)
*Atherinella brasiliensis**	ath.bra	16.94	9.1	0.584	(Cervigón *et al.*, 1992; Bervian and Fontoura, 1997)
*Bagre bagre**	bag.bag	57.10	21.2	0.334	(Cervigón *et al.*, 1992; Véras and Da Silva Almeida, 2016)
*Bagre marinus**	bag.mar	51.99	39	0.349	(Lima *et al.*, 2016)
*Bairdiella ronchus**	bai.ron	36.60	15.8	0.410	(Chao, 1978; Torres Castro *et al.*, 1999)
*Carangoides bartholomaei**	car.bar	102.84	30	0.255	(Cervigón *et al.*, 1992; Santos, 2012)
*Caranx hippos**	car.hip	127.09	66	0.231	(Cervigón *et al.*, 1992; García-Cagide *et al.*, 1994)
*Cathorops spixii**	cat.spi	31.45	**17.1**	0.440	(Carpenter, 2002b)
*Cetengraulis edentulus**	cet.ede	19.23	11.8	0.551	(Souza-Conceição *et al.*, 2005; Joyeux *et al.*, 2009)
Chaetodipterus faber	cha.fab	50.88	15.8	0.220	(Soeth *et al.*, 2019)
*Chirocentrodon bleekerianus**	chi.ble	17.04	7.6	0.583	(Corrêa *et al.*, 2005; Barreto *et al.*, 2018)
*Chloroscombrus chrysurus**	chl.chr	50.25	15.5	0.354	(Cervigón *et al.*, 1992; de Queiroz *et al.*, 2018)
*Citharichthys macrops**	cit.mac	21.10	**11.8**	0.528	(Robins and Ray, 1986)
*Citharichthys spilopterus**	cit.spi	22.14	11.7	0.517	(Dias *et al.*, 2005; Barreto *et al.*, 2018)
*Conodon nobilis**	con.nob	35.77	14.3	0.414	(Pombo *et al.*, 2014; Lira *et al.*, 2019)
Cyclopsetta chittendeni	cyc.chi	33.00	**18.2**	0.780	(Pauly, 1994)
*Cynoscion virescens**	cyn.vir	118.01	**58.6**	0.239	(IGFA, 2001)
Dactylopterus volitans	dac.vol	33.58	**27.3**	0.301	(da Costa *et al.*, 2018)
Diapterus auratus	dia.aur	44.60	17.6	0.374	(Cervigón, 1993; Conceição, 2017)
Diapterus rhombeus	dia.rho	26.25	15.2	0.240	(Bezerra *et al.*, 2001; Elliff *et al.*, 2013)
Diplectrum formosum	dip.for	20.40	**17.1**	0.701	(Bubley and Pashuk, 2010)
Echeneis naucrates	ech.nau	60.30	**39.4**	0.250	(Bachman *et al.*, 2018)
Etropus crossotus	etr.cro	17.00	10.32	1.601	(Rábago-Quiroz *et al.*, 2008; Oliveira and Favaro, 2011)
Eucinostomus argenteus	euc.arg	28.31	8.03	0.610	(Silva *et al.*, 2014; Leão, 2016)
Eucinostomus gula	euc.gul	22.30	11	0.290	(Mexicano-Cíntora, 1999; García and Duarte, 2006)
*Genyatremus luteus **	gen.lut	38.66	34.5	0.400	(Cervigón, 1993; Gómez *et al.*, 2002)
Haemulon aurolineatum	hae.aur	24.20	11.7	0.234	(Lessa *et al.*, 2004; Cardoso de Melo *et al.*, 2020)
Haemulon plumierii	hae.plu	34.21	13.9	0.070	(Vasconcelos-Filho *et al.*, 2018; Cardoso de Melo *et al.*, 2020)

Haemulon steindachneri	hae.ste	31.00	**17.1**	0.210	(García and Duarte, 2006)
*Haemulopsis corvinaeformis**	hae.cor	26.28	11.45	0.477	(Eduardo *et al.*, 2018)
*Harengula clupeola**	har.clu	23.68	**10.7**	0.430	(da Costa *et al.*, 2018)
*Hypanus guttatus**	hyp.gut	203.44	67.2	0.186	(da Silva *et al.*, 2018)
Hyporhamphus unifasciatus	hyp.uni	30.40	18.9	1.460	(Cervigón *et al.*, 1992; Lessa *et al.*, 2004)
*Isopisthus parvipinnis**	iso.par	26.28	14.4	0.477	(Cervigón, 1993; Silva Júnior *et al.*, 2015)
*Lagocephalus laevigatus**	lag.lae	102.84	**51.6**	0.255	(Shipp, 1981)
*Larimus breviceps**	lar.bre	32.482	14.04	0.433	(Cervigón *et al.*, 1992; Silva Júnior *et al.*, 2015)
*Lepophidium brevibarbe**	lep.bre	28.66	**15.7**	0.459	(Robins *et al.*, 2012)
Lutjanus analis	lut.ana	84.50	31.22	0.050	(Lessa *et al.*, 2004; Teixeira *et al.*, 2010)
Lutjanus synagris	lut.syn	46.80	17.1	0.105	(Lessa *et al.*, 2004; Viana *et al.*, 2015)
Lycengraulis grossidens	lyc.gro	26.00	12	0.420	(Goulart *et al.*, 2007; Mai and Vieira, 2013)
Macrodon ancylodon	mac.anc	47.10	21.13	0.430	(Ikeda, 2003; Cardoso *et al.*, 2018)
Menticirrhus americanus	men.ame	41.80	16.7	0.290	(Giannini and Paiva-Filho, 1992; Freitas *et al.*, 2011)
*Menticirrhus littoralis**	men.lit	50.25	23	0.354	(IGFA, 2001; Braun and Fontoura, 2004)
Micropogonias furnieri	mic.fur	60.00	34.1	0.050	(Santos, 2015)
*Myrichthys ocellatus**	myr.oce	112.96	**56.3**	0.244	(Smith, 1997)
*Nebris microps**	neb.mic	41.74	**22.3**	0.386	(Keith *et al.*, 2000)
Odontognathus mucronatus	odo.muc	28.80	**11.4**	0.350	(Silva-Júnior, 2004)
*Ogcocephalus vespertilio**	ogc.ves	31.96	**17.4**	0.436	(Claro, 1994)
*Ophioscion punctatissimus**	oph.pun	26.28	11.1	0.477	(Chao, 1978; Conceição, 2017)
Opisthonema oglinum	opi.ogl	31.80	12.5	1.460	(Lessa *et al.*, 2004; Simoni, 2019)
*Paralichthys brasiliensis**	par.bra	102.84	**51.6**	0.255	(Carvalho-Filho, 1992)
Paralonchurus brasiliensis	par.bra	20.00	14.7	0.535	(Dos S. Lewis and Fontoura, 2005; Silva Júnior *et al.*, 2015)
*Pellona harroweri**	pel.har	19.02	10.7	0.554	(Cervigón *et al.*, 1992; Conceição, 2017)
*Pempheris schomburgkii**	pem.sch	15.89	**9.1**	0.602	(Claro, 1994)
*Peprilus paru**	pep.sch	31.45	15.56	0.440	(Cerqueira and Haimovici, 1990; Claro, 1994)
*Polydactylus virginicus**	pol.vir	34.54	17.4	0.421	(Motomura, 2004; Conceição, 2017)
Prionotus punctatus	pri.pun	52.70	26.2	0.067	(Teixeira and Haimovici, 1989; Andrade, 2004)
Pseudobatos percellens	pse.per	109.31	58.3	0.160	(Rocha and Gadig, 2013; Caltabellotta *et al.*, 2019)
*Rhinosardinia bahiensis**	rhi.bah	8.56	**5.1**	0.800	(Whitehead *et al.*, 1988)
Rhizoprionodon porosus	rhi.por	136.40	65	0.077	(Lessa and Santana, 1998; Mattos *et al.*, 2001)
Sciades herzbergii *	sci.her	96.97	28.3	0.262	(Chacon *et al.*, 1994; Conceição, 2017)
Selene brownii	sel.bro	29.50	**16.6**	0.230	(García and Duarte, 2006)
Selene setapinnis	sel.set	23.80	**13.9**	0.450	(García and Duarte, 2006)
Selene vomer	sel.vom	31.50	**26.5**	0.430	(García and Duarte, 2006)
*Sphoeroides greeleyi**	sph.gre	19.02	7.5	0.554	(Cervigón *et al.*, 1992; Schultz *et al.*, 2002)
Sphoeroides testudineus	sph.tes	30.00	10.8	0.510	(Pauly, 1991; Rocha *et al.*, 2002)
*Sphyraena guachancho**	sph.gua	203.44	28.8	0.186	(Reiner, 1996; Akadje *et al.*, 2019)
*Stellifer brasiliensis**	ste.bra	17.98	7.3	0.569	(Trindade-Santos and Freire, 2015; Barreto *et al.*, 2018)
Stellifer microps	ste.mic	24.96	10.4	0.300	(Sarmento, 2015; Silva Júnior *et al.*, 2015)
Stellifer rastrifer	ste.ras	20.90	11.2	0.370	(Pombo *et al.*, 2013; Conceição, 2017)
*Stellifer stellifer**	ste.ste	15.16	7.5	0.615	(Trindade-Santos and Freire, 2015; Dias *et al.*, 2017)
*Symphurus plagusia**	sym.pla	26.28	**14.5**	0.477	(Keith *et al.*, 2000)
*Symphurus tessellatus**	sym.tes	23.17	**12.9**	0.506	(Barreto *et al.*, 2018)
Trichiurus lepturus	tri.lep	127.40	41.6	0.399	(Al-Nahdi *et al.*, 2009; Barreto *et al.*, 2017)
Trinectes paulistanus *	tri.pau	19.23	**10.8**	0.551	(Barreto *et al.*, 2018)
Umbrina coroides	umb.cor	23.10	**19.7**	0.170	(García and Duarte, 2006)
*Upeneus parvus**	upe.par	31.45	**17.1**	0.440	(Smith, 1997)
Urotrygon microphthalmum	uro.mic	28.13	**7.3**	0.363	(Santander Neto, 2015)

Table S2. Definition of trophic guilds for the main species caught as bycatch by artisanal trawl fishery in Sirinhaém, Pernambuco, Northeast Brazil.

Groups/species	**Code**	**Guilds**	**Source**
Diapterus auratus	dia.sp	Zoobenthivore	(Lira *et al.*, 2021a)
Symphurus tessellatus	sym.tes	Zoobenthivore	(Guedes *et al.*, 2004)
Diapterus rhombeus	dia.sp	Zoobenthivore	(Lira *et al.*, 2021a)
Lutjanus synagris	lut.syn	Zoobenthivore	(Costa, 2013)
Chirocentrodon bleekerianus	chi.ble	Zoobenthivore	(Muto *et al.*, 2008)
Eucinostomus argenteus	euc.sp	Omnivore	(Lira *et al.*, 2021a)
Caranx hippos	car.hip	Piscivore	(Lira *et al.*, 2021a)
Micropogonias furnieri	mic.fur	Omnivore	(Freret and Vanderli, 2003)
Bagre marinus	bag.mar	Zoobenthivore	(Lira *et al.*, 2021a)
Larimus breviceps	lar.bre	Zoobenthivore	(Bessa *et al.*, 2014)
Stellifer microps	ste.mic	Zoobenthivore	(Lira *et al.*, 2021a)
Isopisthus parvipinnis	iso.par	Piscivore	(Lira *et al.*, 2021a)
Conodon nobilis	con.nob	Piscivore/Zoobenthivore	(Lira *et al.*, 2021a)
Paralonchurus brasiliensis	par.bra	Zoobenthivore	(Lira *et al.*, 2021a)
Achirus declivis	flatfish	Zoobenthivore	(Corrêa and Uieda, 2007)
Anchoa spinifer	anc.spi	Piscivore/Zoobenthivore	(Lira *et al.*, 2018)
Aspistor luniscutis	asp.lun	Omnivore	(Denadai *et al.*, 2004)
Cetengraulis edentulus	cet.ede	Zooplanktivore	(Sergipensel *et al.*, 1999; Krumme *et al.*, 2008)
Cynoscion virescens	cyn.vir	Zoobenthivore	(Lucena *et al.*, 2000)
Odontognathus mucronatus	odo.muc	Zooplanktivore	(Muto *et al.*, 2008)
Haemulopsis corvinaeformis	ham.cor	Zoobenthivore	(Regina Denadai *et al.*, 2013)
Hypanus guttatus	hyp.gut	Zoobenthivore	(Gianeti, 2011)
Lycengraulis grossidens	lyc.gro	Zooplanktivore	(Silva, 2012)
Macrodon ancylodon	mac.anc	Piscivore	(Castro *et al.*, 2015)
Menticirrhus americanus	met.ame	Zoobenthivore	(Lira *et al.*, 2021a)
Nebris microps	neb.mic	Zoobenthivore	(Chao, 1978)
Ophioscion punctatissimus	oph.pun	Zoobenthivore	(Zahorcsak *et al.*, 2000)
Stellifer brasiliensis	ste.bra	Zoobenthivore	(Sabinson *et al.*, 2015; Almeida, 2018)
Stellifer rastrifer	ste.ras	Zoobenthivore	(Sabinson *et al.*, 2015)
Stellifer stellifer	ste.ste	Zoobenthivore	(Pombo, 2010)
Trichiurus lepturus	tri.lep	Piscivore	(Martins *et al.*, 2005)
Sphyraena guachancho	sph.gua	Piscivore	(Akadje *et al.*, 2013)
Pellona harroweri	pel.har	Zooplanktivore	(Claudio Höfling *et al.*, 1998; Criales-Hernández, 2003; Muto *et al.*, 2008)
Polydactilus virginicus	pol.vir	Zoobenthivore	(Lopes and Oliveira-Silva, 1998)

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CHAPTER 2. Trophic structure of nektobenthic community exploited by a multispecific bottom trawling fishery in Northeastern Brazil

The supplementary material follows the order according to the manuscript published in the PlosOne:

Supporting information

S1 Table. Complementary sampling information. Mean, minima, maxima size, number of samples (n) in each quarter/year by species/group considered off the Sirinhaém coast, north- eastern Brazil. For fish the size is related to standard length (cm); *for shrimps, carapace length (cm) and ** for mollusk, mantle length (cm).

S2 Table. Additional diet data information considered to present study off the Sirinhaém coast, Northeastern Brazil. Location and year of data, total length range used and whether seasonal or ontogenic characteristics were considered (yes (y) or no (n)).

Table S1

Groups/species	Code	n	Mean size [min – max] (cm)	2013		2014			2015				2019
				third quarter	**fourth quarter**	**first quarter**	**second quarter**	**third quarter**	**first quarter**	**second quarter**	**third quarter**	**fourth quarter**	**fourth quarter**
Basal sources													
Sedimentary organic matter	SOM	8	-						6		2		
Lobophora variegata	lob.var	6	-						3		3		
Gracilaria cervicornis	gra.cer	6	-						3		3		
Sargassum sp.	sar.sp	6	-						3		3		
Particulate organic matter	POM	5	-						3		2		
Invertebrates													
Zooplankton - Copepoda	zoo.cop	6	-						3		3		
**Penaeus subtilis*	pen.sub	14	2.55 [1.9 to 3.3]		3	5			3		3		
**Penaeus schmitti*	pen.sch	20	3.31 [2 to 4.1]		4	6	4		2		4		
**Callinectes danae*	cal.dan	5	6.1 [5.9 to 6.3]								5		
**Callinectes ornatus*	cal.orn	3	4.9 [4.8 to 5.1]						3				
**Xiphopenaeus kroyeri*	xip.kro	17	1.76 [1 to 2.1]	2		7			5		3		
***Lolliguncula brevis*	log.bre	5											5
Fishes													
Citharichthys spilopterus	Cit.spi	3	11.1 [8.9 to 13.2]						1		2		
Diapterus auratus	Dia.aur	7	12.94 [10.5 to 17.5]		1						5	1	
Opisthonema oglinum	Opi.ogl	8	15.08 [9.4 to 17]			6		2					
Symphurus tessellatus	Sym.tes	6	15.01 [14.1 to 16.2]						3		3		
Diapterus rhombeus	Dia.rho	8	10.27 [10.2 to 10.4]					5	2			1	
Lutjanus synagris	Lut.syn	6	13.03 [7.7 to 19]							3	3		
Bairdiella ronchus	Bai.ron	3	11.23 [11 to 11.4]							3			
Chirocentrodon bleekerianus	Chi.ble	4	10.06 [10.3 to 10.9]										4
Eucinostomus argenteus	Euc.arg	14	8.52 [6.5 to 11.9]			3				11			
Bagre bagre	Bag.bag	3	9.67 [7.9 to 13]									3	
Caranx hippos	Car.hip	8	16.8 [16.5 to 17.2]									8	
Micropogonias furnieri	Mic.fur	7	25.45 [24.5 to 26.8]		1	4	1	1					
Bagre marinus	Bag.mar	8	9.13 [7.1 to 12]			5			3				
Larimus breviceps	Lar.bre	3	12.00 [9.6 to 13.7]										3
Stellifer microps	Ste.mic	4	12.02 [11.3 to 13.5]										4
Isopisthus parvipinnis	Iso.par	4	10.25 [9.1 to 13.6]										4
Conodon nobilis	Con.nob	4	9.57 [7.4 to 10.9]										4
Paralonchurus brasiliensis	Par.bra	3	14.33 [11 to 20.1]										3

Table S2

Species	Cod	Site	n	Total length (cm)	Year	Seasonality (y/n)	Ontogeny (y/n)	Source
Bagre bagre	bag.bag	Maranhão, Brazil	-	-	-	-	-	(Pinheiro-Sousa et al. 2015)
Bagre marinus	bag.mar	Pernambuco, Brazil	105	[17.40 ± 9.9 cm]	2013-2014	n	n	our data
Bairdiella ronchus	bai.ron	Pernambuco, Brazil	62	[16.68 ± 1.8 cm]	2013	n	n	our data
Caranx hippos	car.hip	Pernambuco, Brazil	15	[14.18 ± 1.7 cm]	2013	n	n	our data
Chirocentrodon bleekerianus	chi.ble	Sao Paulo, Brazil	-	-	-	-	-	(Muto et al. 2008)
Citharichthys spilopterus	cit.spi	Rio de Janeiro, Brazil	-	-	-	-	-	(Guedes et al. 2004)
Conodon nobilis	con.nob	Pernambuco, Brazil	165	[13.36 ± 3.3 cm]	2011-2012	y	y	our data
Diapterus auratus	dia.aur	Pernambuco, Brazil	74	[17.22 ± 5.6 cm]	2013-2014	n	n	our data
Diapterus rhombeus	dia.rho	Pernambuco, Brazil	25	[8.50 ± 2.0 cm]	2013-2014	n	n	our data
Eucinostomus argenteus	euc.arg	Pernambuco, Brazil	332	[8.62 ± 38 cm]	2013-2014	y	y	our data
Isopisthus parvipinnis	iso.par	Pernambuco, Brazil	69	[14.50 ± 3.6 cm]	2011-2012	n	n	our data
Larimus breviceps	lar.bre	Rio de Janeiro, Brazil	-	-	-	-	-	(Bessa et al. 2014)
Lutjanus synagris	lut.syn	Rio Grande do Norte, Brazil	-	-	-	-	-	(Costa 2013)
Micropogonias furnieri	mic.fur	Rio de Janeiro, Brazil	-	-	-	-	-	(Freret & Vanderli 2003)
Opisthonema oglinum	opi.ogl	Sao Paulo, Brazil	-	-	-	-	-	(Caludio Höfling et al. 1998)
Paralonchurus brasiliensis	par.bra	Pernambuco, Brazil	72	[13.60 ± 23 cm]	2011-2012	n	n	our data
Stellifer microps	ste.mic	Pernambuco, Brazil	145	[11.46 ± 22 cm]	2011-2014	y	y	our data
Symphurus tessellatus	sym.tes	Rio de Janeiro, Brazil	-	-	-	-	-	(Guedes et al. 2004)
Callinectes danae	cal.dan	Santa Catarina, Brazil	-	-	-	-	-	(Branco & Verani 1997)
Callinectes ornatus	cal.orn	Santa Catarina, Brazil	-	-	-	-	-	(Olinto Branco et al. 2002)
Lolliguncula brevis	lol.bre	São Paulo, Brazil	-	-	-	-	-	(Coelho et al. 2010, ZALESKI 2010)
Penaeus schmitti	pen.sch	Pernambuco, Brazil	36	[8.91 ± 20 cm]	2018-2019	y	n	our data
Penaeus subtilis	pen.sub	Pernambuco, Brazil	45	[9.50 ± 22 cm]	2018-2019	y	n	our data
Xiphopenaeus kroyeri	xip.kro	Pernambuco, Brazil	117	[6.98 ± 1.3 cm]	2018-2019	y	n	our data
Zooplankton	zoo	-	-	-	-	-	-	

CHAPTER 3. How the fishing effort control and environmental changes affect the sustainability of a tropical shrimp small scale fishery

The supplementary material follows the order according to the manuscript published in the Fishery Research:

Appendix 1

Ecopath with Ecosim approach

Description and information of the input parameters of the baseline Ecopath model

Appendix 2

Taxonomic data

Table S1. Group name, taxonomic composition and trophic guilds of each compartment

Source of parameters

Table S2. Source of input data by compartment for Barra of Sirinhaém Ecopath model (BSIR)

Estimate of the production/biomass (P/B)

Figure S1. Catch curve to estimate the total mortality (Z) of the fishes and shrimps of the Barra of Sirinhaém Ecopath model (BSIR)

Consumption information

Table S3. Parameters used as input for the estimation of the annual food consumption/biomass ratio (Q/B) of the fish group

Stable isotopes processing and results

Process description and isotopes signature of the groups in baseline Ecopath model

Access to Ecopath model results

Summary of the results found in the baseline Ecopath model

Balance of the model

Table S4. Outputs used for evaluating the balance of the model

Figure S2. Outputs of PRE-BAL routine of the Barra of Sirinhaém Ecopath model (BSIR)

Diet matrix

Table S5. Final diet matrix applied for Barra of Sirinhaém Ecopath model (BSIR)

EwE x Stable Isotope Analysis

Figure S3 Correlation between the trophic level mean (TL) estimated from BSIR Ecopath model and the mean nitrogen composition ($\delta^{15}N$).

Mixed Trophic Impact

Figure S4. Mixed Trophic Impact (MTI) analysis of Barra of Sirinhaém Ecopath model

Key species of the model

Figure S5. Keystonness index for each group of the Barra of Sirinhaém Ecopath model (BSIR)

Primary production

Figure S6. The primary production obtained from satellite image processed data (Level-3) of near-surface concentration of chlorophyll-a, for the period (Jan/97-Dec/13 from SEAWIFS and MODIS/AQUA)

Vulnerability to Ecosim model

Table S6. Vulnerability values applied to provide the best fit for BSIR Ecosim model.

Ecological indicators simulated to past

Figure S7. Ecological indicators estimated from the Ecosim results for the period 1988–2014 for of the Barra of Sirinhaém Ecopath model (BSIR)

Biomass and catch ratio variation

Table S7. Ecosim simulation results for each shrimp species and FMS compared to the baseline scenario

Biomass predicted simulated to future with Monte Carlo routine

Figure S8. Biomass predicted in the model with confidence interval by Monte Carlo routine (1000 runs) for each group and FMS.

Ecological indicators simulated to Future

Figure S9. Ecological indicators estimated from the Ecosim results for the period 1988–2030 for of the Barra of Sirinhaém Ecopath model (BSIR)

Appendix 1

Ecopath with Ecosim approach

Ecopath with Ecosim approach

The Ecopath model (Christensen and Walters, 2004) is built on a system of linear equations to describe average flows of mass among species and/or functional groups. The flow to and from each compartment is described by the main Ecopath equation representing production of each group (Christensen and Pauly, 1992):

$$B_i \times PB_i \times EE_i \times \sum_{j=1} \left(B_j \times QB_j \times DC_{ji}\right) - EX_i = 0 \text{ (eq.1)}$$

where B_i is the biomass of group (i); PB_i is the production/biomass ratio of (i), which is equal to total mortality (Z) or natural mortality (M) (Allen, 1971); EE_i is the ecotrophic efficiency of (i), which varies from 0 to 1 and represents the part of the production of the group that is transferred to higher trophic levels and/or fishing; B_j is the biomass of the predator (j); QB_j is the food consumption per unit of biomass for predator (j); DC_{ji} is the fraction (%) of (i) in the diet of (j); EX_i is the export of (i) and refers to the biomass that is caught through fishing and/or that migrates to other environments. In this case, as for other Ecopath models (Coll et al., 2006; Patrício and Marques, 2006; Han et al., 2016), we considered migration equal to immigration, given the difficulty of estimating the individuals movements.

For *n* groups (compartments), the model has a system of *n* linear equations. At least three from four of the input parameters B, PB, QB and EE have to be fixed in order to parameterize an Ecopath model. By connecting the production of one group with the consumption by the others, the missing parameter can be estimated based on the assumption that the production of one group is utilized by another group inside the system (Christensen and Pauly, 1992). Biomasses were expressed in $t.km^{-2}$ and flows in the food web in $t.km^{-2}.year^{-1}$.

Biomass

Fishes and shrimps were captured monthly (August 2011 to July 2012) by accompanying the local fishers (outrigger trawlers). The fishery operated from 1.5 to 3.0 miles off the coast, mainly between 10 and 20 m depth. For each sample (month), three sets of 2 hourly trawls were performed during the daytime, with boat velocity varying between 2 and 4 knots, using a double trawl (length: 10 m; horizontal opening: 6.1 m; mesh size body: 30 mm; mesh size cod end: 25 mm). Additionally, a GPS was used to access the distance covered for each trawl.

The biomass for these compartments were estimated through swept-area sampling method (Silva Júnior et al., 2019), expressed in $t{\cdot}km^{-2}$ using the sum of the catch individual weights (W, tonnes) divided by the total swept area (a; km^2). The covered area was estimated as: a= D.H.X; where, D is the distance covered (km) obtained by GPS tracking; H is the head-rope length (0.012 km) and X is the fraction of the head rope length = 0.5 (Pauly, 1980).

The phytoplankton, macroalgae, zooplankton, squid, birds and turtle groups biomasses were obtained from studies conducted in tropical systems near our study site (Freire et al., 2008; Guimarães et al., 2018; Mello, 2009; Opitz, 1996; Silva et al., 2016; Sousa and Cocentino, 2017). For all other groups, the biomasses were estimated by fixing the EE (Table S2).

Production (P/B)

For all groups, except fishes and shrimps (*Xiphopenaeus kroyeri, Penaeus subtilis* and *P. schmitti*), the production/biomass rates (P/B) were obtained from the literature (Table S2). The (P/B) can be estimated under mass-balance conditions as total mortality (Z) (Allen, 1971), which is the sum of the fishing mortality (F) and natural mortality (M). Here, Z was estimated by Length-based methods (e.g., Catch curve and Powell–Wetherall plot) (Pauly, 1983; Wetherall, 1986; Schwamborn, 2018) (see Figure S1). For the species that is not fished, P/B was equal to M, which was computed in accordance with Pauly (1980):

$$M = k^{0.65} \times L_{\infty}^{-0.279} \times T^{0.463} \text{ (eq.3)}$$

where M is the 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 with the empirical equations of Le Quesne and Jennings (2012) and Froese and Binohlan (2000), respectively. T was measured *in situ* and considered to be the mean annual temperature, 28°C.

Consumption (Q/B)

The consumption/biomass rate (Q/B) was estimated according to the following equation (Palomares and Pauly, 1998):

$$\log Q/B = 7.964 - 0.204 \times \log W_{\infty} - 1.965 \times T' + 0.083 \times Ar + 0.532 \times H + 0.398 \times D \text{ (eq. 4)}$$

where W_{∞} is the asymptotic weight (g), T' is the temperature in Kelvin (T' = 1000/(T°C+273.15)), and Ar is the aspect ratio of the caudal fin. W_{∞} was estimated by the equation $W_{\infty} = a \times L_{\infty}^{b}$, where (*a*) and (*b*) were based on Viana et al. (2016). Photographic records of the caudal fin were taken for each species with the *ImageJ* software (see Table S2). Ar was calculated as $Ar = h^2 / s$, where (h) is height of the caudal fin and (s) is the surface area of the fin, extending to the narrowest part of the caudal peduncle (Palomares and Pauly, 1998). H and D represent the feeding type (H = 1 for herbivores; D = 1 for detritivores; H = D = 0 for other feeding habits). This method was applied specifically for fishes, while that values of literature was used for another organisms Table S2. See Table S3 for the parameters used to calculate the consumption/biomass rate (Q/B) and the references.

Diet composition

The diet information for each fish compartment was primarily estimated from stomach content analyses carried out in the study area or, when data from a stomach content analysis was not available, based on the literature (see Table S2 for sources). For phytoplankton feeders, the excretion/egestion physiological rate was fixed at 40% in accordance with the recommendation of Heymans et al. (2016).

Fishery landings

Data of the BSIR fishery landings for bottom trawl, gillnet and line gear applied to characterize the fisheries in the baseline model were based on the Brazilian official statistics for the period from 1988 to 2007 (IBAMA, 2017). Particularly for shrimp species which are caught exclusively with bottom trawl, additional information of logbooks for the period of 2008-2014 were also collected from vessel owners and intermediaries of the shrimp fishery and used in this study for landing estimation.

Balancing and metrics of the Ecopath model

According to Heymans et al. (2016) and Link (2010), we analyzed the confidence of our model by observing a set of criteria and assumptions using the pre-balanced (PREBAL) diagnostics routine (Link, 2010). The EE values must be lower than 1. If this assumption was not reached, we adapted the diet matrix based on the literature and/or scientific advice. The production/consumption ratios (P/Q) is recommended to range from 0.1 to 0.3; the respiration/assimilation and respiration/production ratios need to be lower than 1.0. The respiration/biomass ratios must range between 1 and 10 for fish and 50 to 100 for groups with higher values of P/B and Q/B. A significant and negative relationship of the biomass, production and consumption with the trophic levels was also a required assumption for the model. Additionally, the pedigree index was calculated to quantify the uncertainty related to each input value (B, P/B, Q/B, diet and catch) in the model (Christensen et al., 2005), ranging from 0 (low precision information) to 1 (data and parameters fully rooted in local data).

From the network analysis routine (Christensen and Pauly, 1992), based in theory of Ulanowicz (1986), the Ecopath model estimates several ecological attributes related to the maturity resilience, stability (sensu Odum, 1969) and dynamics of the ecosystem. Some of these attributes were selected based on Christensen (1995) to explain the ecosystem bioenergetics, community structure, system recycling and balance (Gubiani et al., 2011). To analyze the direct and indirect impacts that a group has on other groups of the system, we performed the Mixed Trophic Impact (MTI) analysis (Ulanowicz and Puccia, 1990), which allows, together with the approach developed by Valls et al. (2015), the identification of key groups quantified by the Keystonness indexes (Power et al., 1996; Libralato et al., 2006; Valls et al., 2015) .

Trophic level Comparison between Ecopath and nitrogen stable isotope

The Stable Isotope Analysis (SIA) was performed on 21 species/groups, including macroalgae (1), zooplankton (1), crustaceans (4), mollusks (1) and fishes (14). Isotopes data, processing and analysis are detailed in supplementary material V. In order to validate the BSIR model, we compared and evaluate the trophic level (TL) estimated from Ecopath model with nitrogen composition ($\delta^{15}N$) through linear regression analysis (Navarro et al., 2011; Deehr et al., 2014) tested by the Pearson correlation coefficient with a significance level of 5% (Zar, 2009). Statistical analyses were performed with the R software (Core Team, 2020).

Appendix 2

Taxonomic data

Table S1. Group name, taxonomic composition and trophic guilds of each compartment of the Barra of Sirinhaém Ecopath model (BSIR), Pernambuco, Northeast of Brazil.

	compartment	Family	Scientific name	Guilds
1	Macroalgae	-	-	Primary producer
2	Phytoplankton	-	-	Primary producer
3	Zooplankton	-	-	Filter-feeder
4	Polychaeta	-	-	Several guilds
5	Amphipoda	-	-	Omnivore
6	Blue crabs	Portunidae	*Callinectes ornatus* *Callinectes danae*	Zoobenthivore
7	Crabs	Leucosiidae Calappidae Pinnotheridae	*Persephona lichtensteinii* *Calappa sulcata* *Pinnixa* sp	Deposit-feeder
8	Isopoda	-	-	Detritivore
9	Pen.sub	Peneidae	*Penaeus subtilis*	Omnivore
10	Pen.sch	Peneidae	*Penaeus schmitti*	Detritivore
11	Stomatopoda	Squillidae	*Squilla* sp	
12	Xip.kro	Peneidae	*Xiphopenaeus kroyeri*	Zooplanktivore
13	Other crustaceans	Palaemonidae Sergestidae Lysmatidae	*Nematopalaemon schmitti* *Acetes* *Exhippolysmata oplophoroides*	Filter-feeder Zooplanktivore
14	Squids	Loliginidae	*Lolliguncula brevis* *Loligo pleii*	Piscivore/Zoobenthivore
15	Flatfish	Achiridae	*Trinectes paulistanus* *Achirus declivis*	Zoobenthivore
16	Anc.spi	Engraulidae	*Anchoa spinifer*	Piscivore/Zoobenthivore
17	Asp.lun	Ariidae	*Aspistor luniscutis*	Omnivore
18	Bag.mar	Ariidae	*Bagre marinus*	Zoobenthivore
19	Car.hip	Carangidae	*Caranx hippos*	Piscivore
20	Cet.ede	Engraulidae	*Cetengraulis edentulus*	Zooplanktivore
21	Chi.ble	Pristigasteridae	*Chirocentrodon bleekerianus*	Zooplanktivore
22	Con.nob	Haemulidae	*Conodon nobilis*	Piscivore/Zoobenthivore
23	Cyn.vir	Sciaenidae	*Cynoscion virescens*	Zoobenthivore
24	Dia.sp	Gerreidae	*Diapterus auratus* *Diapterus rhombeus*	Zoobenthivore
25	Euc.sp	Gerreidae	*Eucinostomus argenteus* *Eucinostomus gula*	Omnivore
26	Ham.cor	Haemulidae	*Haemulopsis corvinaeformis*	Zoobenthivore
27	Hyp.gut	Dasyatidae	*Hypanus guttatus*	Zoobenthivore
28	Iso.par	Sciaenidae	*Isopisthus parvipinnis*	Piscivore
29	Lar.bre	Sciaenidae	*Larimus breviceps*	Zoobenthivore
30	Snappers	Lutjanidae	*Lutjanus analis* *Lutjanus synagris*	Piscivore/Zoobenthivore
31	Lyc.gro	Engraulidae	*Lycengraulis grossidens*	Piscivore
32	Mac.anc	Sciaenidae	*Macrodon ancylodon*	Piscivore
33	Met.ame	Sciaenidae	*Menticirrhus americanus*	Zoobenthivore
34	Mic.fur	Sciaenidae	*Micropogonias furnieri*	Omnivore
35	Neb.mic	Sciaenidae	*Nebris microps*	Zoobenthivore
36	Odo.muc	Pristigasteridae	*Odontognathus mucronatus*	Zooplanktivore
37	Oph.pun	Sciaenidae	*Ophioscion punctatissimus*	Zoobenthivore
38	Par.bra	Sciaenidae	*Paralonchurus brasiliensis*	Zoobenthivore
39	Pel.har	Pristigasteridae	*Pellona harroweri*	Zooplanktivore
40	Pol.vir	Polynemidae	*Polydactilus virginicus*	Zoobenthivore
41	Sph.gua	Sphyraenidae	*Sphyraena guachancho*	Piscivore
42	Ste.bra	Sciaenidae	*Stellifer brasiliensis*	Zoobenthivore
43	Ste.mic	Sciaenidae	*Stellifer microps*	Zoobenthivore
44	Ste.ras	Sciaenidae	*Stellifer rastrifer*	Zoobenthivore
45	Ste.ste	Sciaenidae	*Stellifer stellifer*	Zoobenthivore
46	Sym.tes	Cynoglossidae	*Symphurus tessellatus*	Zoobenthivore
47	Tri.lep	Trichiuridae	*Trichiurus lepturus*	Piscivore
48	Birds	Laridae	*Larus* sp	Piscivore
49	Seaturtles	Cheloniidae	*Caretta caretta* *Lepidochelys olivacea*	Piscivore/Zoobenthivore
50	Detritus	-	-	-

Source of parameters

Table S2. Input data and references by group for the Barra of Sirinhaém Ecopath model (BSIR), 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	Unit	Reference
1	Macroalgae			
	B	7.37	t. km^{-2}	(Soares and Fujii, 2012)
	P/B	13.25	year^{-1}	(Opitz, 1996)
	EE			Estimation from ecopath
2	Phytoplankton			
	B	2.2		(Mello, 2009; Silva, 2009)
	P/B	682		(Mello, 2009; Silva, 2009)
	EE			Estimation from ecopath
3	Zooplankton			
	B	3.48	t. km^{-2}	(Silva *et al.*, 2016b)
	P/B	50.21	year^{-1}	(Albouy *et al.*, 2010; Angelini and Vaz-Velho, 2011)
	Q/B	150.65	year^{-1}	(Albouy *et al.*, 2010; Angelini and Vaz-Velho, 2011)
	EE			Estimation from ecopath
	Diet			(Kleppel *et al.*, 1996; Schwamborn, 1997; Schnetzer and Steinberg, 2002))
4	Polychaeta			
	B		t. km^{-2}	Estimation from ecopath
	P/B	3.6	year^{-1}	(Rocha *et al.*, 2003, 2007)
	Q/B	25.52	year^{-1}	(Rocha *et al.*, 2003, 2007)
	EE	0.95		-
	Diet			(Checon *et al.*, 2017)
5	Amphipoda			
	B		t. km^{-2}	Estimation from ecopath
	P/B	6.64	year^{-1}	(Rocha *et al.*, 2003, 2007)
	Q/B	34.51	year^{-1}	(Rocha *et al.*, 2003, 2007)
	EE	0.95		-
	Diet			(Navarro-Barranco *et al.*, 2013)
6	Blue crabs			
	B		t. km^{-2}	Estimation from ecopath
	P/B	2	year^{-1}	(Walters *et al.*, 2008; Christensen *et al.*, 2009)
	Q/B	8	year^{-1}	(Walters *et al.*, 2008; Christensen *et al.*, 2009)
	EE	0.9		(Lira *et al.*, 2018)
	Diet			(Olinto Branco *et al.*, 2002)
7	Crabs			
	B		t. km^{-2}	Estimation from ecopath
	P/B	5.23	year^{-1}	(Freire *et al.*, 2008)
	Q/B	10.82	year^{-1}	(Freire *et al.*, 2008)
	EE	0.95		(Lira *et al.*, 2018)

	Diet			(Medina Mantelatto and Petracco, 1997)
8	Isopoda			
	B		t. km^{-2}	Estimation from ecopath
	P/B	13.75	year^{-1}	(Rocha *et al.*, 2003, 2007)
	Q/B	34.51	year^{-1}	(Rocha *et al.*, 2003, 2007)
	EE	0.95		-
	Diet			(Lopes-Leitzke *et al.*, 2011)
9	Pen.sub			
	B	0.208	t. km^{-2}	Estimates from our samples data
	P/B	5.25	year^{-1}	Estimates from our data
	Q/B	13.45	year^{-1}	(Opitz, 1996)
	EE			Estimation from ecopath
	Diet			(Albertoni *et al.*, 2003; Soares *et al.*, 2005)
10	Pen.sch			
	B	0.23	t. km^{-2}	Estimates from our samples data
	P/B	3.75	year^{-1}	Estimates from our data
	Q/B	13.45	year^{-1}	(Opitz, 1996)
	EE			Estimation from ecopath
	Diet			(Albertoni *et al.*, 2003; Soares *et al.*, 2005)
11	Stomatopoda			
	B		t. km^{-2}	Estimation from ecopath
	P/B	23.68	year^{-1}	(Arias-González *et al.*, 1997)
	Q/B	85.27	year^{-1}	(Arias-González *et al.*, 1997)
	EE	0.95		-
	Diet			(Opitz, 1996)
12	Xip.kro			
	B	1.53	t. km^{-2}	Estimates from our samples data
	P/B	10.4	year^{-1}	Estimates from our data
	Q/B	26	year^{-1}	(Opitz, 1996)
	EE			Estimation from ecopath
	Diet			(Branco and Junior, Moritz, 2001)
13	Other crustaceans			
	B		t. km^{-2}	Estimation from ecopath
	P/B	5.8	year^{-1}	(Deehr *et al.*, 2014)
	Q/B	19.2	year^{-1}	(Deehr *et al.*, 2014)
	EE	0.95		-
	Diet			(Metillo *et al.*, 2016)
14	Squids			
	B	0.18	t. km^{-2}	(Freire *et al.*, 2008)
	P/B	6.4	year^{-1}	(Freire *et al.*, 2008)
	Q/B	36.5	year^{-1}	(Freire *et al.*, 2008)
	EE			Estimation from ecopath
	Diet			(Coelho *et al.*, 2010; Gasalla *et al.*, 2010)

15	*Flatfish*			
	B	0.087	t. km^{-2}	Estimates from our samples data
	P/B	3.07	year^{-1}	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	11.26	year^{-1}	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Corrêa and Uieda, 2007)
16	*Anc.spi*			
	B	0.012	t. km^{-2}	Estimates from our samples data
	P/B	2.68	year^{-1}	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	13.3	year^{-1}	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Lira *et al.*, 2018)
17	Asp.lun			
	B	0.042	t. km^{-2}	Estimates from our samples data
	P/B	2.27	year^{-1}	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	12.5	year^{-1}	(Lira *et al.*, 2018)
	EE			Estimation from ecopath
	Diet			(Denadai *et al.*, 2004)
18	Bag.mar			
	B	0.183	t. km^{-2}	Estimates from our samples data
	P/B	2.3	year^{-1}	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	8.49	year^{-1}	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			Estimates from our samples data
19	Car.hip			
	B	0,0001	t. km^{-2}	Estimates from our samples data
	P/B	0.46	year^{-1}	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	6.66	year^{-1}	(Lira *et al.*, 2018)
	EE			Estimation from ecopath
	Diet			Estimates from our samples data
20	Cet.ede			
	B	0.072	t. km^{-2}	Estimates from our samples data
	P/B	2.29	year^{-1}	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	53.42	year^{-1}	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Sergipensel *et al.*, 1999; Krumme *et al.*, 2008)
21	Chi.ble			
	B	0.135	t. km^{-2}	Estimates from our samples data
	P/B	3.05	year^{-1}	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	20.19	year^{-1}	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Muto *et al.*, 2008)
22	Con.nob			

	B	0.164	t. Km^{-2}	Estimates from our samples data
	P/B	3.22	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	8.78	$year^{-1}$	(Lira *et al.*, 2018)
	EE			Estimation from ecopath
	Diet			(Lira *et al.*, 2019)
23	Cyn.vir			
	B	0.027	t. Km^{-2}	Estimates from our samples data
	P/B	2.53	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	5	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Lucena *et al.*, 2000)
24	Dia.sp			
	B	0.027	t. Km^{-2}	Estimates from our samples data
	P/B	2.9	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	10.61	$year^{-1}$	(Lira *et al.*, 2018)
	EE			Estimation from ecopath
	Diet			Estimates from our samples data
25	Euc.sp			
	B	0.042	t. Km^{-2}	Estimates from our samples data
	P/B	1.33	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	12.84	$year^{-1}$	(Lira *et al.*, 2018)
	EE			Estimation from ecopath
	Diet			Estimates from our samples data
26	Ham.cor			
	B	0.366	t. Km^{-2}	Estimates from our samples data
	P/B	2.48	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	11.19	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Regina Denadai *et al.*, 2013)
27	Hyp.gut			
	B	0.015	t. Km^{-2}	Estimates from our samples data
	P/B	0.35	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	2.68	$year^{-1}$	(Pauly, D. Christensen, V. & Sambilay, 1990)
	EE			Estimation from ecopath
	Diet			(Gianeti, 2011)
28	Iso.par			
	B	0.246	t. Km^{-2}	Estimates from our samples data
	P/B	1.93	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	8.13	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			
29	Lar.bre			
	B	0.275	t. Km^{-2}	Estimates from our samples data

	P/B	2.49	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	8.48	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			Estimates from our samples data
30	Snappers			
	B	0.006	t. Km^{-2}	Estimates from our samples data
	P/B	0.27	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	6.47	$year^{-1}$	(Lira *et al.*, 2018)
	EE			Estimation from ecopath
	Diet			(Fonseca, 2009; Costa, 2013)
31	Lyc.gro			
	B	0.068	t. Km^{-2}	Estimates from our samples data
	P/B	3.03	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	20.69	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Silva, 2012)
32	Mac.anc			
	B	0.051	t. Km^{-2}	Estimates from our samples data
	P/B	1.75	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	8.2	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Castro *et al.*, 2015)
33	Met.ame			
	B	0.14	t. Km^{-2}	Estimates from our samples data
	P/B	2.15	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	7.19	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			Estimates from our samples data
34	Mic.fur			
	B	0.162	t. Km^{-2}	Estimates from our samples data
	P/B	2.69	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	6.9	$year^{-1}$	(Lira *et al.*, 2018)
	EE			Estimation from ecopath
	Diet			(Denadai *et al.*, 2015)
35	Neb.mic			
	B	0.037	t. Km^{-2}	Estimates from our samples data
	P/B	1.44	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	8.5	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Chao, 1978)
36	Odo.muc			
	B	0.257	t. Km^{-2}	Estimates from our samples data
	P/B	4.58	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))

	Q/B	17.7	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Muto *et al.*, 2008)
37	Oph.pun			
	B	0.077	t. Km^{-2}	Estimates from our samples data
	P/B	1.93	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	10.88	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Zahorcsak *et al.*, 2000)
38	Par.bra			
	B	0.162	t. Km^{-2}	Estimates from our samples data
	P/B	3.89	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	8.7	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			Estimates from our samples data
39	Pel.har			
	B	0.783	t. Km^{-2}	Estimates from our samples data
	P/B	2.9	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	81.00	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Claudio Höfling *et al.*, 1998; Criales-Hernández, 2003; Muto *et al.*, 2008)
40	Pol.vir			
	B	0.083	t. Km^{-2}	Estimates from our samples data
	P/B	3.83	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	12.05	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Lopes and Oliveira-Silva, 1998)
41	Sph.gua			
	B	0.028	t. Km^{-2}	Estimates from our samples data
	P/B	0.49	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	4.65	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Akadje *et al.*, 2013)
42	Ste.bra			
	B	0.047	t. Km^{-2}	Estimates from our samples data
	P/B	2.19	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	12.9	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Sabinson *et al.*, 2015; Almeida, 2018)
43	Ste.mic			
	B	0.396	t. Km^{-2}	Estimates from our samples data
	P/B	5.47	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	11.07	$year^{-1}$	(Palomares and Pauly, 1998)

	EE			Estimation from ecopath
	Diet			Estimates from our samples data
44	Ste.ras			
	B	0.148	t. Km^{-2}	Estimates from our samples data
	P/B	3.56	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	8.09	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Sabinson *et al.*, 2015)
45	Ste.ste			
	B	0.094	t. Km^{-2}	Estimates from our samples data
	P/B	2.11	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	11.6	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Pombo, 2010)
46	Sym.tes			
	B	0.031	t. Km^{-2}	Estimates from our samples data
	P/B	1.27	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	10.51	$year^{-1}$	(Palomares and Pauly, 1998)
	EE			Estimation from ecopath
	Diet			(Guedes *et al.*, 2004)
47	Tri.lep			
	B	0.139	t. Km^{-2}	Estimates from our samples data
	P/B	1.68	$year^{-1}$	Estimates from our data (Z=P/B from Allen (1971))
	Q/B	3.62	$year^{-1}$	(Pauly, D. Christensen, V. & Sambilay, 1990)
	EE			Estimation from ecopath
	Diet			(Martins *et al.*, 2005)
48	Birds			
	B	0.015	t. Km^{-2}	(Opitz, 1996)
	P/B	5.4	$year^{-1}$	(Opitz, 1996)
	Q/B	80.00	$year^{-1}$	(Opitz, 1996)
	EE			Estimation from ecopath
	Diet			(Miotto *et al.*, 2017)
49	Seaturtles			
	B	0.003	t. Km^{-2}	(Guimarães *et al.*, 2018)
	P/B	0.15	$year^{-1}$	(Telles, 1998; Freire *et al.*, 2008)
	Q/B	22.00	$year^{-1}$	(Telles, 1998; Freire *et al.*, 2008)
	EE			Estimation from ecopath
	Diet			(Bugoni *et al.*, 2003; Colman *et al.*, 2014)

Estimate of the production/biomass (P/B)

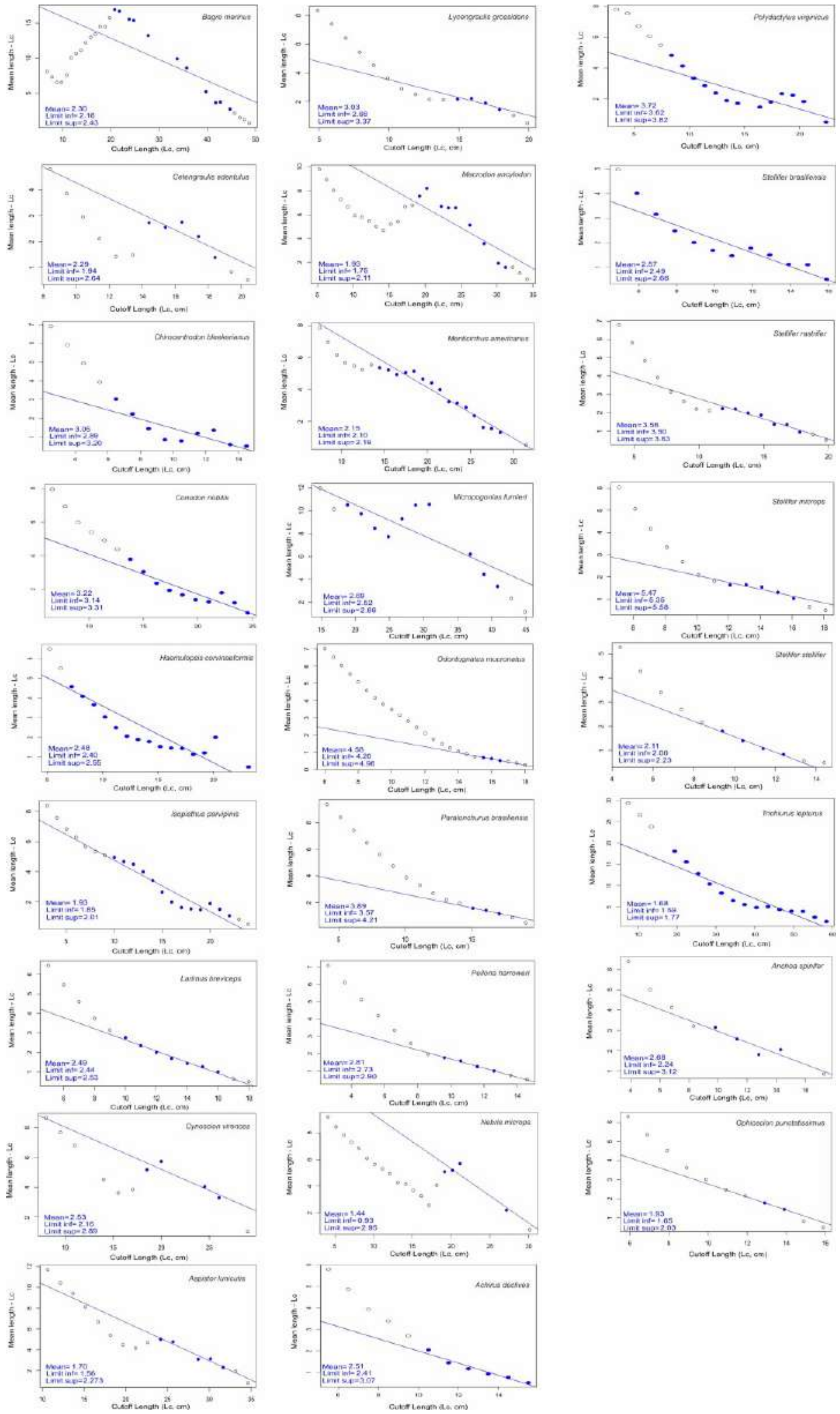


Figure S1. The Powell–Wetherall method is based on a linearizing transformation of size classes to estimate the total mortality (Z ± SE) (Pauly, 1983; Wetherall, 1986; Schwamborn, 2018) for the main compartments caught by local fishery.

Consumption information

Table S3. Parameters used as input for the estimation of the annual food consumption/biomass ratio (Q/B) of the 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
Flatfish	0.0102	3.25	86.74	0	0	19.58	394.08	1.08
Anchoa spinifer	0.005	3.18	253.9	0	0	22.18	111.24	2.12
Bagre marinus	0.0028	3.29	3987.09	0	0	46.43	838.93	2.71
Cetengraulis edentulus	0.004	2.72	31.62	1	0	22.08	528.55	0.93
Chirocentrodon bleekerianus	0.002	3.41	47.98	0	0	16.28	107.1	2.52
Cynoscion virescens	0.0108	2.86	11848.98	0	0	30.52	677.15	1.38
Haemulopsis corvinaeformis	0.0093	3.15	400.78	0	0	18.12	197.48	1.7
Isopisthus parvipinnis	0.0056	3.19	989.54	0	0	30.52	677.15	1.38
Larimus breviceps	0.0075	3.16	578.2	0	0	17.94	360.99	0.9
Lycengraulis grossidens	0.004	3.22	118.1	0	0	23.7	156.16	3.61
Macrodon ancylodon	0.0056	3.08	1390.2	0	0	16.28	107.1	1.38
Menticirrhus americanus	0.0045	*3.28*	1973.94	0	0	25.64	574.01	1.19
Nebris microps	0.0094	*3*	696.42	0	0	17.95	368	0.88
Odontognathus mucronatus	0.0281	*2.23*	90.1	0	0	16.28	107.1	2.52
Ophioscion punctatissimus	0.0062	*3.28*	293.98	0	0	21.06	371.98	1.22
Paralonchurus brasiliensis	0.0023	*3.47*	563.8	0	0	18.2	415.68	0.8
Pellona harroweri	0.0102	*3.02*	107.01	0	0	23.21	245.22	2.23
Polydactilus virginicus	0.0065	*3.13*	458.73	0	0	27.28	379.27	2.23
Sphyraena guachancho	0.0094	*2.76*	37024.21	0	0	22.17	256.49	1.94
Stellifer brasiliensis	0.0096	*3.03*	92.1	0	0	17.95	368	0.88
Stellifer microps	0.0058	*3.26*	196.38	0	0	17.95	368	0.88
Stellifer rastrifer	0.005	*3.36*	838.5	0	0	18.31	320.59	1.05
Stellifer stellifer	0.0059	*3.26*	49.98	0	0	17.95	368	0.88
Symphurus tessellatus	0.0237	*2.5*	166.31	0	0	2.98	21.47	0.43

Stable isotopes processing and results

Stable isotopes processing

White muscle samples (about 0.5g) from each fish, squid, blue crab and shrimp species were extracted (except for POM, SOM and zooplankton which whole organism/sample was analyzed), rinsed with distilled water to remove exogenous materials (e.g., remaining scales, bones and carapace), and dried in an oven at 60 °C for 48 h. Then, dried samples were ground into a fine powder with a mortar and pestle.

Nitrogen results reported as $\delta^{15}N$ values were measured using a mass spectrometer (Thermo Delta V+) coupled to an element analyzer (Thermo Flash, 2000; interface Thermo ConFio IV) at the Pôle de Spectrométrie Océan (PSO - IUEM, Plouzané, France). These values are derived from the relation between the isotopic value for the sample ($R_{sample:}$ $^{15}N/^{14}N$) and a known international standard ($R_{standard}$ ($\delta^{15}N$): atmospheric nitrogen):

$$\delta^{15}\mathrm{N} = \left[\left(\frac{R_{sample}}{R_{standard}}\right) - 1\right] \times 10^{3} (\text{eq.}1)$$

The analytical precision of the analysis monitored from a known standard (Thermo – Acétanilide) every six samples was defined as ±0.11‰ (standard error) and±0.07‰ for carbon and nitrogen, respectively.

Stable isotopes results

Guilds, Number of samples (n), isotopic means (±S.D.), minimum and maximum of nitrogen ($\delta^{15}N$) of basal sources and consumers (invertebrates and fishes) sampled in Sirinhaém coast, Northeastern Brazil.

Groups/species	Code	Guilds	n	δ15N (‰)	Min-Max
Basal sources					
Sargassum sp.	sar.sp	-	6	4.44 ±0.24	[4.07-4.73]
Invertebrates					
Zooplankton - Copepoda	zoo.cop	Filter-feeder	6	7.26 ±1.14	[6.45-9.49]
Callinectes ornatus	cal.orn	Zoobenthivore	3	9.27 ±0.86	[8.47-10.18]
Penaeus subtilis	pen.sub	Omnivore	16	8.83 ±2.19	[3.49-11.72]
Penaeus schmitti	pen.sch	Detritivore	22	8.98 ±1.51	[5.21-11.18]
Xiphopenaeus kroyeri	xip.kro	Zooplanktivore	23	9.49 ±0.56	[8.05-10.33]
Lolliguncula brevis	log.bre	Piscivore/Zoobenthivore	5	12.6 ±0.1	12.53-12.75]
Fishes					
Achirus lineatus	Ach.lin	Zoobenthivore	3	9.65 ±4.3	[5.06-13.6]
Bagre marinus	Bag.mar	Zoobenthivore	8	12.18 ±0.7	[11.33-13.47]
Caranx hippos	Car.hip	Piscivore	8	11.75 ±0.5	[10.36-10.7]
Chirocentrodon bleekerianus	Chi.ble	Zooplanktivore	4	10.59 ±0.8	[8.28-11.81]
Conodon nobilis	Con.nob	Piscivore/Zoobenthivore	4	12.71 ±1.5	11.45-14.94]
Diapterus auratus	Dia.aur	Zoobenthivore	7	8.84 ±1.23	[7.74-11.47]
Eucinostomus argenteus	Euc.arg	Omnivore	15	10.96 ±1.4	[6.49-13.19]
Isopisthus parvipinnis	Iso.par	Piscivore	4	12.5 ±0.19	[12.33-12.74]
Larimus breviceps	Lar.bre	Zoobenthivore	3	12.19 ±1	[11.18-13.18]
Lutjanus synagris	Lut.syn	Piscivore/Zoobenthivore	6	10.21 ±1.5	[8.71-11.76]
Paralonchurus brasiliensis	Par.bra	Zoobenthivore	3	12.89 ±1.6	[11.23-14.45]
Stellifer microps	Ste.mic	Zoobenthivore	4	12.21 ±1.6	[10.4-13.64]
Symphurus tessellatus	Sym.tes	Zoobenthivore	6	9.69 ±1.22	[8.71-11.86]
Bairdiella ronchus	Bai.ron	Zoobenthivore	3	10.54 ±0.10	[10.36-10.70]

Access to Ecopath model results

Ecopath model

Basic estimation

To balance the model, we adapted the predation rate from diet matrix for four trophic groups which initially presented EE > 1 (e.g., *Paralonchurus brasiliensis* - Par.bra, *Micropogonias furnieri* - Mic.fur, *Aspistor luniscutis* - Asp.lun and *Penaeus schmitti* - Pen.sch). The criteria and assumptions applied to evaluate the balance of the model, the production/consumption (P/Q), respiration/assimilation and respiration/biomass ratios reached the accepted ranges (see Table S4). Based on the PREBAL routine, the relations between B, P/B and Q/B had negative correlations with the trophic level (TL) (Figure S2).

High EE values were reported for some groups, due to the high predation (e.g., *Chirocentrodon bleekerianus* - Chi.ble, *Anchoa spinifer* - Anc.spi and *Stellifer brasiliensis* - Ste.bra) and others due to the fishing (e.g., *Xiphopenaeus kroyeri* - Xip.kro and *Macrodon ancylodon* - Mac.anc). However, the EE values of the *Conodon nobilis* - Con.nob, *Haemulopsis corvinaeformis* - Ham.cor, *Sphyraena guachancho* - Sph.gua, Birds and Seaturtles were considerably lower than those of other groups, since they are neither heavily predated nor fished (Table 3 in main text). Table S5 shows the final diet matrix used in the balanced model. The EE values of the groups targeted by fishing activities ranged between 0.04 and 0.99. The pedigree index for the SIR model was 0.65.

Relationship of Ecopath and stable nitrogen isotope values

The $\delta^{15}N$ mean values for 21 functional groups, obtained by Stable Isotopes Analysis ranged between 4.4 to 12.9‰, and were positively correlated with the Trophic Level estimated by the Ecopath model (Fig. 3, Person's correlation coefficient, cor = 0.87, *p-value* < 0.001). Except for squids, the highest $\delta^{15}N$ values were observed for fish species with TL > 3.1 (Fig. S3), while the shrimps had intermediate values (Pen.sch = 8.98‰, Pen.sub = 8.83‰ and Xip.kro = 9.49‰)

Food web structure and trophic analysis

Trophic structure

The omnivory of the functional groups, estimated by the Omnivory Index (OI) was overall low (0.05–0.55), except for Blue crabs and Seaturtles (OI = 0.69) (Table S4). The MTI included both direct and indirect impacts of all groups of the system. A positive (blue blocks) trophic interaction occurred between X. *kroyeri* and several groups (e.g., P. *brasiliensis* - Par.bra, *Menticirrhus americanus* – Met.ame and *Pellona harroweri* – Pel.har), as well as the positive impact in many groups by phytoplankton and zooplankton (Fig. S4). Conversely, negative impacts (red blocks) were observed for S. *guachancho* - Sph.gua on Snappers, blue crab on *Eucinostomus* sp. - Euc.sp, Squid on *Lycengraulis grossidens* - Lyc.gro and P. *brasiliensis* - Par.bra, as well as for Birds on *M. furnieri* - Mic.fur and *Stellifer rastrifer* - Ste.ras (Fig. S4). An increase in the capture rate in Trawling would cause relatively strong negative effects on the *Hypanus guttatus* - Hyp.gut, C. *nobilis* - Con.nob and *Penaeus schmitti* - Pen.sch. Similarly, *Trichiurus lepturus* - Tri.lep, S. *guachancho* - Sph.gua and M. ancylodon - Mac.anc are negativity affected by line fishing, while the biomass of their preys are positively influenced (Fig. S4).

Birds, Squid and X. *kroyeri* - Xip.kro were considered as the keystone species of the BSIR, presenting lower relative biomass and a higher impact in the food chain compared to other groups (Figure S5). Several groups had total impact values higher than 0.5, but they were not considered as keystone species, despite their importance for the transfer of energy from the base of the trophic chain to top predators (e.g., Stomatopoda, Blue crabs, Other crustaceans and P. *harroweri* - Pel.har).

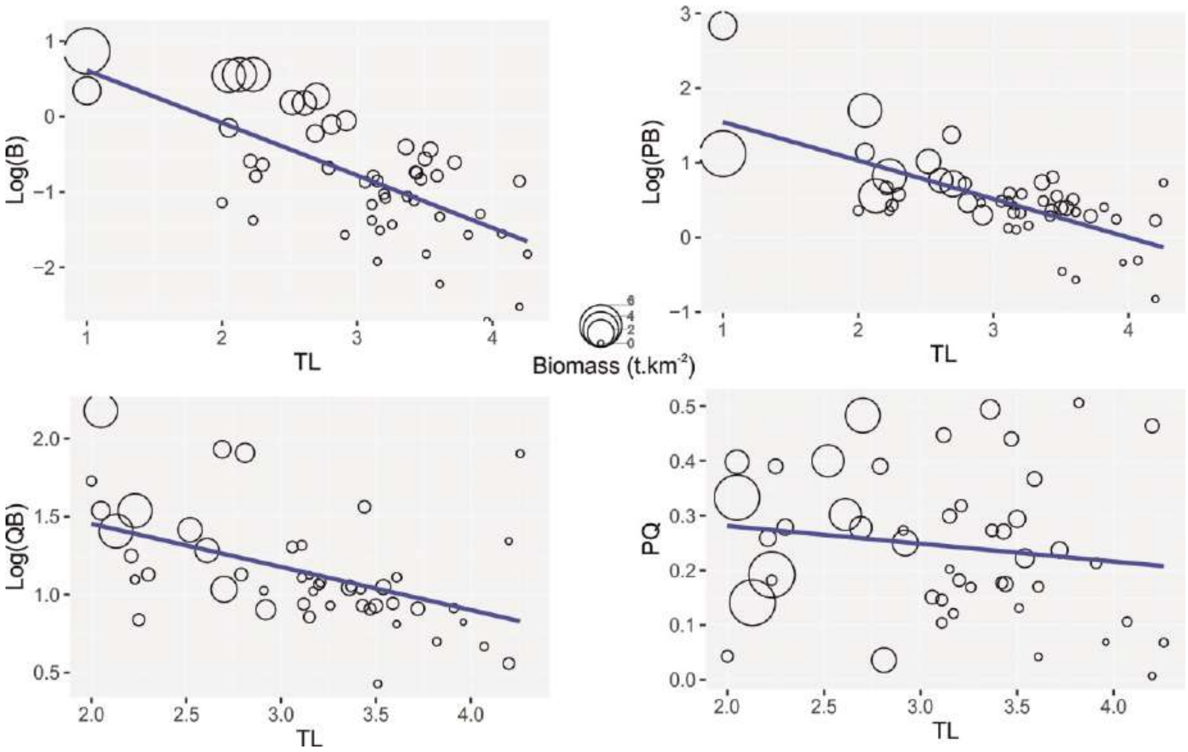
Balance of the model

Table S4. Omnivory index (OI), Production/consumption (P/Q) and respiration rates used for evaluating of the Barra of Sirinhaém Ecopath model (BSIR), Pernambuco, Northeast of Brazil.

Group name	OI	P/Q	Respiration/ assimilation	Respiration/ biomass
Mracoalgae	-	-	-	-
Phytoplankton	-	-	-	-
Zooplankton	0.052	0.333	0.445	40.18
Polychaeta	0.121	0.141	0.824	16.816
Amphipoda	0.187	0.192	0.759	20.968
Blue crabs	0.691	0.25	0.688	4.4
Crabs	0.552	0.483	0.396	3.426
Isopoda	0.052	0.398	0.502	13.858
Pen.sub	0.437	0.39	0.349	2.82
Pen.sch	0.233	0.279	0.535	4.32
Stomatopoda	0.418	0.278	0.653	44.536
Xip.kro	0.342	0.4	0.333	5.2
Other crustaceans	0.289	0.302	0.497	5.72
Squids	0.294	0.175	0.781	22.8
Flatfish	0.229	0.273	0.659	5.936
Anc.spi	0.065	0.202	0.748	7.96
Asp.lun	0.249	0.182	0.773	7.73
Bag.mar	0.420	0.271	0.661	4.492
Car.hip	0.284	0.069	0.914	4.868
Cet.ede	-	0.043	0.946	40.446
Chi.ble	0.272	0.151	0.811	13.102
Con.nob	0.307	0.367	0.542	3.804
Cyn.vir	0.470	0.506	0.368	1.47
Dia.sp	0.257	0.273	0.658	5.588
Euc.sp	0.166	0.104	0.871	8.942
Ham.cor	0.068	0.222	0.723	6.472
Hyp.gut	0.058	0.131	0.837	1.794
Iso.par	0.516	0.237	0.703	4.574
Lar.bre	0.276	0.294	0.633	4.294
Snappers	0.134	0.042	0.947	4.902
Lyc.gro	0.181	0.146	0.817	13.522
Mac.anc	0.272	0.213	0.733	4.81
Met.ame	0.494	0.299	0.626	3.602
Mic.fur	0.307	0.39	0.513	2.83
Neb.mic	0.228	0.169	0.788	5.36
Odo.muc	0.177	0.259	0.677	9.58
Oph.pun	0.079	0.177	0.778	6.774
Par.bra	0.454	0.447	0.441	3.07
Pel.har	0.399	0.036	0.955	61.9
Pol.vir	0.163	0.318	0.603	5.81
Sph.gua	0.273	0.106	0.868	3.228
Ste.bra	0.074	0.17	0.788	8.13
Ste.mic	0.417	0.494	0.382	3.386
Ste.ras	0.224	0.44	0.45	2.912
Ste.ste	0.056	0.182	0.773	7.17
Sym.tes	0.007	0.121	0.849	7.138
Tri.lep	0.313	0.464	0.42	1.216
Birds	0.178	0.068	0.916	58.6
Seaturtles	0.699	0.007	0.991	17.45

Balance of the model

Figure S2. Relation between the input data (B; PB; QB, and PQ) and trophic level (TL) obtained through of the PRE-BAL routine.



Diet matrix

Table S6. Final diet matrix applied for Barra of Sirinhaém Ecopath model (BSIR).

	Part I	Predator																							
	Prey	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26
1	Mracoalgae		0.09	0.3499	0.0458	0.0799	0.45	0.0036	0.15	0.012	0.15	0.06		0.0124	0.0018	0.016	0.035	0.0046			0.0014	0.0099			
2	Phytoplankton	0.8	0.409	0.0092	0.006	0.0689		0.05	0.14	0.11	0.07	0.16			0.0001				0.95						
3	Zooplankton	0.05	0.11	0.1999	0.0662	0.027	0.05	0.1357	0.201	0.15	0.028	0.55	0.106	0.0196	0.8198	0.08				0.4993	0.015	0.0049	0.3923	0.2269	0.06
4	Polychaeta		0.001	0.013	0.02	0.0147		0.2	0.08	0.06	0.12		0.0801	0.1637	0.0086	0.05	0.015						0.1471	0.4599	0.0086
5	Amphipoda		0.01		0.011	0.0596		0.0905		0.14	0.25		0.0917	0.1423		0.001				0.025		0.001	0.1746	0.024	0.16
6	Blue crabs				0.0679								0.0815				0.155	0.0115			0.0501	0.0198		0.024	0.034
7	Crabs				0.1696	0.0696				0.065		0.014	0.1019	0.0266	0.15	0.05	0.3999	0.0115			0.0016			0.042	0.0857
8	Isopoda				0.0139	0.1039		0.0044		0.075	0.04		0.0031		0.006	0.005	0.0001	0.0011		0.005	0.0042	0.001	0.0103		
9	Pen.sub									0.0008			0.0102					0.0006			0.0742	0.012			0.0493
10	Pen.sch					0.0006				0.0001			0.005	0.0114				0.0011			0.03	0.012		0.0003	0.0093
11	Stomatopoda				0.0778	0.1193		0.1809		0.05	0.002		0.1019	0.4489			0.01	0.0057			0.0789	0.0297	0.0098		0.0857
12	Xip.kro				0.0347	0.01				0.0092			0.06	0.0266			0.06	0.0023		0.059	0.3388	0.09	0.0294	0.094	0.36
13	Other crustaceans				0.0689	0.0696						0.005	0.0509	0.0152	0.0054		0.075	0.0001		0.2499	0.079	0.029	0.001	0.04	0.0671
14	Squids					0.001							0.0001					0.0229		0.002	0.0501	0.0099			
15	Flatfish				0.009								0.0051					0.0126				0.0198			
16	Anc.spi																	0.0023				0.0099			
17	Asp.lun																0.02	0.0138				0.0198			
18	Bag.mar																0.01	0.0115				0.0297			
19	Car.hip																	0.0273							
20	Cet.ede												0.0005					0.0023			0.0105				
21	Chi.ble													0.0038	0.0032		0.05	0.1719			0.0732	0.0693			0.0002
22	Con.nob				0.0007																	0.002			
23	Cyn.vir				0,0000													0.0011				0.0027			
24	Dia.sp				0.0021													0.0016				0.1287			
25	Euc.sp				0.002													0.0011							

Continue…

	Part II	Predator																						
	Prey	27	28	29	30	31	32	33	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48	49
1	Mracoalgae				0.0063	0.0001			0.0158		0.02				0.0202			0.001						
2	Phytoplankton					0.0085					0.65			0.06										
3	Zooplankton	0.1502	0.0201	0.0583	0.0415	0.5528		0.0326	0.0074	0.1489	0.2	0.222	0.005	0.3799	0.3731		0.0404	0.118	0.1174	0.7029	0.008			
4	Polychaeta	0.025			0.0006				0.039	0.062			0.09		0.0001		0.06		0.0005		0.6659			
5	Amphipoda	0.0005		0.0101	0.0501	0.0005			0.001	0.0869		0.09	0.02	0.02	0.1499		0.0122		0.0098		0.2914			
6	Blue crabs	0.02			0.023			0.0001				0.087	0.003				0.03							0.1199
7	Crabs	0.02	0.0135	0.0051	0.189	0.0213	0.0051	0.0217	0.021	0.1117			0.09		0.003		0.0969	0.19			0.0077			0.064
8	Isopoda	0.005	0.035	0.0061	0.0313	0.0106	0.001	0.0005	0.003	0.0496		0.045	0.003	0.02	0.001			0.0301		0.001				
9	Pen.sub			0.0344		0.012	0.02	0.055	0.003	0.0372		0.079	0.04		0.0276	0.0099	0.04	0.03		0.012				
10	Pen.sch	0.0451		0.0103	0.04	0.0213	0.0303	0.02	0.003	0.0448		0.045	0.05		0.0421	0.005	0.04	0.03		0.023		0.0019		0.001
11	Stomatopoda	0.0003	0.0503	0.0091	0.0667	0.0106		0.0163	0.021	0.062			0.02	0.07			0.2957	0.005	0.7429					0.005
12	Xip.kro	0.0701	0.1	0.2	0.2679	0.15	0.1712	0.5499	0.069	0.25		0.36	0.36	0.12	0.2329	0.0597	0.1999	0.25	0.0487	0.12	0.02	0.0954		0.01
13	Other crustaceans	0.6485	0.024	0.5021	0.06	0.072		0.01	0.005	0.05		0.072	0.059	0.04	0.09		0.135	0.08	0.0029	0.135	0.007	0.0107		0.005
14	Squids		0.052	0.081	0.045	0.0021	0.0202							0.0001		0.1682	0.05	0.0857				0.1863	0.047	0.08
15	Flatfish	0.0006			0.0104			0.007																
16	Anc.spi	0.002	0.009		0.0243		0.0036								0.005	0.014								
17	Asp.lun	0.003																				0.0049	0.0187	0.015
18	Bag.mar	0.004																				0.0049	0.1069	0.04
19	Car.hip																							
20	Cet.ede	0.0006			0.0208		0.0808									0.0697						0.0343	0.0156	0.01
21	Chi.ble		0.05	0.0004	0.0236		0.0801								0.005	0.0995				0.003		0.0873		0.01
22	Con.nob																							0.02
23	Cyn.vir																					0.0147	0.0156	0.4302
24	Dia.sp				0.0243		0.0051									0.01								
25	Euc.sp				0.0178		0.0001									0.0229								

Continue…

	Part III	Predator																							
	Prey	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26
26	Ham.cor																	0.0229				0.0495			
27	Hyp.gut																								
28	Iso.par																	0.0229				0.0396			
29	Lar.bre				0.0012													0.0344				0.0297			
30	Snappers																	0.004							
31	Lyc.gro				0.001								0.0153					0.0023			0.0005				
32	Mac.anc																	0.0034							
33	Met.ame				0.001													0.0172							
34	Mic.fur																	0.0172							
35	Neb.mic																	0.0229			0.0109				
36	Odo.muc				0.0105								0.046	0.019			0.05	0.1719			0.0157	0.0505			
37	Oph.pun																	0.0229			0.0128	0.0297			
38	Par.bra												0.0489					0.0331			0.0296	0.0201	0.0196		
39	Pel.har				0.0016	0.01							0.0901	0.0381				0.2509			0.0227	0.0594			0.0491
40	Pol.vir																				0.0052				
41	Sph.gua																	0.0034							
42	Ste.bra																				0.0052				
43	Ste.mic				0.016								0.0234					0.0146			0.0039	0.0247			0.031
44	Ste.ras													0.0038				0.0157							
45	Ste.ste																				0.0052	0.1188			
46	Sym.tes																	0.0011			0.0064				
47	Tri.lep																	0.0321							
48	Birds																								
49	Seaturtles																								
50	Detritus	0.15	0.38	0.4279	0.3733	0.3659	0.5	0.3349	0.429	0.328	0.34	0.211	0.0784	0.0685	0.0052	0.798	0.12		0.05	0.1598	0.075	0.077	0.2158	0.089	

Continue…

	Part IV	Predator																						
	Prey	27	28	29	30	31	32	33	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48	49
26	Ham.cor	0.003					0.01									0.05						0.0682	0.0156	0.02
27	Hyp.gut																							
28	Iso.par		0.03				0.07									0.057						0.038	0.0235	0.04
29	Lar.bre		0.0704																			0.0314	0.1022	
30	Snappers															0.0057								
31	Lyc.gro						0.0303															0.0584		
32	Mac.anc																					0.007		
33	Met.ame	0.0003	0.0317				0.037																0.0355	
34	Mic.fur																						0.0786	
35	Neb.mic						0.0202																0.0098	
36	Odo.muc		0.0987	0.0005	0.0139	0.0532	0.0808	0.0054							0.002	0.0796		0.02		0.003		0.0588	0.0059	0.01
37	Oph.pun		0.006													0.0199						0.0088	0.0156	
38	Par.bra		0.025				0.006									0.0382						0.0076	0.0844	
39	Pel.har	0.0009	0.14	0.0008	0.0271		0.0909	0.0022								0.0896		0.0201				0.0834	0.0187	0.03
40	Pol.vir		0.02													0.0786						0.0019		
41	Sph.gua															0.02								
42	Ste.bra			0.0008			0.0556	0.0084								0.0199						0.0304	0.0238	
43	Ste.mic		0.08				0.0504									0.0299						0.0343	0.1125	
44	Ste.ras		0.04				0.0606									0.01						0.0255	0.2503	
45	Ste.ste		0.0097	0.0009			0.0707									0.0129						0.0177		
46	Sym.tes	0.0003			0.0139			0.0109					0.005			0.0199								
47	Tri.lep	0.0005														0.01						0.0883		
48	Birds																							
49	Seaturtles																							
50	Detritus		0.0946	0.08	0.0026	0.085		0.26	0.8119	0.0968	0.13		0.255	0.2899	0.048			0.14	0.0779					0.09

EwE x Stable Isotope Analysis

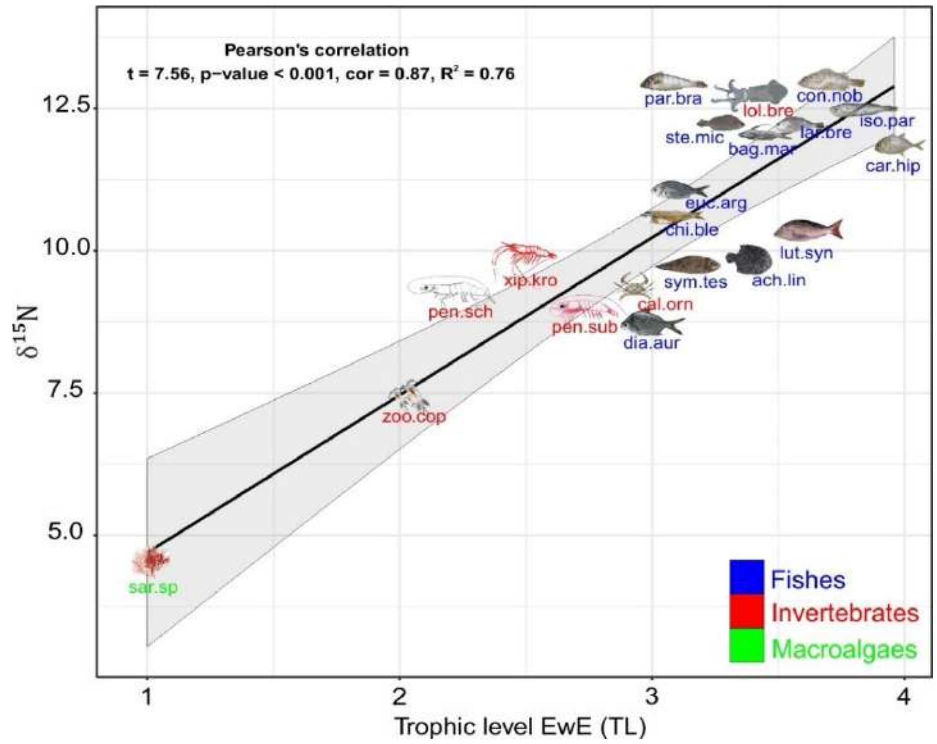


Figure S3. Correlation between the trophic level mean (TL) estimated from BSIR Ecopath model and the mean nitrogen composition ($\delta^{15}N$) for twenty species in the Barra of Sirinhaém, Pernambuco, Northeast of Brazil. The solid line is the regression line, and the gray area indicate the confidence interval 95%.

Mixed Trophic Impact

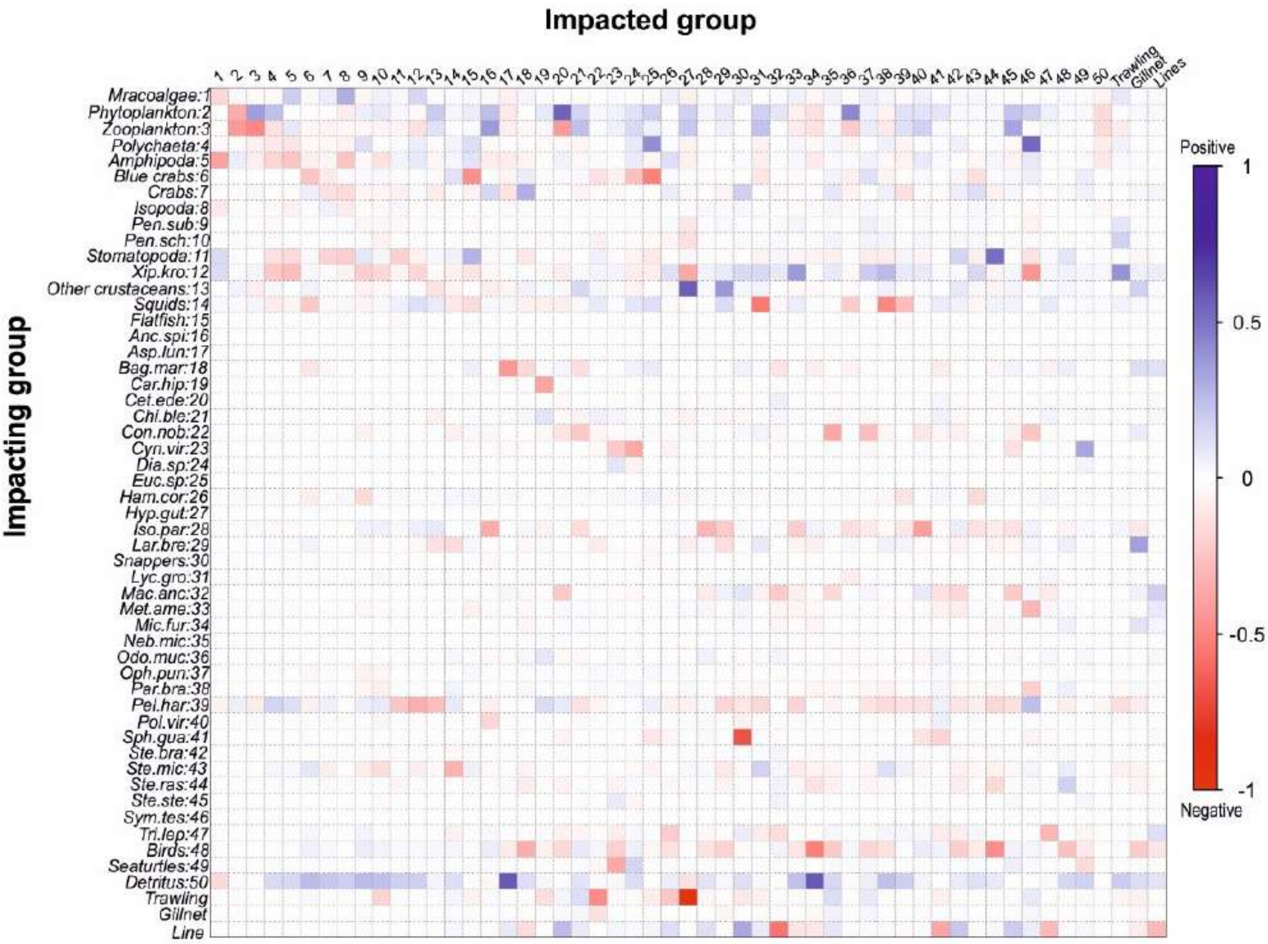


Figure S4. Mixed Trophic Impact of the Barra of Sirinhaém Ecopath model, Pernambuco, Northeast of Brazil. The color boxes indicate negative (red) or positive (blue) impacts. The color intensity is proportional to the degree of the impacts.

Key species of the model

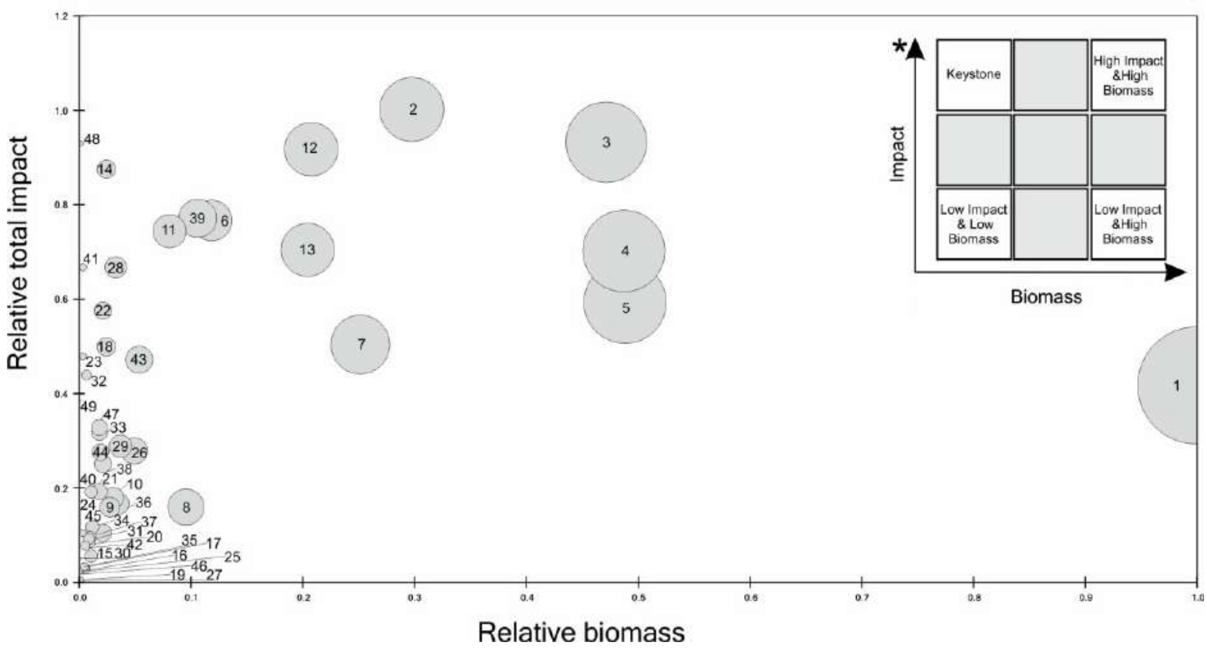


Figure S5. Relationship between relative total impact and relative biomass of each compartment for the Barra of Sirinhaém Ecopath model (BSIR), Pernambuco, Northeast of Brazil. Circle size is proportional to relative biomass for each group. *Conceptual identification of keystone species in food-web Valls et al. (2015). See table below to identify species/groups based in the numbers.

Primary production

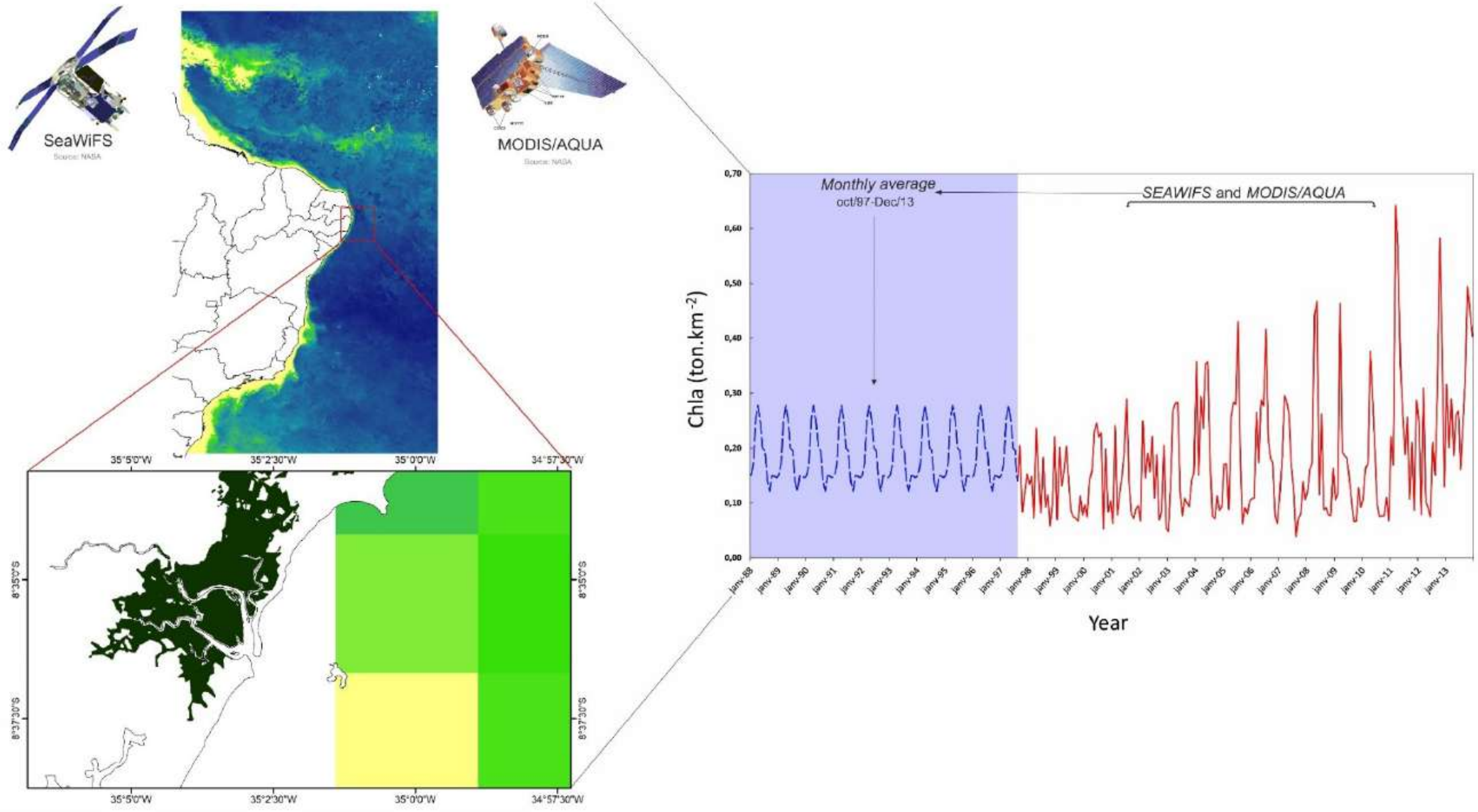


Figure S6. The primary production obtained from satellite image processed data (Level-3) of near-surface concentration of chlorophyll-a, for the period (Jan/97-Dec/13 from SEAWIFS and MODIS/AQUA) applied as forcing function in ECOSIM model.

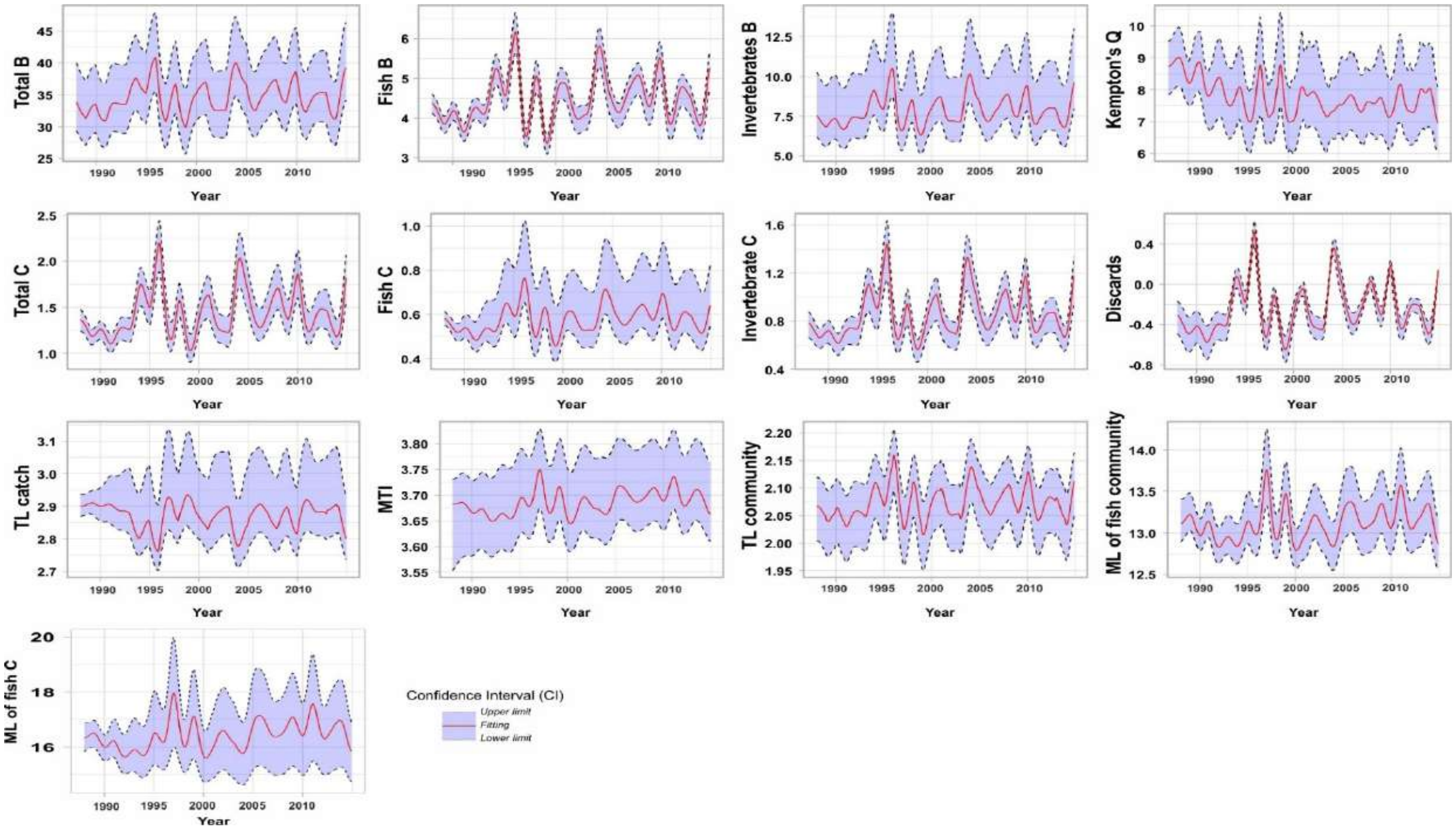
Vulnerability to Ecosim model

Table S6. Vulnerability values applied to provide the best fit for of the Barra of Sirinhaém Ecosim model (BSIR)

		Predator																																														
	Prey	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48	49
1	Mracoalgae		1	1	1	1	2	27	8	1	12	1		10	2	2	1	2			1	1							2	1			1		10			1	1			1						
2	Phytoplankton	1	1	2	1	1		27	6	1	13	1			2				10											1					10			1										
3	Zooplankton	1	1	1	1	1	2	28	6	1	10	1	3	10	2	2	1			1	1	1	1	1	1	10	5	1	2	1		5	1	2	10	2	7	1	1		1	1	2	1	2			
4	Polychaeta		1	2	1	1		70	6	1	8		3	10	2	2	1						1	1	1	10			2				1	2			13		1		1		2		2			
5	Amphipoda		1		1	1		28		1	9		3	10		2				1		1	1	1	1	10		1	2	1			1	2		2	10	1	1		1		2		2			
6	Blue crabs				1								3				1	2			1	1		1	1	10			2			5				2	10				1							10
7	Crabs				1	1				1		1	3	10	2	2	1	2			1			1	1	10	5	1	2	1	10	5	1	2			13		1		1	1			2			10
8	Isopoda				1	1		27		1	10		3		2	2	1	2		1	1	1	1			10	5	1	2	1	10	5	1	2		2	10	1	1			1		1				
9	Pen.sub									1			3					2			1	1			2			1		1	10	5	1	2		2	16	2	1	2	1	1		1				
10	Pen.sch					1				1			3	10				2			2	1		1	1	10		1	2	1	10	5	1	2		2	2	1	1	2	1	1		1		10		10
11	Stomatopoda				1	1		62		1	9		3	12			1	2			1	1	1		1	10	5	1	2	1		5	1	2			10	1			1	1	3					10
12	Xip.kro				1	1				1			3	10			1	2		1	1	1	1	1	1	10	4	1	2	1	10	1	1	2		2	9	1	1	2	1	1	2	1	2	10		10
13	Other crustaceans				1	1						1	3	10	2		1	2		1	1	1	1	1	1	10	5	1	2	1		5	1	2		2	10	1	1		1	1	2	1	2	10		10
14	Squids					1							3					2		1	1	1					45	1	2	1	10							1		2	1	1				48	5	10
15	Flatfish				1								3					2				1				10			2			5																
16	Anc.spi																	2				1				10	5	1	2		10								1	2								
17	Asp.lun																1	2				1				10																				10	5	10
18	Bag.mar																1	2				1				10																				10	5	10
19	Car.hip																	2																													5	
20	Cet.ede												3					2			1					10			2		1									2						10	5	10
21	Chi.ble													10	2		1	2			1	1			1	10	5	1	2		10								1	2				1		10		10
22	Con.nob				1																	1																										10
23	Cyn.vir																	2				1																								10	5	10
24	Dia.sp				1													2				1							2		10									2								
25	Euc.sp				1													2											2		10									2								
26	Ham.cor																	2				1				10					10									2						10	5	10
27	Hyp.gut																																															
28	Iso.par																	2				1					5				2									2						10	5	10
29	Lar.bre				1													2				1					5																			10	5	
30	Snappers																	2																						2								
31	Lyc.gro				1								3					2			1										10															10		
32	Mac.anc																	2																												10		
33	Met.ame				1	1												2								10	37				10																5	
34	Mic.fur																	2																													5	
35	Neb.mic																	2			1										10																5	
36	Odo.muc				1								4	10			1	2			1	1					5	1	2	1	10	5							1	2		1		1		10	5	10
37	Oph.pun																	2			2	1					50													2						10	5	
38	Par.bra												4					2			1	1	1				1				10									2						10	5	
39	Pel.har				1	1							3	10				2			1	1			1	10	5	1	2		10	5								2		1				10	5	10
40	Pol.vir																				1						5													2						10		
41	Sph.gua																	2																						2								
42	Ste.bra																				1							1			10	5								2						10	5	
43	Ste.mic				1								3					2			1	1			1		5				10									2						10	5	
44	Ste.ras													10				2									5				10									2						10	33	
45	Ste.ste																				1	1					5	1			10									2						10		
46	Sym.tes																	2			1					10			2			5					10			2								
47	Tri.lep																	2								10														2						10		
48	Birds																																															
49	Seaturtles																																															

Ecological indicators Ecosim model

Figure S7. Ecological indicators estimated from the Ecosim results for the period 1988–2014 for of the Barra of Sirinhaém Ecopath model (BSIR), Pernambuco, Northeast of Brazil. Total biomass - Total B (t·km^{-2}); biomass of fish and invertebrate - Fish B and Inver. B (t·km^{-2}); Kempton's biodiversity index (Q) - Kemp.Q; Total Catch - Total C (t·km^{-2}·year^{-1}); Catch of fish and invertebrate - Fish C and Inver. C (t·km^{-2}·year^{-1}); Total discarded catch – Disc (t·km^{-2}·year^{-1}); Tropic level (TL) of the catch and of the community (including all organisms) – mTLc and mTLco; Marine trophic index – MTI; Mean length of fish community and of fish catch – MLFco and MLFc (cm). The results are based on 1000 Ecosim model runs, obtained through the Monte Carlo routine, where the red line is fitting model and blue shadow represents the confidence interval 95%.



Biomass and catch ratios

Table S7. Mean of the biomass and catch ratios between 1988-2014 simulated to past and 1988-2030 to future. Red represents less (ratio < 0.99) and green gain (ratio > 1.01) biomass or catch.

S5a.Penaeus subtilis simulated to past

	Biomass ratio						Catch ratio					
	4 months	3 months	increase effort		reduce effort		4 months	3 months	increase effort		reduce effort	
Years	clo1s	clo2s	inc(+50%)	inc(+100%)	dec(-50%)	no_fishing	clo1s	clo2s	inc(+50%)	inc(+100%)	dec(-50%)	no_fishing
1988	1.0463	1.0392	0.9445	0.8922	1.0589	1.1216	0.6884	0.7664	1.4162	1.7844	0.5299	
1989	1.0861	1.0651	0.8873	0.784	1.1227	1.2557	0.7245	0.8035	1.3301	1.5674	0.562	
1990	1.1138	1.0848	0.8502	0.7163	1.1659	1.348	0.7238	0.7942	1.2745	1.4317	0.5837	
1991	1.1367	1.1027	0.8183	0.6598	1.2047	1.4319	0.7666	0.8381	1.2266	1.3186	0.6032	
1992	1.1608	1.1199	0.7901	0.611	1.2398	1.5084	0.7652	0.8319	1.1843	1.2211	0.6209	
1993	1.1812	1.1364	0.761	0.5619	1.2764	1.5875	0.8064	0.8766	1.1408	1.123	0.6393	
1994	1.1969	1.1465	0.7424	0.5285	1.2951	1.6215	0.7864	0.8428	1.1129	1.0564	0.6487	
1995	1.2013	1.1519	0.7263	0.4981	1.309	1.6443	0.8301	0.8999	1.0887	0.9955	0.6558	
1996	1.2064	1.1508	0.7237	0.4874	1.3023	1.6196	0.7376	0.8066	1.0849	0.9742	0.6525	
1997	1.2015	1.1512	0.7158	0.4701	1.3091	1.6324	0.8249	0.8995	1.073	0.9395	0.656	
1998	1.2133	1.1557	0.7117	0.4621	1.3136	1.64	0.7538	0.8167	1.0668	0.9236	0.6583	
1999	1.2214	1.1654	0.6926	0.4342	1.3386	1.694	0.8359	0.9059	1.0383	0.8677	0.6709	
2000	1.2329	1.1735	0.68	0.4131	1.3505	1.7146	0.8237	0.8796	1.0194	0.8257	0.677	
2001	1.2357	1.1742	0.6741	0.3997	1.3511	1.7114	0.802	0.8595	1.0105	0.7988	0.6773	
2002	1.2351	1.1746	0.6693	0.3878	1.352	1.7116	0.8134	0.8754	1.0033	0.7751	0.6779	
2003	1.2374	1.1788	0.6624	0.3751	1.362	1.7388	0.8497	0.9209	0.9929	0.7497	0.683	
2004	1.2386	1.178	0.6626	0.3709	1.3563	1.722	0.8155	0.8636	0.9933	0.7414	0.6802	
2005	1.2308	1.1709	0.6665	0.37	1.3424	1.6846	0.7972	0.8618	0.9992	0.7396	0.6733	
2006	1.2269	1.1701	0.6672	0.3665	1.3403	1.6832	0.8163	0.8841	1.0001	0.7325	0.6724	
2007	1.2293	1.1723	0.6657	0.3625	1.3455	1.7005	0.8283	0.8873	0.9979	0.7246	0.675	
2008	1.2294	1.1707	0.666	0.3596	1.3404	1.6846	0.7981	0.8561	0.9984	0.7188	0.6726	
2009	1.2247	1.1695	0.6654	0.3555	1.3401	1.6836	0.8351	0.9039	0.9974	0.7106	0.6713	
2010	1.2241	1.1658	0.6712	0.3592	1.3296	1.6598	0.7698	0.8324	1.0062	0.718	0.6651	
2011	1.2171	1.1629	0.6728	0.3584	1.3282	1.6597	0.8232	0.8896	1.0086	0.7164	0.6644	
2012	1.2245	1.1674	0.6707	0.3553	1.3358	1.6785	0.8061	0.8659	1.0054	0.7102	0.6682	
2013	1.2276	1.1687	0.6687	0.3516	1.3399	1.6887	0.7966	0.8564	1.0025	0.7027	0.6702	
2014	1.226	1.1725	0.663	0.3437	1.3508	1.714	0.9663	0.9168	0.9944	0.6872	0.6757	

S5b.Penaeus schimit simulated to past

	Biomass ratio						Catch ratio					
	4 months	3 months	increase effort		reduce effort		4 months	3 months	increase effort		reduce effort	
Years	clo1s	clo2s	inc(+50%)	inc(+100%)	dec(-50%)	no_fishing	clo1s	clo2s	inc(+50%)	inc(+100%)	dec(-50%)	no_fishing
1988	1.095	1.0768	0.8994	0.8105	1.1138	1.2426	0.7242	0.7974	1.3486	1.6209	0.5574	
1989	1.2029	1.1541	0.7643	0.5786	1.2946	1.6558	0.7995	0.8656	1.1457	1.1566	0.6481	
1990	1.2823	1.2093	0.678	0.4459	1.4247	1.9547	0.8404	0.8918	1.0163	0.8912	0.7133	
1991	1.3385	1.2511	0.6143	0.3554	1.5215	2.1604	0.8935	0.9428	0.9208	0.7103	0.7619	
1992	1.381	1.2819	0.5668	0.2919	1.5873	2.2802	0.92	0.9595	0.8497	0.5834	0.7949	
1993	1.3999	1.2983	0.5323	0.2458	1.6184	2.3081	0.9481	0.9939	0.798	0.4913	0.8106	
1994	1.3994	1.2974	0.5144	0.2172	1.6058	2.2369	0.9312	0.9656	0.7712	0.4341	0.8044	
1995	1.3661	1.277	0.513	0.2019	1.5535	2.0897	0.9295	0.9812	0.769	0.4036	0.7783	
1996	1.3441	1.255	0.5219	0.1968	1.5019	1.9725	0.8483	0.8971	0.7824	0.3934	0.7525	
1997	1.3102	1.2343	0.5347	0.1957	1.4611	1.8924	0.8664	0.9347	0.8016	0.3912	0.7321	
1998	1.3298	1.2448	0.5257	0.1831	1.4853	1.9625	0.8536	0.8995	0.788	0.3661	0.7444	
1999	1.3448	1.2587	0.5093	0.1662	1.5195	2.0435	0.8916	0.9523	0.7635	0.3322	0.7616	
2000	1.3716	1.2796	0.4863	0.1455	1.5633	2.1406	0.9278	0.9709	0.729	0.2907	0.7837	
2001	1.3769	1.2808	0.4768	0.1325	1.5615	2.1178	0.8974	0.9415	0.7147	0.2648	0.7828	
2002	1.3628	1.2718	0.4755	0.124	1.5376	2.0478	0.8909	0.9414	0.7127	0.2478	0.771	
2003	1.3619	1.2753	0.4679	0.1136	1.5463	2.0735	0.9286	0.9862	0.7014	0.227	0.7754	
2004	1.3642	1.2744	0.4654	0.1064	1.5424	2.0656	0.9112	0.9494	0.6976	0.2126	0.7736	
2005	1.332	1.2486	0.4821	0.107	1.4837	1.921	0.8566	0.9122	0.7226	0.2138	0.7442	
2006	1.3109	1.2355	0.4939	0.1064	1.4567	1.8674	0.8635	0.925	0.7404	0.2127	0.7308	
2007	1.3163	1.2404	0.4928	0.1018	1.4699	1.9097	0.8846	0.9378	0.7387	0.2034	0.7375	
2008	1.3173	1.2384	0.4955	0.0987	1.4657	1.8999	0.86	0.9102	0.7428	0.1973	0.7355	
2009	1.3032	1.231	0.5026	0.0972	1.4503	1.8622	0.8756	0.9372	0.7534	0.1943	0.7265	
2010	1.3053	1.2284	0.5075	0.0957	1.4455	1.855	0.8401	0.8919	0.7608	0.1912	0.723	
2011	1.2923	1.2209	0.5158	0.0954	1.4315	1.8235	0.8515	0.915	0.7732	0.1907	0.716	
2012	1.3026	1.2289	0.5112	0.0912	1.4488	1.8678	0.8661	0.9187	0.7663	0.1822	0.7247	
2013	1.313	1.2339	0.5069	0.0871	1.4602	1.8927	0.8485	0.9024	0.7598	0.1741	0.7304	
2014	1.3096	1.2394	0.4999	0.082	1.4728	1.9225	1.0112	0.9545	0.7498	0.164	0.7367	

S5c.Xiphopenaeus kroyeri simulated to past

	Biomass ratio						Catch ratio					
	4 months	3 months	increase effort		reduce effort		4 months	3 months	increase effort		reduce effort	
Years	clo1s	clo2s	inc(+50%)	inc(+100%)	dec(-50%)	no_fishing	clo1s	clo2s	inc(+50%)	inc(+100%)	dec(-50%)	no_fishing
1988	1.0313	1.0294	0.9537	0.9085	1.0475	1.0961	0.6746	0.7565	1.4299	1.8169	0.5242	
1989	1.035	1.0267	0.9422	0.8814	1.0545	1.1056	0.6878	0.7727	1.4124	1.7619	0.5279	
1990	1.0235	1.0169	0.957	0.9066	1.0358	1.0651	0.6649	0.7463	1.4346	1.8121	0.5186	
1991	1.0126	1.0097	0.9698	0.9298	1.0214	1.0361	0.6805	0.7659	1.4538	1.8584	0.5114	
1992	1.0061	1.0049	0.9821	0.9532	1.009	1.0127	0.6639	0.7467	1.4723	1.9051	0.5053	
1993	0.9997	1.0011	0.9883	0.9648	1.0031	1.0025	0.6839	0.7742	1.4816	1.9283	0.5024	
1994	0.997	0.9974	0.9999	0.9872	0.9926	0.9833	0.6397	0.7222	1.4989	1.9732	0.4972	
1995	0.9896	0.9934	1.0056	0.9974	0.9889	0.979	0.6871	0.7748	1.5075	1.9936	0.4954	
1996	0.9928	0.9908	1.0186	1.0247	0.9802	0.9645	0.5727	0.6884	1.527	2.0481	0.4911	
1997	0.9858	0.9889	1.0215	1.032	0.9835	0.9754	0.6682	0.7685	1.5313	2.0627	0.4928	
1998	0.992	0.9922	1.0244	1.047	0.9842	0.9753	0.6159	0.7043	1.5356	2.0926	0.4932	
1999	0.9958	0.9971	1.007	1.0158	0.9977	0.9976	0.675	0.7729	1.5095	2.0303	0.5001	
2000	0.9958	0.9984	1.0016	0.9997	0.9964	0.9906	0.6757	0.7549	1.5013	1.998	0.4995	
2001	0.998	0.9972	1.0042	1.0004	0.9923	0.9832	0.628	0.7206	1.5053	1.9995	0.4975	
2002	0.994	0.995	1.0056	0.9978	0.9896	0.9782	0.6529	0.7398	1.5074	1.9943	0.4962	
2003	0.9874	0.9925	1.0093	1.0052	0.988	0.977	0.6887	0.7879	1.513	2.009	0.4955	
2004	0.9918	0.9934	1.0146	1.0194	0.9852	0.9718	0.6331	0.714	1.5209	2.0375	0.4941	
2005	0.9926	0.9922	1.0173	1.0246	0.984	0.9698	0.6268	0.7207	1.5249	2.0479	0.4936	
2006	0.9872	0.9903	1.0207	1.0315	0.9828	0.9687	0.6569	0.7513	1.5301	2.0616	0.493	
2007	0.9877	0.9917	1.02	1.0348	0.9852	0.9738	0.6696	0.7528	1.529	2.0682	0.4943	
2008	0.9931	0.9937	1.0159	1.0272	0.986	0.972	0.6284	0.7169	1.5228	2.053	0.4947	
2009	0.9901	0.9935	1.012	1.0186	0.9892	0.9776	0.6793	0.7721	1.517	2.0358	0.4955	
2010	0.993	0.9927	1.0186	1.0328	0.9838	0.9685	0.6056	0.7033	1.5268	2.0643	0.4921	
2011	0.989	0.9915	1.0169	1.0288	0.985	0.9709	0.6631	0.7542	1.5244	2.0563	0.4927	
2012	0.99	0.993	1.0162	1.0297	0.9858	0.972	0.6572	0.7404	1.5233	2.0582	0.4931	
2013	0.9939	0.9943	1.0127	1.0219	0.9872	0.9736	0.6349	0.7247	1.518	2.0425	0.4938	
2014	0.987	0.9943	1.0061	1.0062	0.9908	0.9801	0.7918	0.7883	1.5091	2.0122	0.4956	

S5d.Penaeus subtilis simulated to future (Biomass)

	Biomass ratio												
	4 months	3 months	increase effort				reduce effort				PP changes		
Years	clo1s	clo2s	inc(+10%)	inc(+25%)	inc(+50%)	inc(+100%)	dec(-10%)	dec(-25%)	dec(-50%)	no_fishing(dec(-100%)	env3	env4	env5
1988 to 2014	Baseline (effort constat)												
2015	1.0455	1.0393	0.9778	0.9449	0.892	0.7925	1.0231	1.0572	1.1172	1.2426	1.0279	0.9988	0.9602
2016	1.0837	1.064	0.9695	0.9248	0.8532	0.7217	1.0318	1.0789	1.1625	1.339	1.0113	0.9465	0.8513
2017	1.1093	1.0827	0.964	0.9111	0.8269	0.6734	1.0375	1.0932	1.1919	1.399	0.9968	0.9092	0.7831
2018	1.1297	1.0977	0.9584	0.8976	0.801	0.6271	1.0434	1.1079	1.2224	1.4641	0.9828	0.8751	0.723
2019	1.1488	1.1118	0.9537	0.886	0.7788	0.5874	1.0483	1.1202	1.2476	1.5157	0.9696	0.8465	0.678
2020	1.165	1.1237	0.9496	0.8759	0.7594	0.5528	1.0527	1.131	1.2697	1.5609	0.9572	0.8191	0.6342
2021	1.1786	1.1337	0.9462	0.8676	0.7432	0.5238	1.0562	1.1397	1.2871	1.5946	0.946	0.7945	0.5945
2022	1.1901	1.1422	0.9433	0.8604	0.7292	0.4986	1.0592	1.1471	1.3019	1.6226	0.9357	0.7721	0.5581
2023	1.1996	1.1492	0.9409	0.8542	0.7171	0.4767	1.0617	1.1533	1.314	1.6447	0.9262	0.752	0.5251
2024	1.2074	1.1551	0.9388	0.849	0.7067	0.4577	1.0638	1.1585	1.3238	1.6621	0.9176	0.7343	0.496
2025	1.214	1.16	0.937	0.8444	0.6976	0.4408	1.0656	1.1628	1.332	1.6765	0.9096	0.7185	0.4697
2026	1.2194	1.164	0.9355	0.8405	0.6898	0.4259	1.0671	1.1665	1.3387	1.6879	0.9022	0.7045	0.4464
2027	1.2239	1.1674	0.9341	0.8372	0.6829	0.4126	1.0684	1.1696	1.3443	1.6976	0.8953	0.6921	0.4254
2028	1.2276	1.1702	0.933	0.8343	0.6769	0.4007	1.0695	1.1721	1.3489	1.7054	0.8888	0.6809	0.4066
2029	1.2307	1.1726	0.9321	0.8318	0.6716	0.39	1.0704	1.1743	1.3527	1.7121	0.8827	0.6709	0.3896
2030	1.2308	1.1745	0.9313	0.8297	0.6671	0.3804	1.0711	1.1761	1.3558	1.7177	0.877	0.6619	0.3742

S5e.Penaeus subtilis simulated to future (Catch)

	Catch ratio												
	4 months	3 months	increase effort				reduce effort				PP changes		
Years	clo1s	Clo2s	inc(+10%)	inc(+25%)	inc(+50%)	inc(+100%)	dec(-10%)	dec(-25%)	dec(-50%)	no_fishing(dec(-100%)	env3	env4	env5
1988 to 2014	Baseline (effort constat)												
2015	0.6822	0.7615	1.0751	1.1807	1.3374	1.5845	0.9189	0.794	0.5572		1.0279	0.9988	0.9602
2016	0.7217	0.8019	1.066	1.1555	1.2793	1.4428	0.9267	0.8102	0.5798		1.0113	0.9465	0.8513
2017	0.7327	0.8046	1.06	1.1385	1.2398	1.3462	0.9319	0.8208	0.5945		0.9968	0.9092	0.7831
2018	0.7493	0.8222	1.0539	1.1215	1.201	1.2537	0.9372	0.8317	0.6097		0.9828	0.8751	0.723
2019	0.7608	0.8302	1.0487	1.1071	1.1677	1.1743	0.9416	0.8408	0.6223		0.9696	0.8465	0.678
2020	0.772	0.8399	1.0441	1.0944	1.1386	1.1052	0.9455	0.8488	0.6333		0.9572	0.8191	0.6342
2021	0.781	0.8471	1.0404	1.084	1.1144	1.0473	0.9486	0.8551	0.642		0.946	0.7945	0.5945
2022	0.7887	0.8534	1.0372	1.075	1.0933	0.9967	0.9513	0.8606	0.6493		0.9357	0.7721	0.5581
2023	0.795	0.8587	1.0345	1.0673	1.0752	0.953	0.9536	0.8651	0.6554		0.9262	0.752	0.5251
2024	0.8002	0.863	1.0322	1.0608	1.0597	0.915	0.9555	0.8688	0.6603		0.9176	0.7343	0.496
2025	0.8046	0.8667	1.0302	1.0551	1.046	0.8812	0.9571	0.8719	0.6644		0.9096	0.7185	0.4697
2026	0.8082	0.8697	1.0286	1.0502	1.0342	0.8515	0.9585	0.8745	0.6677		0.9022	0.7045	0.4464
2027	0.8112	0.8722	1.0271	1.046	1.0239	0.8248	0.9596	0.8767	0.6705		0.8953	0.6921	0.4254
2028	0.8136	0.8743	1.0259	1.0425	1.0149	0.8011	0.9606	0.8784	0.6728		0.8888	0.6809	0.4066
2029	0.8157	0.876	1.0249	1.0394	1.007	0.7797	0.9614	0.8799	0.6747		0.8827	0.6709	0.3896
2030	0.8174	0.8774	1.0241	1.0368	1.0003	0.7608	0.9627	0.8813	0.6768		0.877	0.6619	0.3742

S5f.Penaeus schimtti simulated to future (Biomass)

	Biomass ratio												
	4 months	3 months	increase effort				reduce effort				PP changes		
Years	clo1s	clo2s	inc(+10%)	inc(+25%)	inc(+50%)	inc(+100%)	dec(-10%)	dec(-25%)	dec(-50%)	no_fishing(dec(-100%)	env3	env4	env5
1988 to 2014	Baseline (effort constat)												
2015	1.096	1.079	0.9483	0.8748	0.7632	0.5769	1.0552	1.1399	1.2971	1.6592	0.9903	0.9645	0.9288
2016	1.2015	1.1534	0.929	0.8291	0.6812	0.449	1.0765	1.1945	1.4154	1.9201	0.9685	0.8971	0.7931
2017	1.2716	1.203	0.9163	0.799	0.6272	0.3684	1.0901	1.2286	1.4853	2.0502	0.9556	0.8539	0.7069
2018	1.3152	1.2341	0.9077	0.7781	0.5889	0.3124	1.0991	1.2501	1.5256	2.1097	0.9455	0.8224	0.6458
2019	1.3418	1.2537	0.9019	0.7634	0.5608	0.2712	1.1047	1.2631	1.5468	2.1298	0.9388	0.8026	0.6073
2020	1.3557	1.2644	0.8982	0.7536	0.5408	0.2407	1.1079	1.2695	1.5544	2.1261	0.934	0.7898	0.5813
2021	1.3619	1.2695	0.8961	0.7472	0.5264	0.2173	1.1094	1.272	1.5545	2.1128	0.9309	0.7824	0.564
2022	1.3633	1.2711	0.895	0.7433	0.5161	0.199	1.1099	1.272	1.5507	2.0958	0.9287	0.7786	0.553
2023	1.3623	1.2709	0.8945	0.7409	0.5087	0.1842	1.1098	1.2709	1.5454	2.0793	0.9271	0.7772	0.546
2024	1.3599	1.2695	0.8944	0.7398	0.5035	0.1722	1.1093	1.269	1.5393	2.0632	0.9259	0.7775	0.5424
2025	1.357	1.2676	0.8946	0.7394	0.4997	0.1621	1.1086	1.2669	1.5334	2.0486	0.9248	0.7785	0.5409
2026	1.3539	1.2655	0.895	0.7395	0.4971	0.1536	1.1079	1.2648	1.5277	2.035	0.9237	0.7801	0.5409
2027	1.3508	1.2634	0.8955	0.7399	0.4953	0.1461	1.1071	1.2627	1.5224	2.0225	0.9225	0.7817	0.5419
2028	1.3478	1.2612	0.896	0.7406	0.4942	0.1396	1.1063	1.2606	1.5173	2.0108	0.9213	0.7832	0.5434
2029	1.3448	1.2592	0.8965	0.7414	0.4935	0.1339	1.1056	1.2587	1.5126	1.9998	0.92	0.7846	0.5452
2030	1.3379	1.2572	0.8971	0.7423	0.4933	0.1287	1.1048	1.2568	1.508	1.9892	0.9186	0.7858	0.5472

S5g.Penaeus schimtti simulated to future (Catch)

	Catch ratio												
	4 months	3 months	increase effort				reduce effort				PP changes		
Years	clo1s	clo2s	inc(+10%)	inc(+25%)	inc(+50%)	inc(+100%)	dec(-10%)	dec(-25%)	dec(-50%)	no_fishing(dec(-100%)	env3	env4	env5
1988 to 2014	Baseline (effort constat)												
2015	0.7333	0.8053	1.0427	1.093	1.1443	0.9756	0.9477	0.8561	0.6469		0.9903	0.9645	0.9288
2016	0.7995	0.864	1.0214	1.036	1.0214	0.9854	0.9669	0.897	0.7059		0.9685	0.8971	0.7931
2017	0.844	0.8979	1.0076	0.9984	0.9404	0.9917	0.9791	0.9225	0.7408		0.9556	0.8539	0.7069
2018	0.8723	0.9221	0.9981	0.9722	0.8829	0.9959	0.9872	0.9385	0.7609		0.9455	0.8224	0.6458
2019	0.8897	0.9361	0.9917	0.9538	0.8409	0.9986	0.9923	0.948	0.7715		0.9388	0.8026	0.6073
2020	0.8979	0.9436	0.9877	0.9416	0.8109	1.0002	0.9951	0.9527	0.7753		0.934	0.7898	0.5813
2021	0.9018	0.9473	0.9853	0.9336	0.7893	1.001	0.9964	0.9544	0.7753		0.9309	0.7824	0.564
2022	0.9024	0.9482	0.9841	0.9287	0.7739	1.0013	0.9969	0.9543	0.7734		0.9287	0.7786	0.553
2023	0.9016	0.948	0.9835	0.9258	0.7627	1.0013	0.9968	0.9533	0.7708		0.9271	0.7772	0.546
2024	0.8999	0.9468	0.9835	0.9244	0.7549	1.0012	0.9963	0.9517	0.7677		0.9259	0.7775	0.5424
2025	0.8979	0.9454	0.9837	0.9238	0.7492	1.0009	0.9957	0.95	0.7648		0.9248	0.7785	0.5409
2026	0.8958	0.9438	0.9841	0.924	0.7453	1.0006	0.9951	0.9482	0.762		0.9237	0.7801	0.5409
2027	0.8937	0.9422	0.9846	0.9245	0.7426	1.0003	0.9944	0.9465	0.7593		0.9225	0.7817	0.5419
2028	0.8917	0.9406	0.9852	0.9253	0.7409	1.0001	0.9937	0.9448	0.7568		0.9213	0.7832	0.5434
2029	0.8898	0.939	0.9858	0.9264	0.74	0.9996	0.993	0.9432	0.7544		0.92	0.7846	0.5452
2030	0.9931	0.9375	0.9864	0.9275	0.7397	0.9992	0.993	0.9418	0.7528		0.9186	0.7858	0.5472

S5h.Xiphopenaeus kroyeri simulated to future (Biomass)

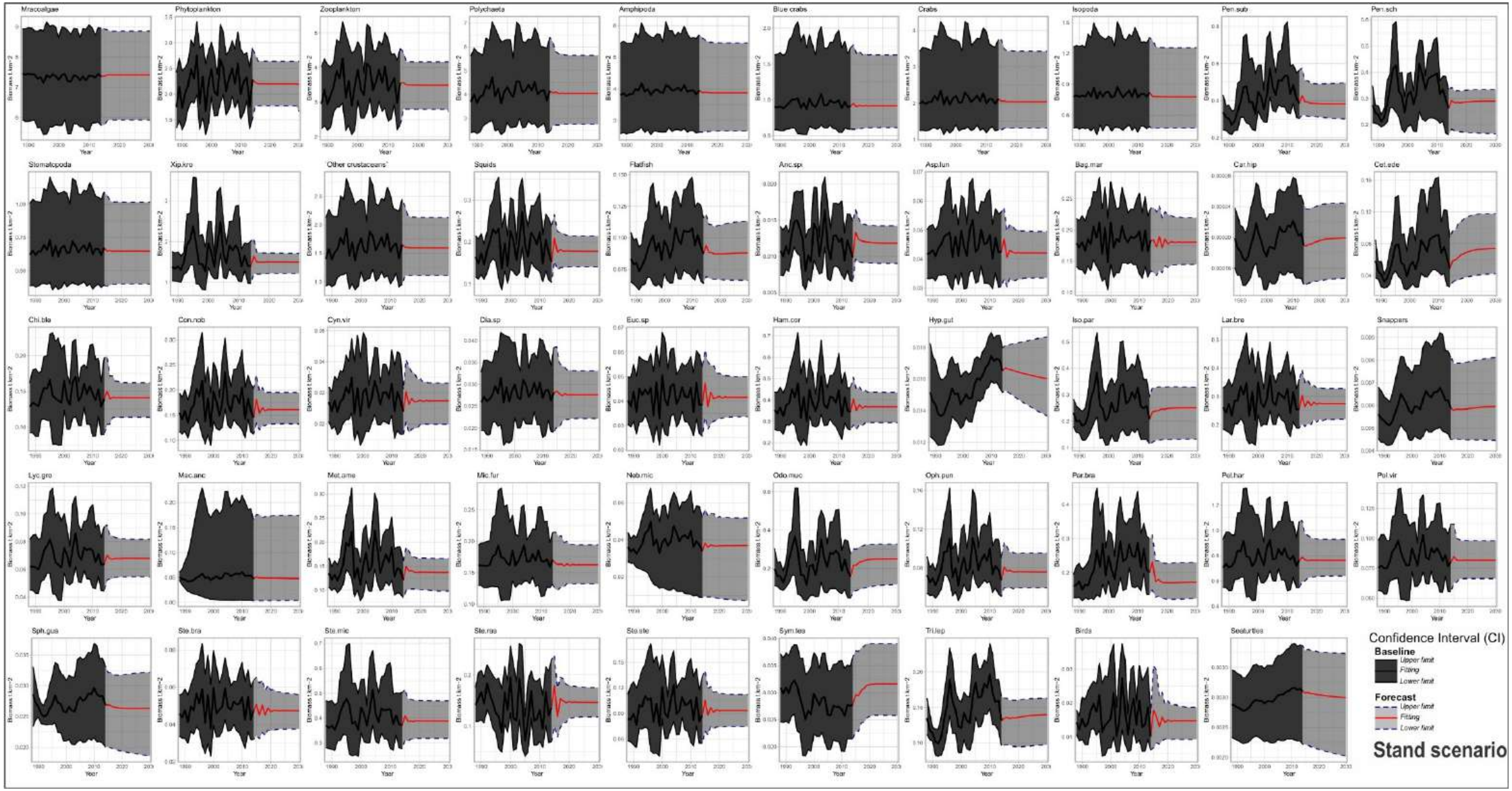
	Biomass ratio												
	4 months	3 months	increase effort				reduce effort				PP changes		
Years	clo1s	clo2s	inc(+10%)	inc(+25%)	inc(+50%)	inc(+100%)	dec(-10%)	dec(-25%)	dec(-50%)	no_fishing(dec(-100%)	env3	env4	env5
1988 to 2014	Baseline (effort constat)												
2015	1.0301	1.0281	0.9901	0.9749	0.9489	0.8942	1.01	1.0241	1.0476	1.0912	0.9689	0.9383	0.9
2016	1.033	1.0247	0.9924	0.9804	0.9592	0.912	1.0076	1.018	1.0347	1.0635	0.9707	0.9162	0.8327
2017	1.0214	1.0154	0.9956	0.9883	0.9742	0.9386	1.0042	1.0096	1.0173	1.028	0.9743	0.9182	0.828
2018	1.0117	1.0085	0.9975	0.9931	0.9836	0.9554	1.0022	1.0047	1.0078	1.0112	0.9763	0.9227	0.8338
2019	1.0053	1.004	0.9989	0.9966	0.9904	0.9674	1.0008	1.0014	1.0014	1.0001	0.9787	0.9324	0.8542
2020	1.0009	1.0007	1.0001	0.9993	0.9959	0.9775	0.9998	0.9991	0.9973	0.9939	0.9802	0.9391	0.8684
2021	0.998	0.9985	1.0007	1.0013	1.0002	0.9856	0.999	0.9974	0.9944	0.9895	0.9816	0.9451	0.8803
2022	0.9962	0.9971	1.0013	1.0028	1.0033	0.9921	0.9986	0.9964	0.9927	0.9869	0.9825	0.9497	0.8893
2023	0.9952	0.9963	1.0016	1.0037	1.0056	0.9973	0.9983	0.9958	0.9917	0.9853	0.9831	0.9531	0.8961
2024	0.9944	0.9957	1.0019	1.0045	1.0075	1.0017	0.9981	0.9953	0.9909	0.9839	0.9835	0.9558	0.9019
2025	0.994	0.9954	1.002	1.0049	1.0087	1.005	0.9979	0.9951	0.9905	0.983	0.9836	0.9577	0.9061
2026	0.9936	0.9951	1.0021	1.0053	1.0096	1.0078	0.9979	0.9949	0.9901	0.9821	0.9837	0.9592	0.9096
2027	0.9934	0.995	1.0022	1.0055	1.0103	1.0099	0.9978	0.9948	0.9898	0.9813	0.9836	0.9602	0.9122
2028	0.9932	0.9948	1.0022	1.0056	1.0108	1.0117	0.9977	0.9946	0.9895	0.9806	0.9835	0.961	0.9143
2029	0.9931	0.9947	1.0023	1.0057	1.0111	1.013	0.9977	0.9945	0.9893	0.9799	0.9833	0.9615	0.916
2030	0.9906	0.9946	1.0023	1.0058	1.0114	1.0141	0.9976	0.9944	0.9891	0.9793	0.9832	0.962	0.9174

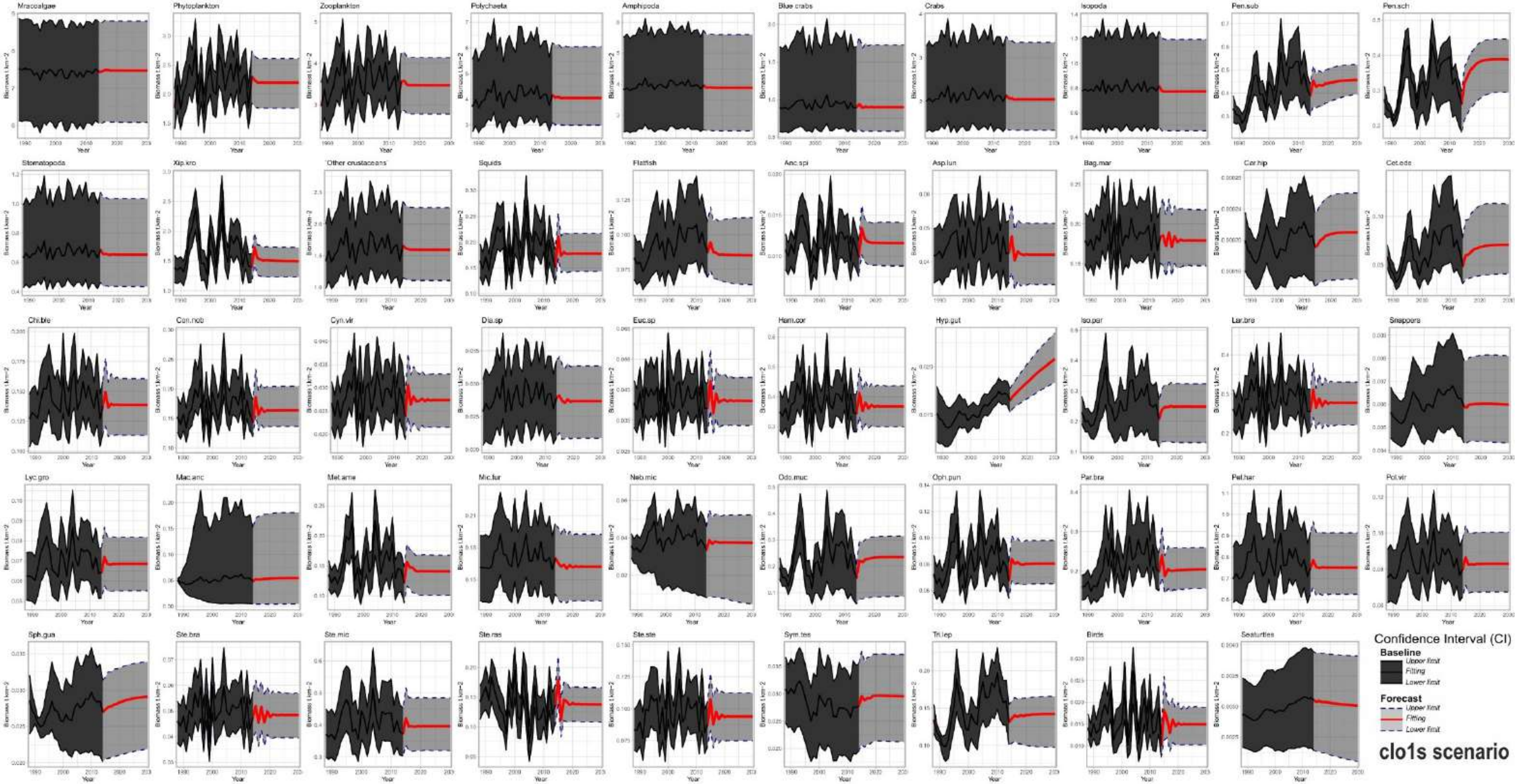
S5i.Xiphopenaeus kroyeri simulated to future (Catch)

	Biomass ratio												
	4 months	3 months	increase effort				reduce effort				PP changes		
Years	clo1s	clo2s	inc(+10%)	inc(+25%)	inc(+50%)	inc(+100%)	dec(-10%)	dec(-25%)	dec(-50%)	no_fishing(dec(-100%)	env3	env4	env5
1988 to 2014	Baseline (effort constat)												
2015	0.6696	0.7467	1.0887	1.2181	1.4227	1.7879	0.9072	0.7692	0.5225		0.9689	0.9383	0.9
2016	0.6841	0.7658	1.0912	1.225	1.4382	1.8233	0.905	0.7644	0.5161		0.9707	0.9162	0.8327
2017	0.6709	0.7541	1.0947	1.2348	1.4606	1.8763	0.9019	0.758	0.5074		0.9743	0.9182	0.828
2018	0.6683	0.752	1.0968	1.2409	1.4748	1.9101	0.9001	0.7542	0.5026		0.9763	0.9227	0.8338
2019	0.6636	0.7476	1.0984	1.2452	1.4849	1.934	0.8989	0.7516	0.4995		0.9787	0.9324	0.8542
2020	0.6604	0.7452	1.0995	1.2486	1.4932	1.9541	0.898	0.7498	0.4974		0.9802	0.9391	0.8684
2021	0.6588	0.7437	1.1004	1.2511	1.4996	1.9704	0.8973	0.7484	0.496		0.9816	0.9451	0.8803
2022	0.6574	0.7426	1.1009	1.2529	1.5044	1.9834	0.8969	0.7475	0.4951		0.9825	0.9497	0.8893
2023	0.6569	0.742	1.1013	1.2541	1.5078	1.9937	0.8966	0.7469	0.4946		0.9831	0.9531	0.8961
2024	0.6563	0.7415	1.1016	1.2551	1.5106	2.0026	0.8964	0.7465	0.4942		0.9835	0.9558	0.9019
2025	0.6561	0.7413	1.1018	1.2556	1.5124	2.0092	0.8963	0.7462	0.494		0.9836	0.9577	0.9061
2026	0.6558	0.7411	1.1019	1.256	1.5138	2.0148	0.8963	0.7459	0.4938		0.9837	0.9592	0.9096
2027	0.6557	0.741	1.1019	1.2563	1.5148	2.019	0.8962	0.7456	0.4937		0.9836	0.9602	0.9122
2028	0.6556	0.7409	1.102	1.2565	1.5155	2.0225	0.8962	0.7454	0.4935		0.9835	0.961	0.9143
2029	0.6555	0.7408	1.1021	1.2567	1.516	2.0253	0.8961	0.7452	0.4934		0.9833	0.9615	0.916
2030	0.7346	0.7407	1.1022	1.2568	1.5166	2.0282	0.8967	0.7452	0.4937		0.9832	0.962	0.9174

Biomass predicted simulated to future with Monte Carlo routine

Figure S8. Biomass predicted in the model with confidence interval 95% by Monte Carlo routine (1000 runs) for each group and FMS.

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Zooplankton
Polychaeta
Amphipoda
Blue crabs
Crabs
Isopoda
Pen.sub
Pen.sch
Stomatopoda
Xip.kro
'Other crustaceans'
Squids
Flatfish
Anc.spi
Asp.lun
Bag.mar
Car.hip
Cet.ede
Chl.ble
Con.nob
Cyn.vir
Dia.sp
Euc.sp
Ham.cor
Hyp.gut
Iso.par
Lar.bre
Snappers
Lyc.gro
Mac.anc
Met.ame
Mic.fur
Neb.mic
Odo.muc
Oph.pun
Par.bra
Pel.har
Pol.vir
Sph.gua
Ste.bra
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Ste.ras
Ste.ste
Sym.tes
Tri.lep
Birds
Seaturtles
Confidence Interval (CI)
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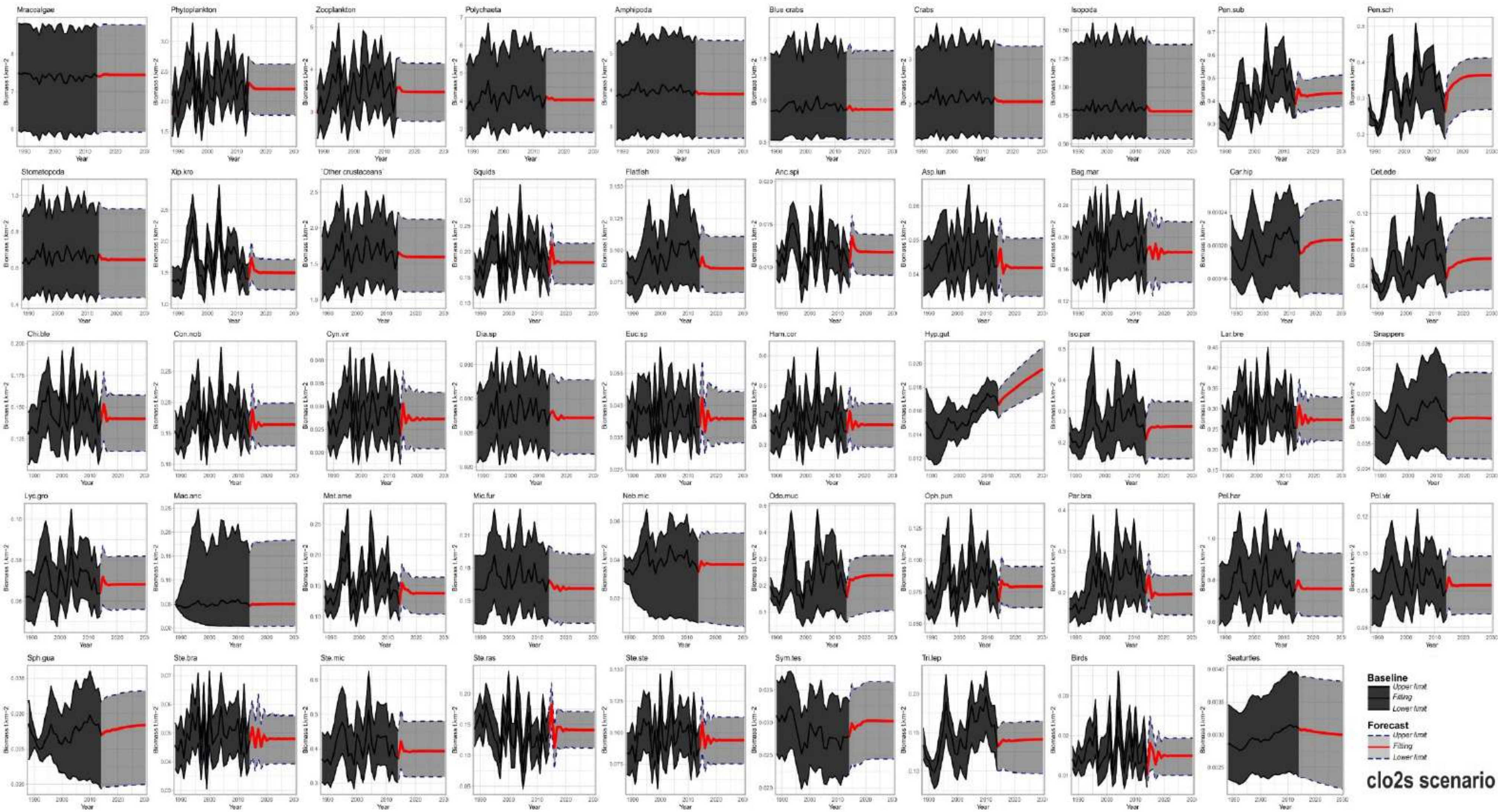
Upper limit
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
clo1s scenario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Birds
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Biomass t.km-2
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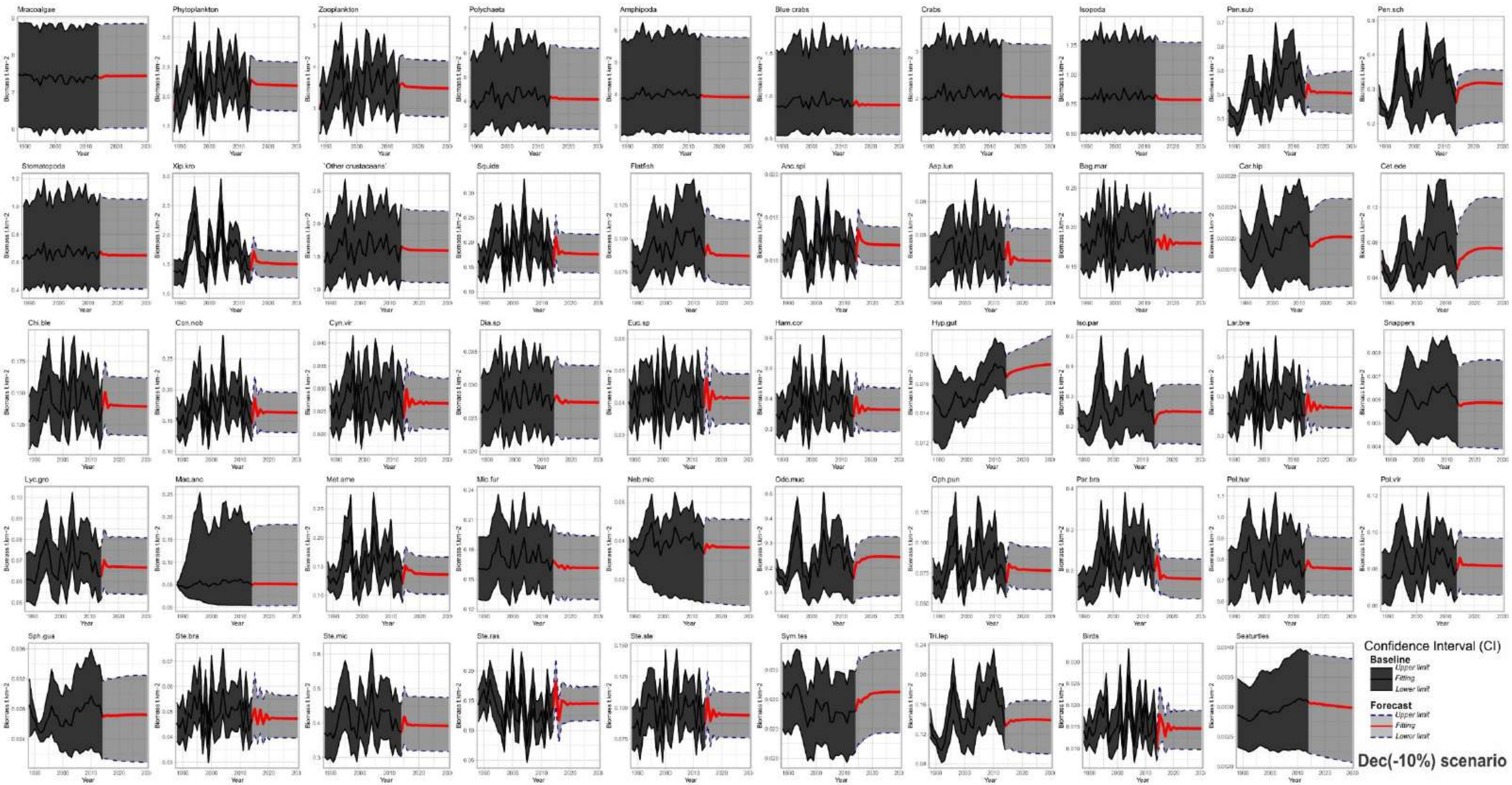
Upper limit
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
clo2s scenario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Biomass t.km-2
Year
Confidence Interval (CI)
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Upper limit
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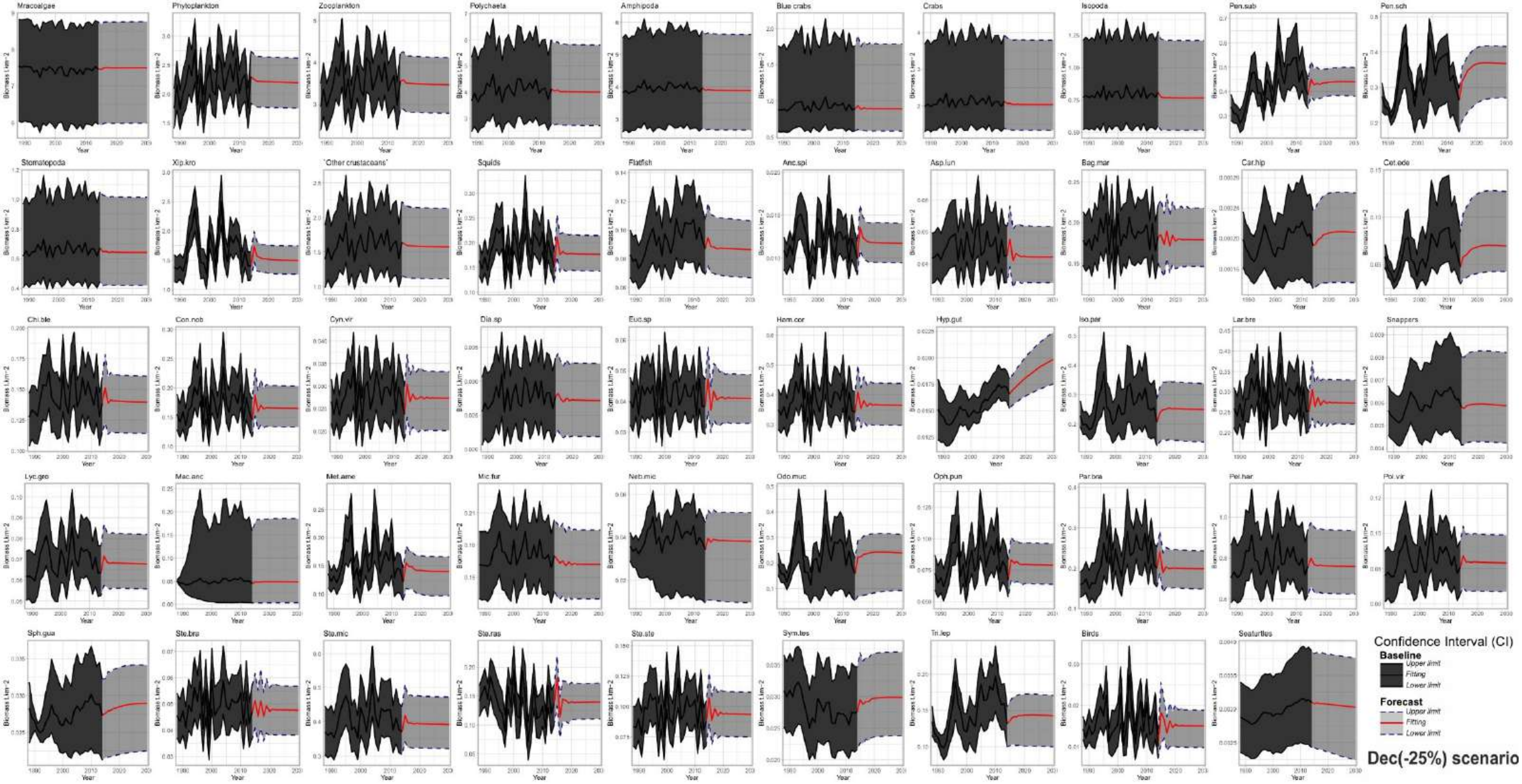
Forecast
Upper limit
Fitting
Lower limit
Dec(-10%) scenario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Birds
Seaturtles
Biomass t.km−2
Year
Confidence Interval (CI)
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Upper limit
Fitting
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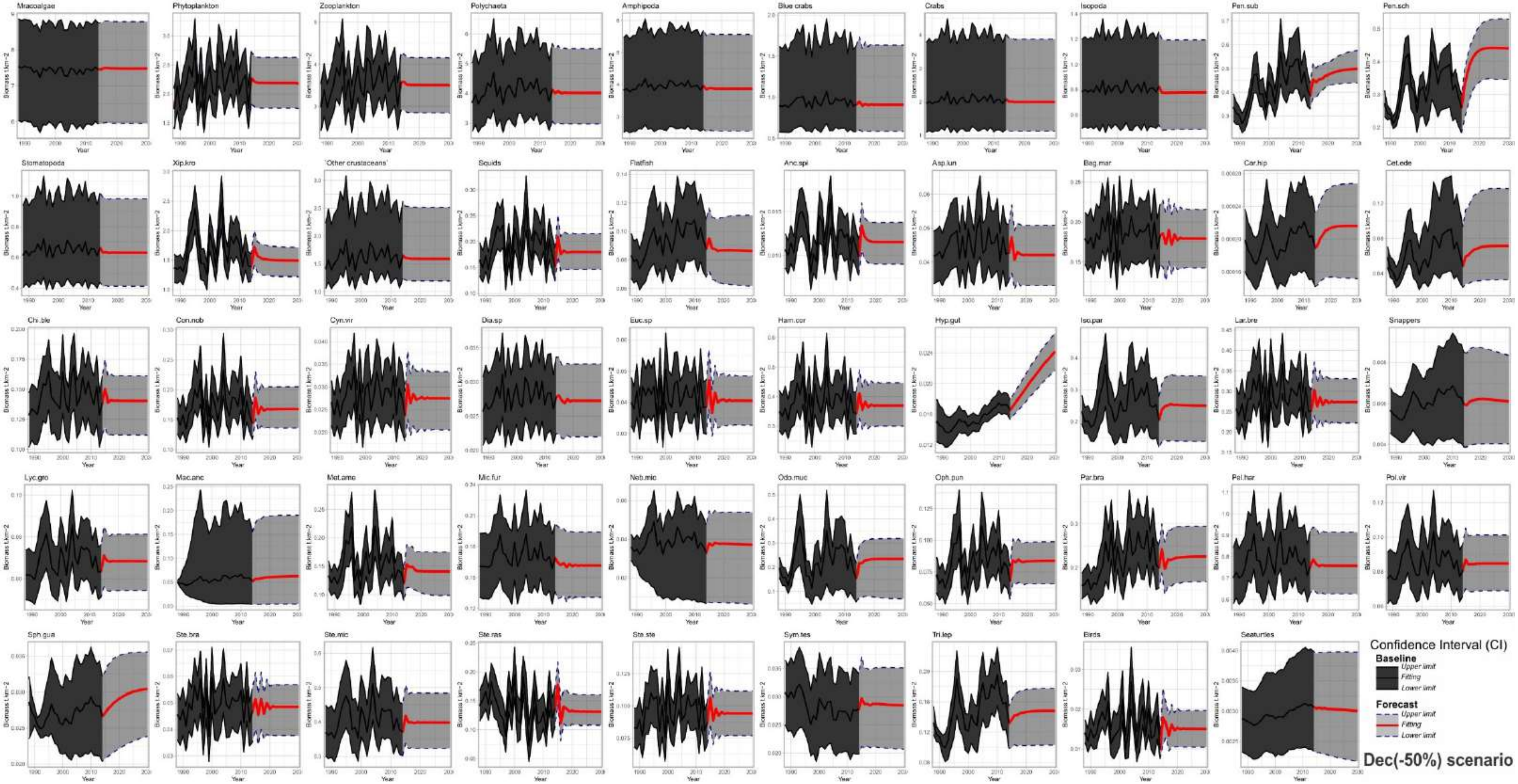
Forecast
Upper limit
Fitting
Lower limit
Dec(-25%) scenario



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Biomass t.km-2
Year
Confidence Interval (CI)
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Upper limit
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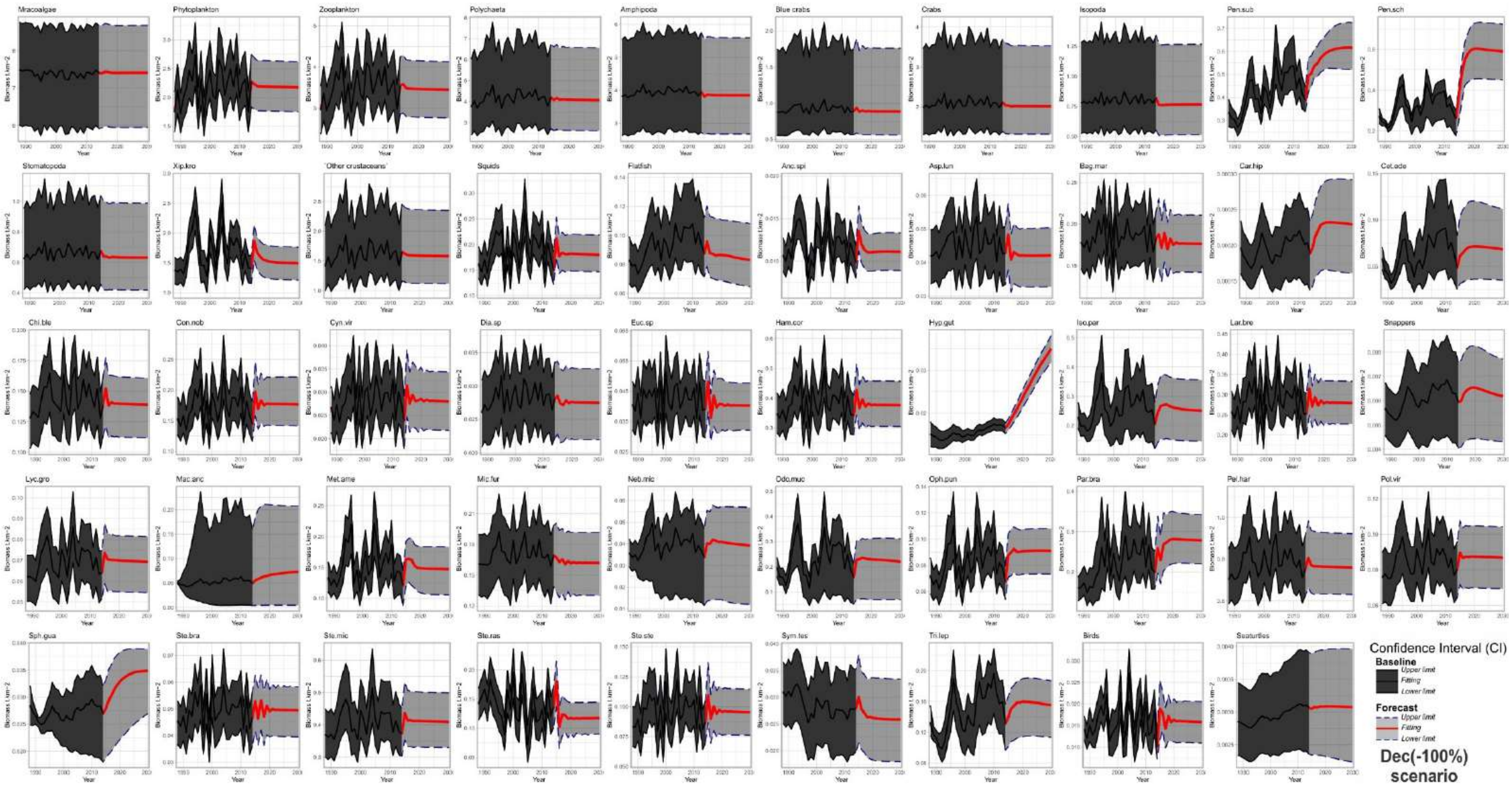
Forecast
Upper limit
Fitting
Lower limit
Dec(-50%) scenario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Birds
Seaturtles
Biomass t.km-2
Year
Confidence Interval (CI)
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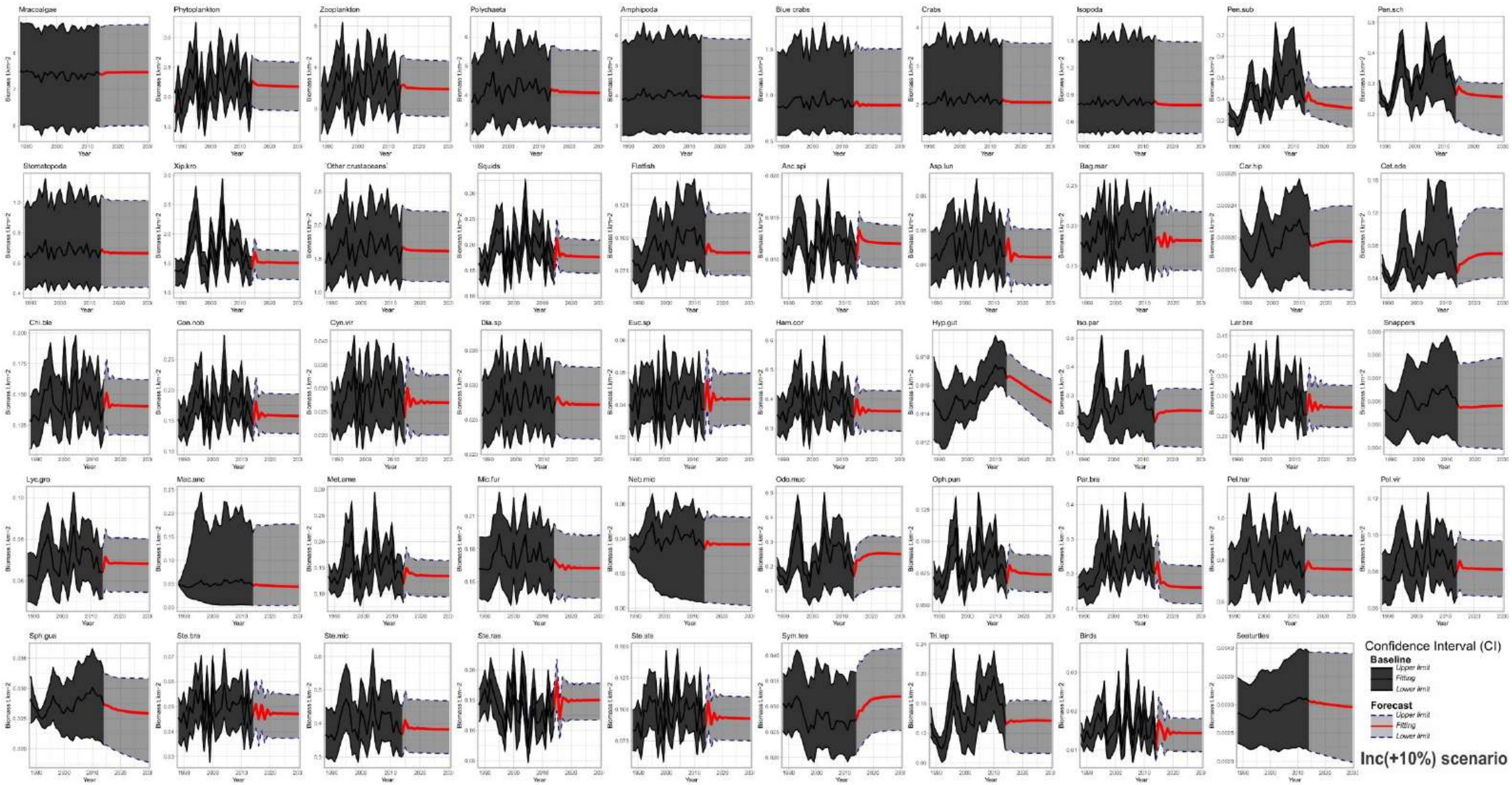
Lower limit
Forecast
Upper limit
Fitting
Lower limit
Dec(-100%)
scenario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Seaturtles
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Year
Confidence Interval (CI)
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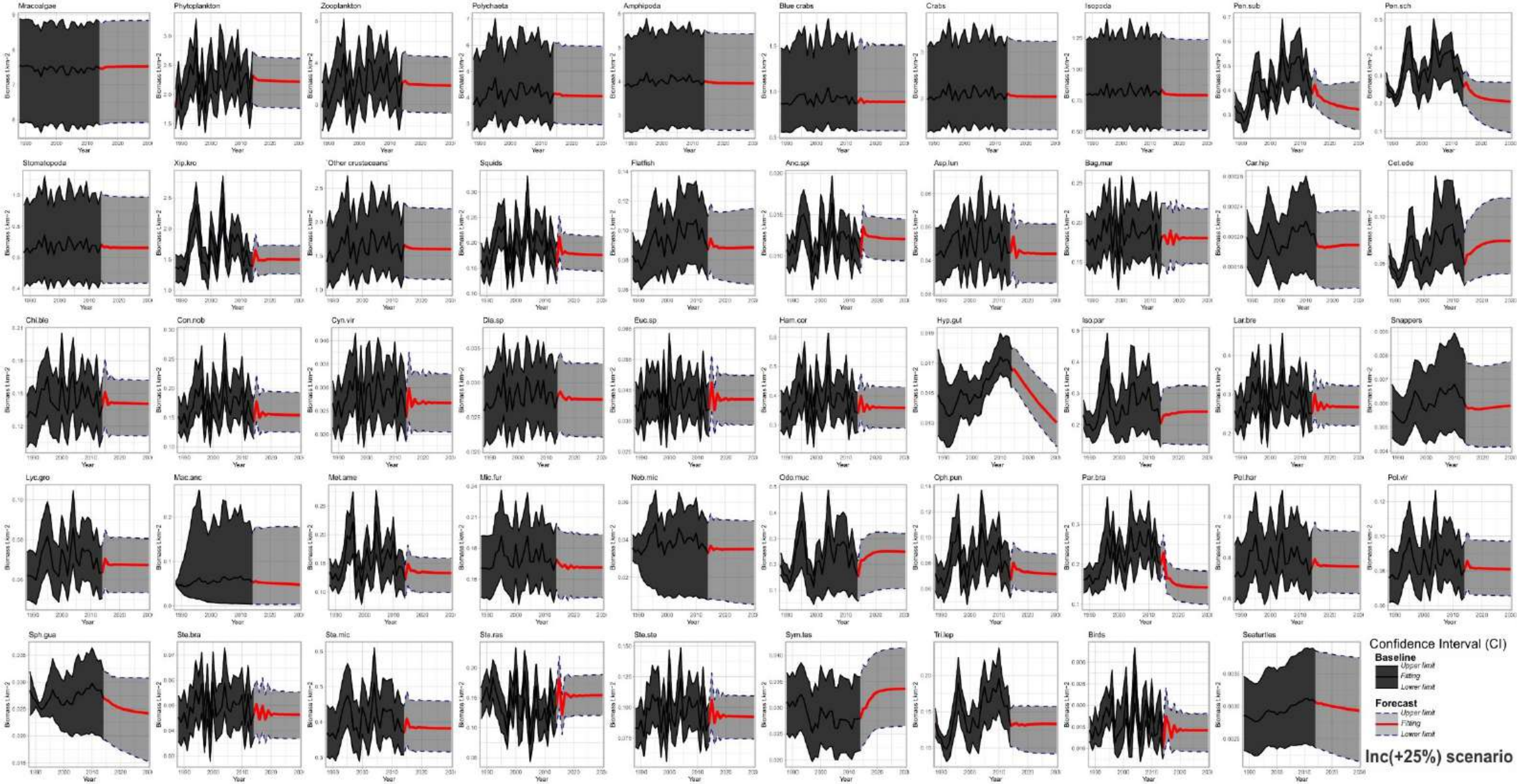
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
Inc(+10%) scenario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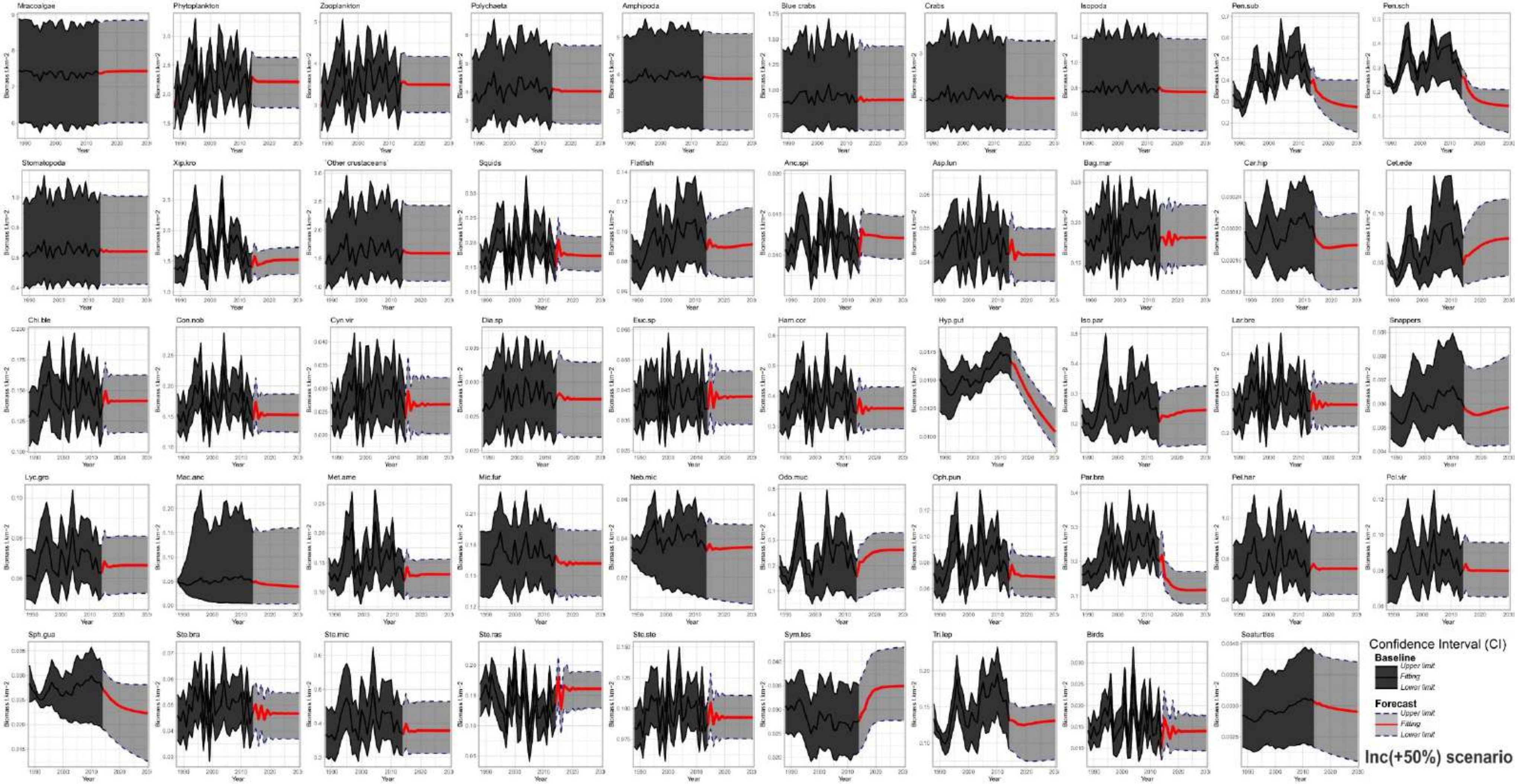
Year
Confidence Interval (CI)
Baseline
Upper limit
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
Inc(+25%) scenario



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Seaturtles
Year
Biomass t.km-2
Confidence Interval (CI)
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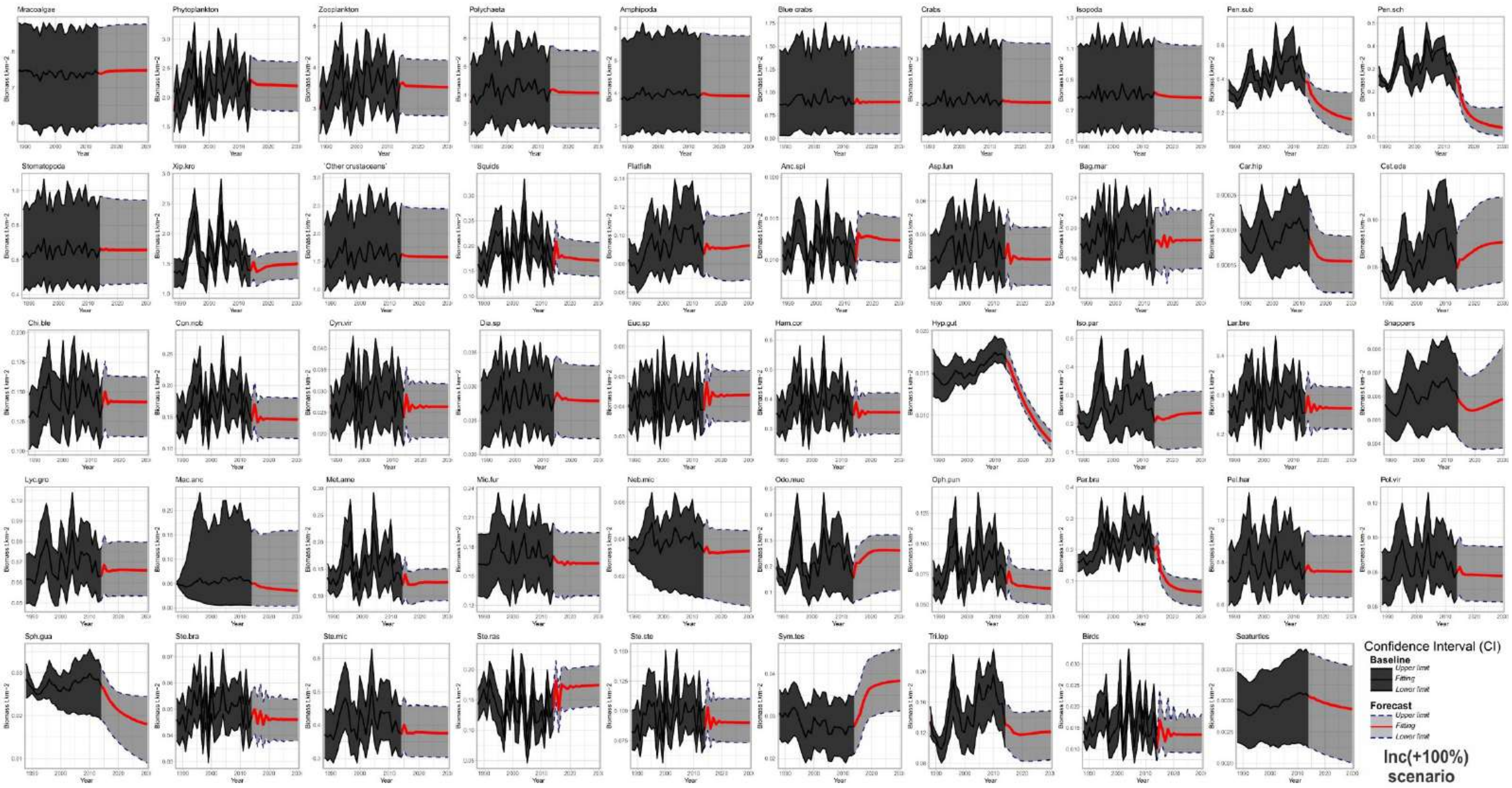
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
Inc(+50%) scenario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Birds
Seaturtles
Year
Biomass t.km-2
Confidence Interval (CI)
Baseline
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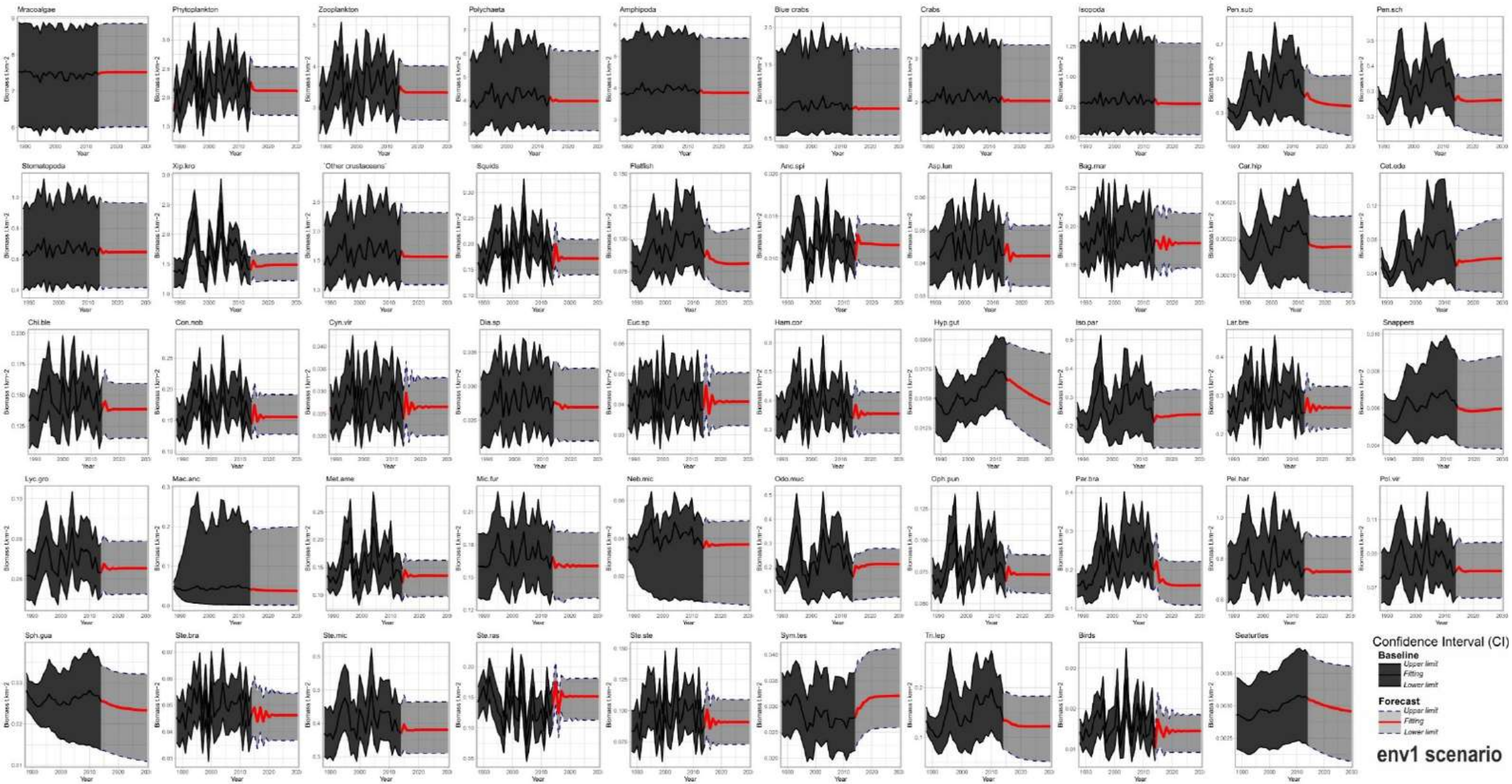
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
Inc(+100%)
scenario



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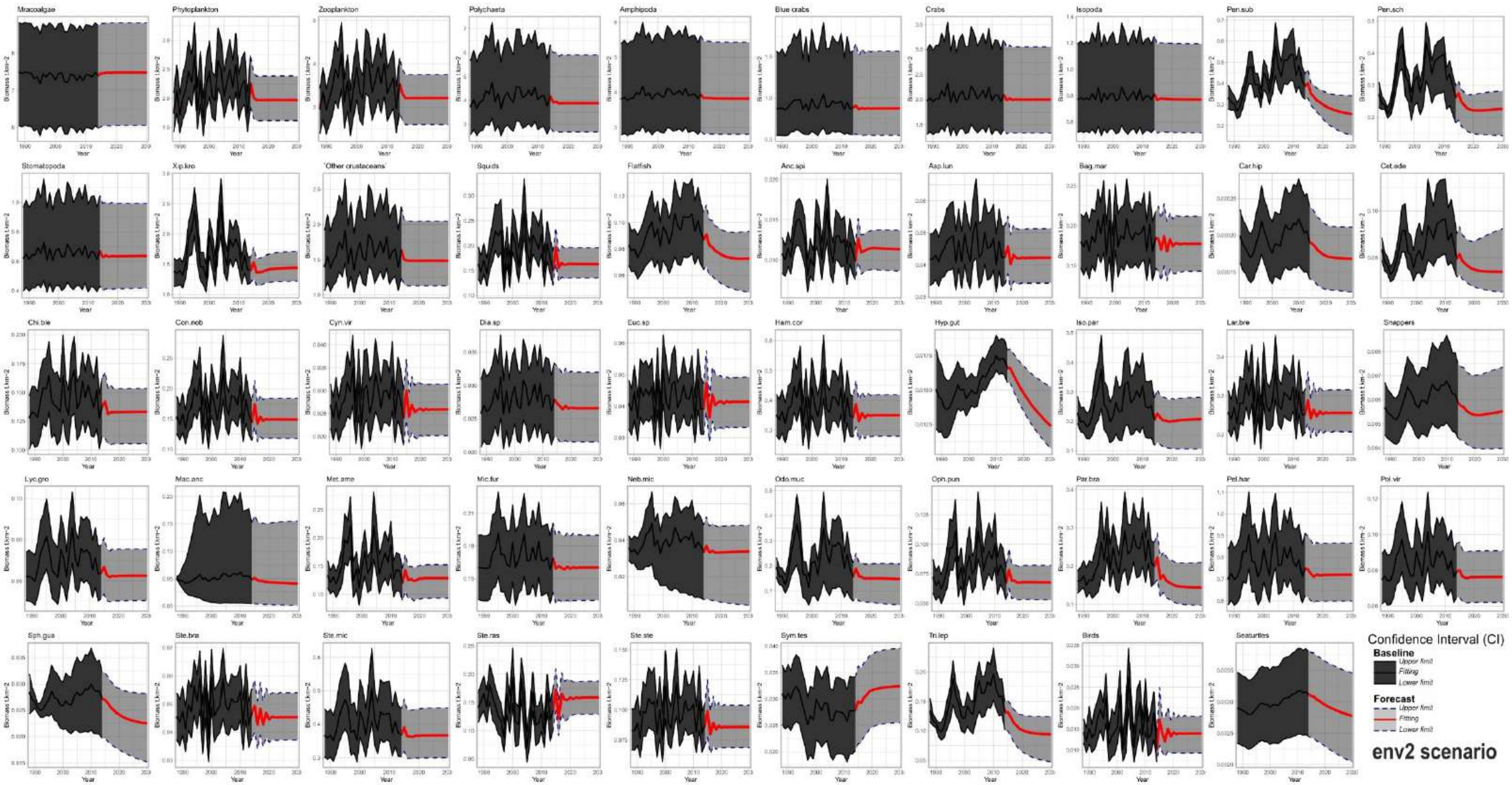
Ste.ste
Sym.tes
Tri.lep
Birds
Seaturtles
Confidence Interval (CI)
Baseline
Upper limit
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
env1 scenario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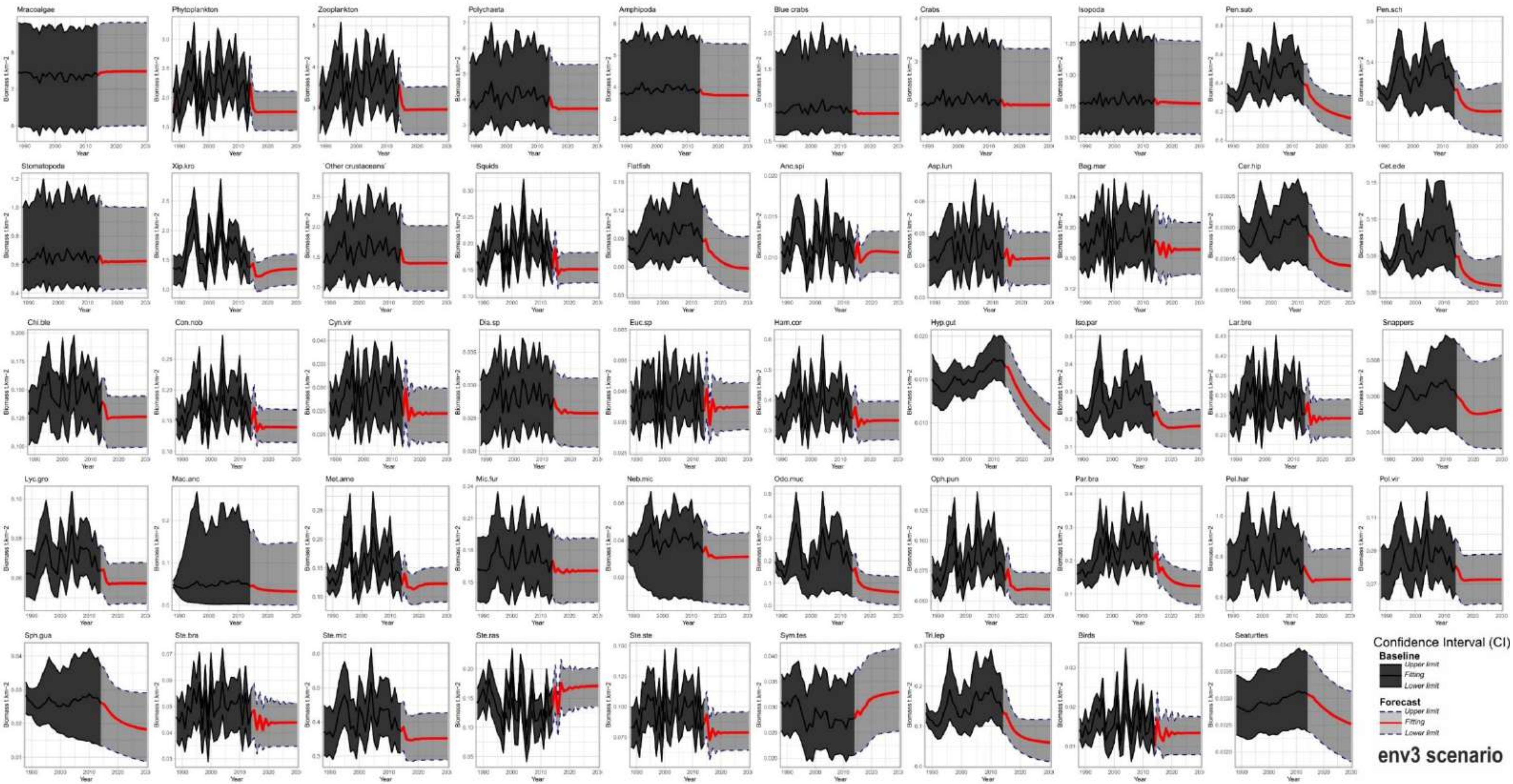
Sym.tes
Tri.lep
Birds
Seaturtles
Biomass t.km-2
Year
Confidence Interval (CI)
Baseline
Upper limit
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
env2 scenario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Blue crabs
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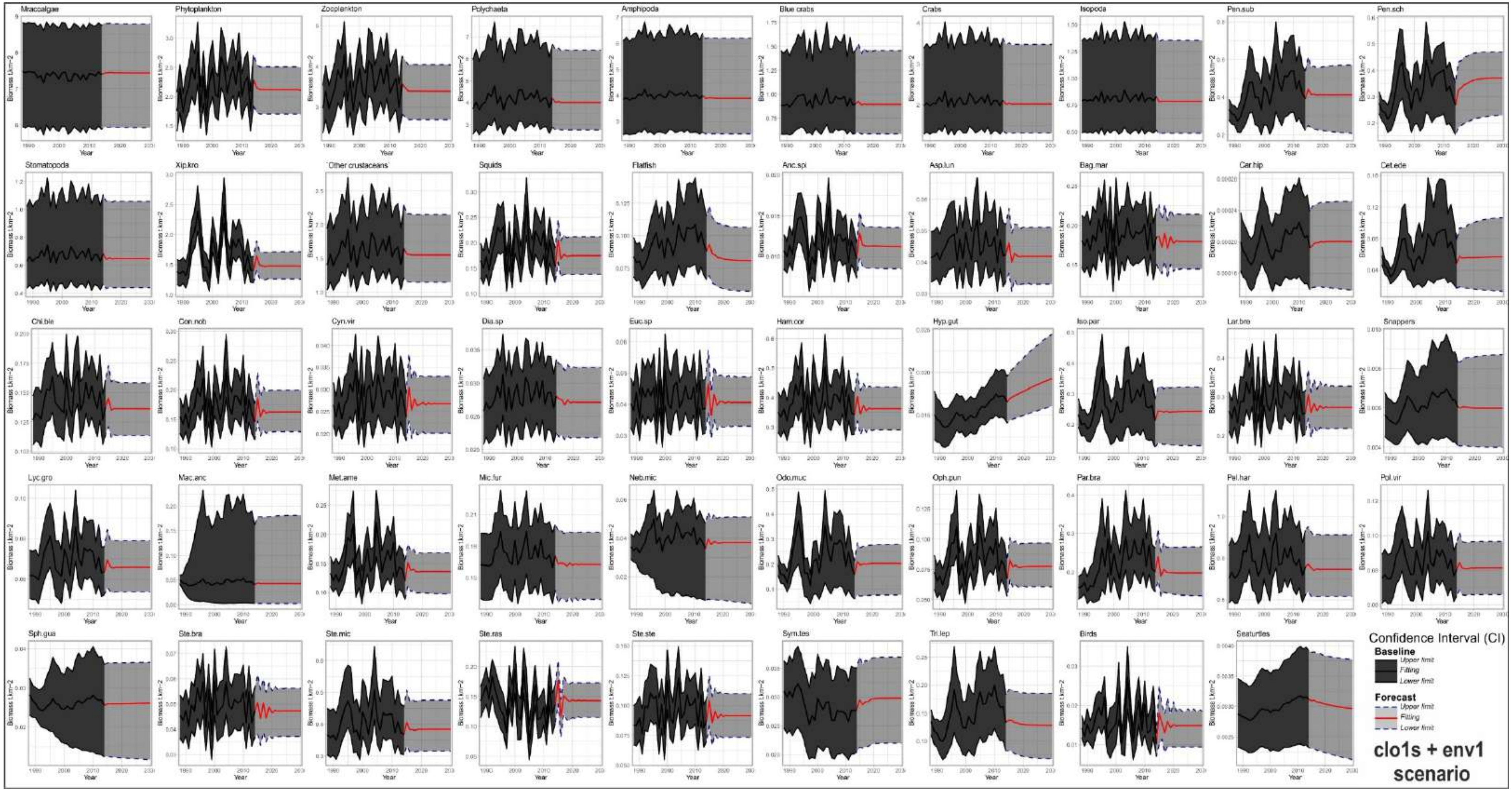
Sym.tes
Tri.lep
Birds
Seaturtles
Biomass t.km−2
Year
Confidence Interval (CI)
Baseline
Upper limit
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
env3 scenario



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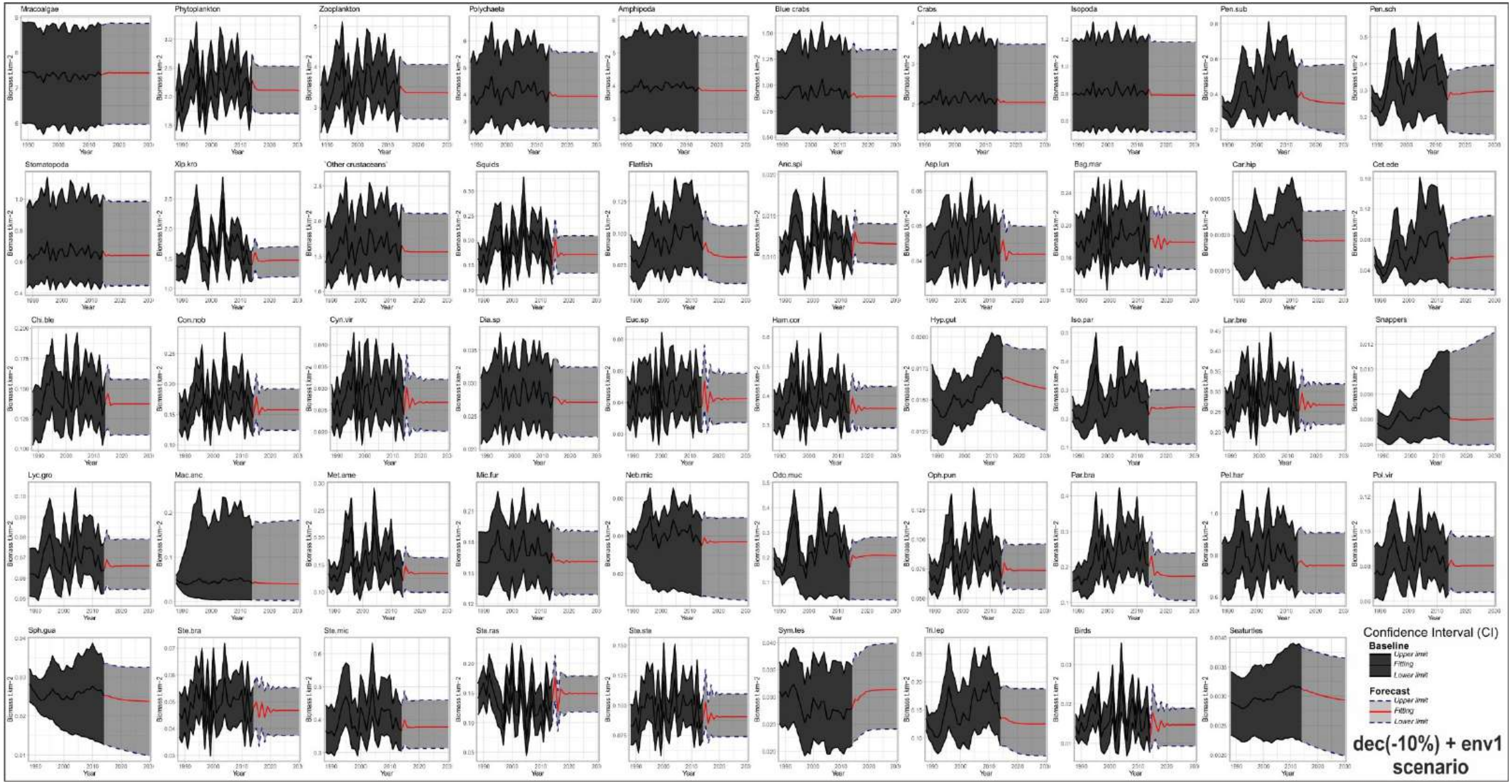
Birds
Seaturtles
Year
Biomass t.km-2
Confidence Interval (CI)
Baseline
Upper limit
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
clo1s + env1
scenario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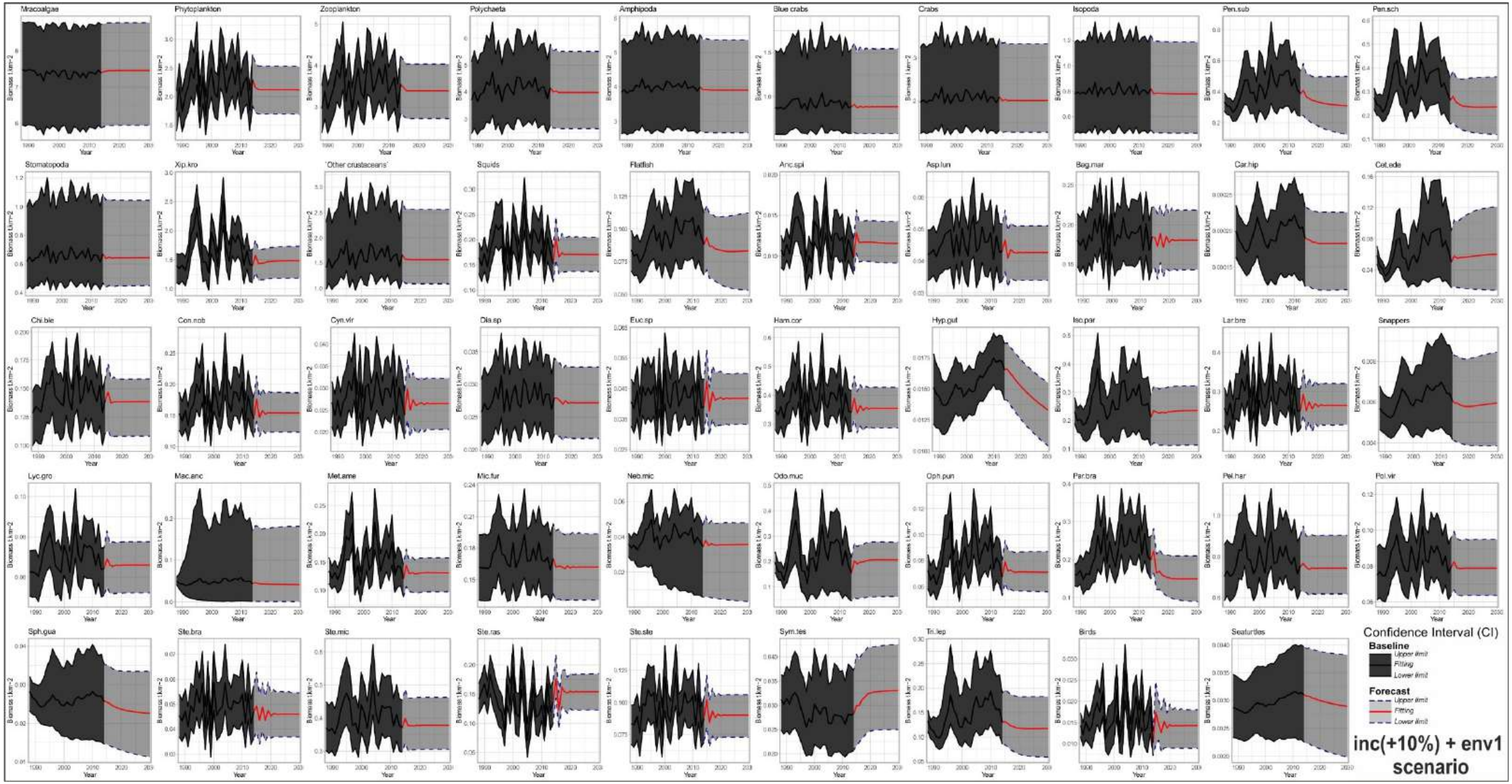
Tri.lep
Birds
Seaturtles
Biomass t.km-2
Year
Confidence Interval (CI)
Baseline
Upper limit
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
dec(-10%) + env1 scenario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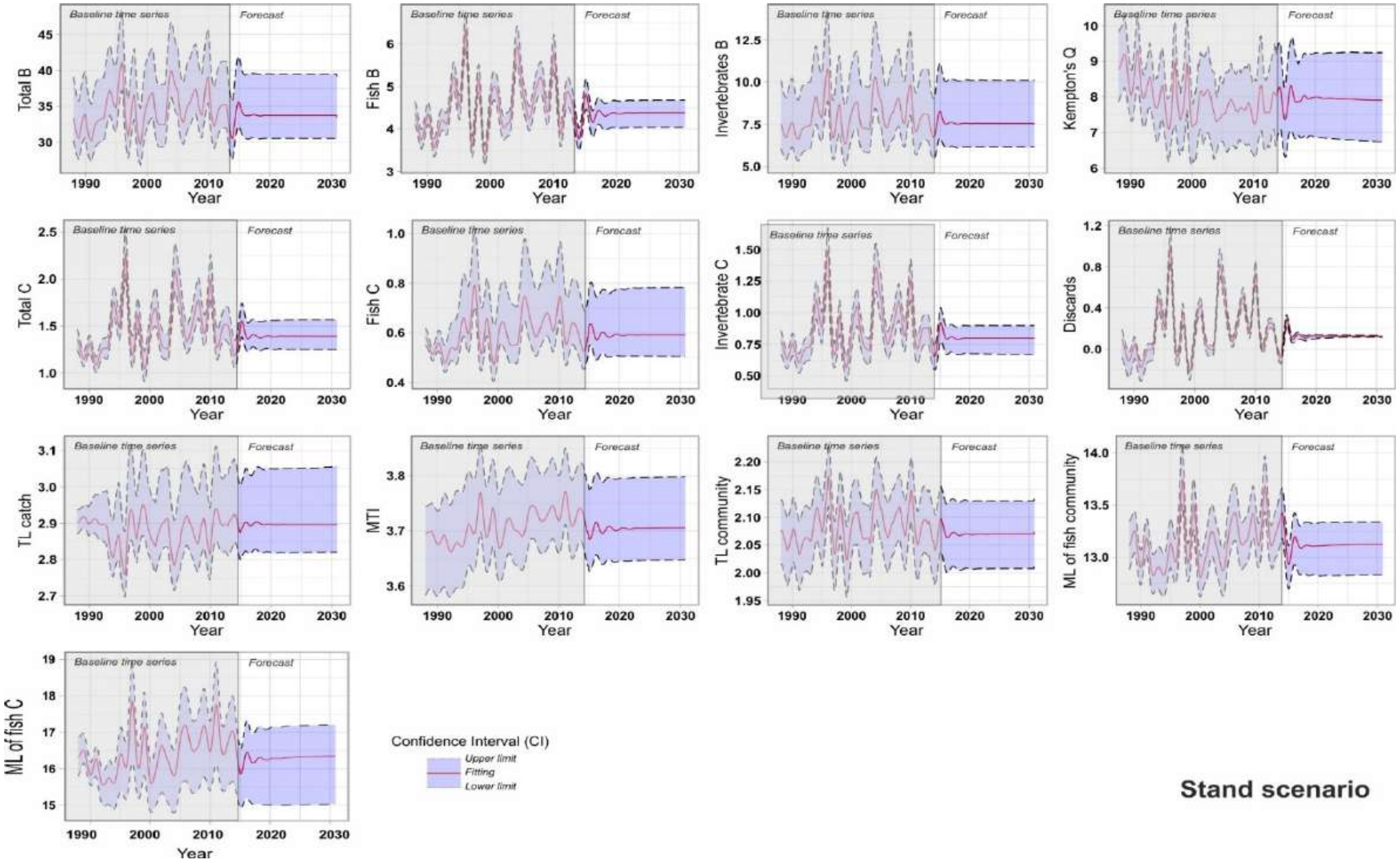
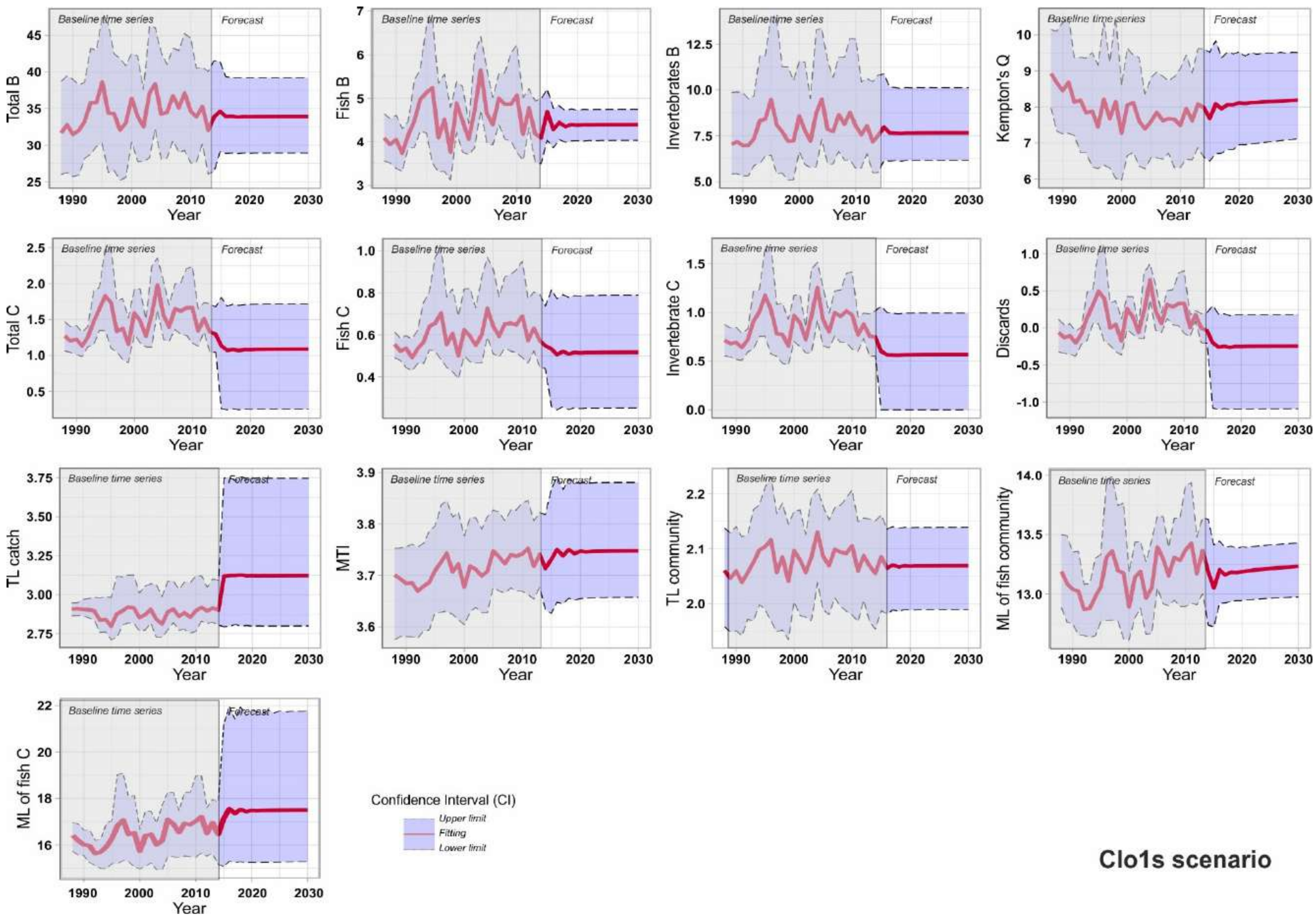
Birds
Seaturtles
Biomass t.km−2
Year
Confidence Interval (CI)
Baseline
Upper limit
Fitting
Lower limit
Forecast
Upper limit
Fitting
Lower limit
inc(+10%) + env1
scenario

Ecological indicators Ecosim model (back to future – 1988 to 2030)

Figure S9. Ecological indicators estimated for each scenario from the Ecosim for the period 1988–2030 for of the Barra of Sirinhaém Ecopath model (BSIR), Pernambuco, Northeast of Brazil. Total biomass - Total B (t·km^{-2}); biomass of fish and invertebrate - Fish B and Inver. B (t·km^{-2}); Kempton's biodiversity index (Q) - Kemp.Q; Total Catch - Total C (t·km^{-2}·year^{-1}); Catch of fish and invertebrate - Fish C and Inver. C (t·km^{-2}·year^{-1}); Total discarded catch – Disc (t·km^{-2}·year^{-1}); Tropic level (TL) of the catch and of the community (including all organisms) – mTLc and mTLco; Marine trophic index – MTI; Mean length of fish community and of fish catch – MLFco and MLFc (cm). The results are based on 1000 Ecosim model runs, obtained through the Monte Carlo routine, where the red line is fitting model and blue shadow represents the confidence interval 95%.





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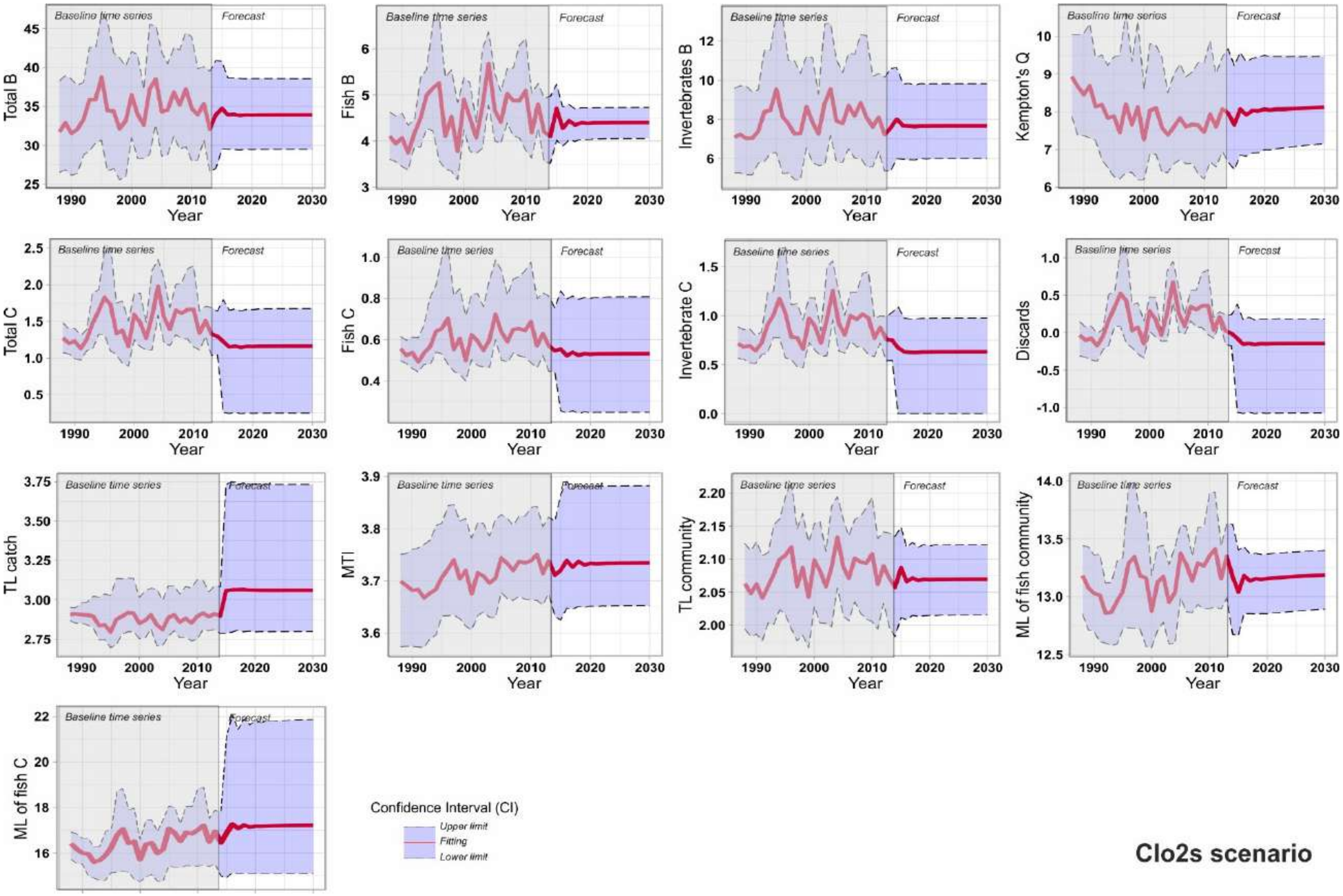
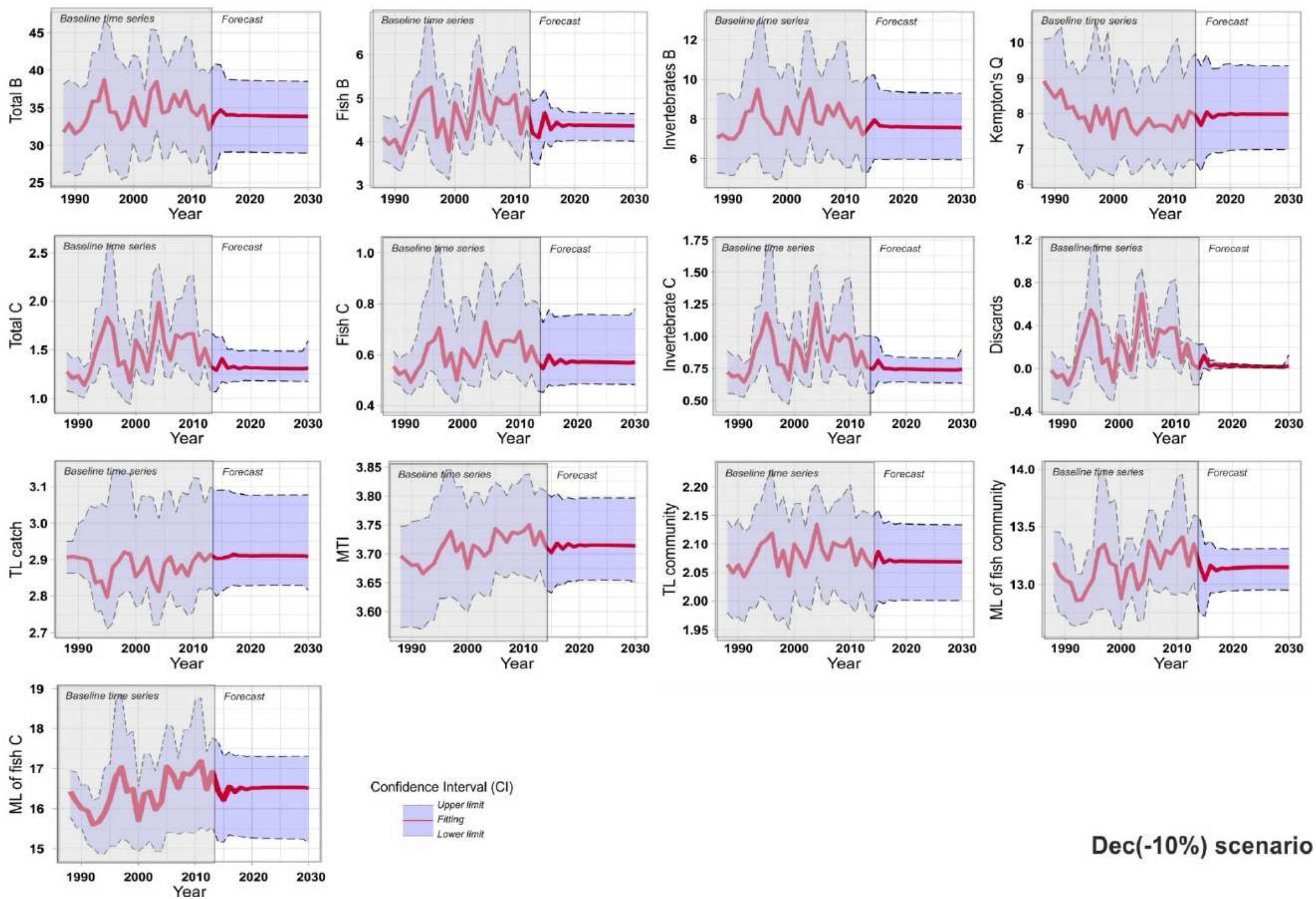
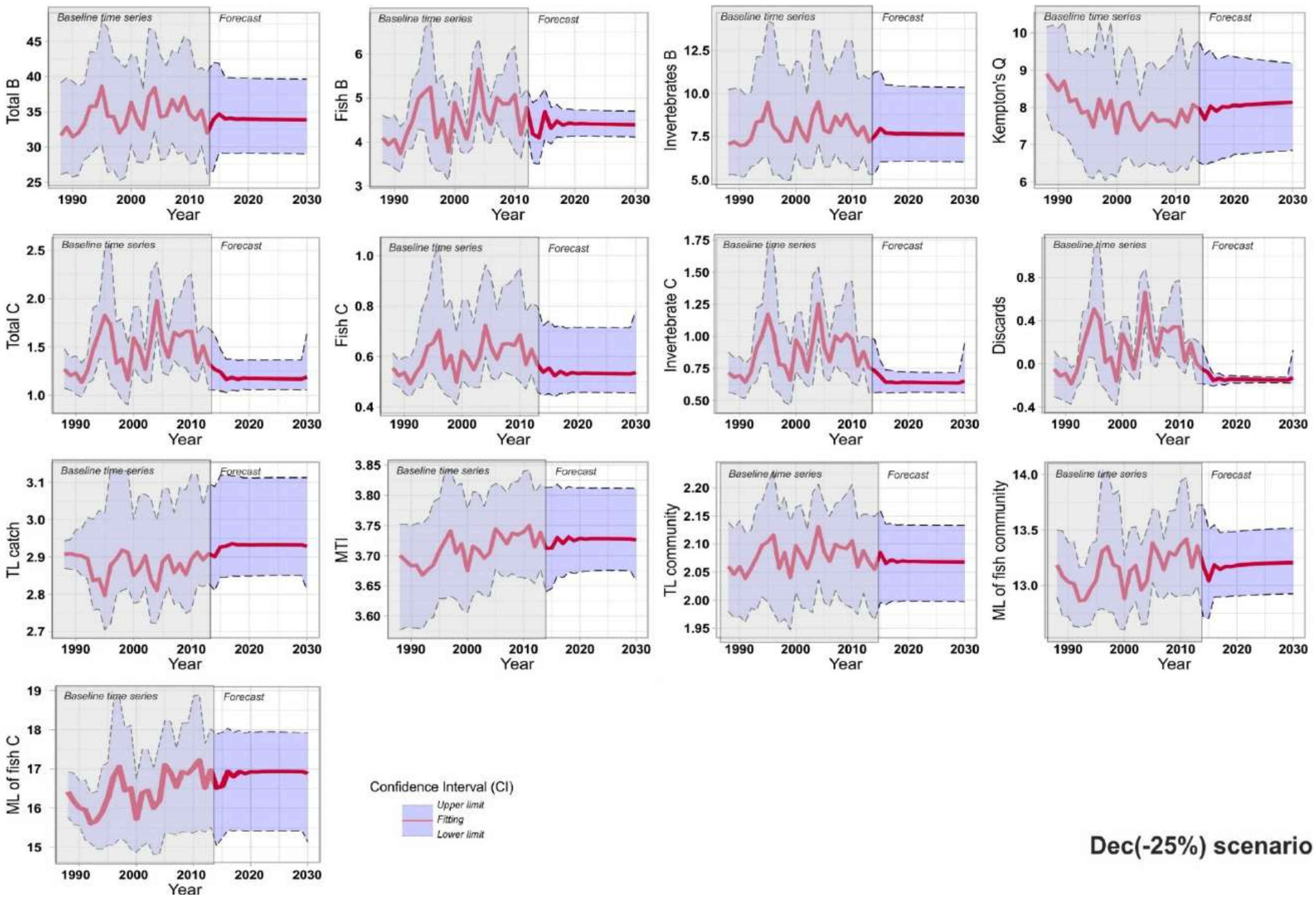
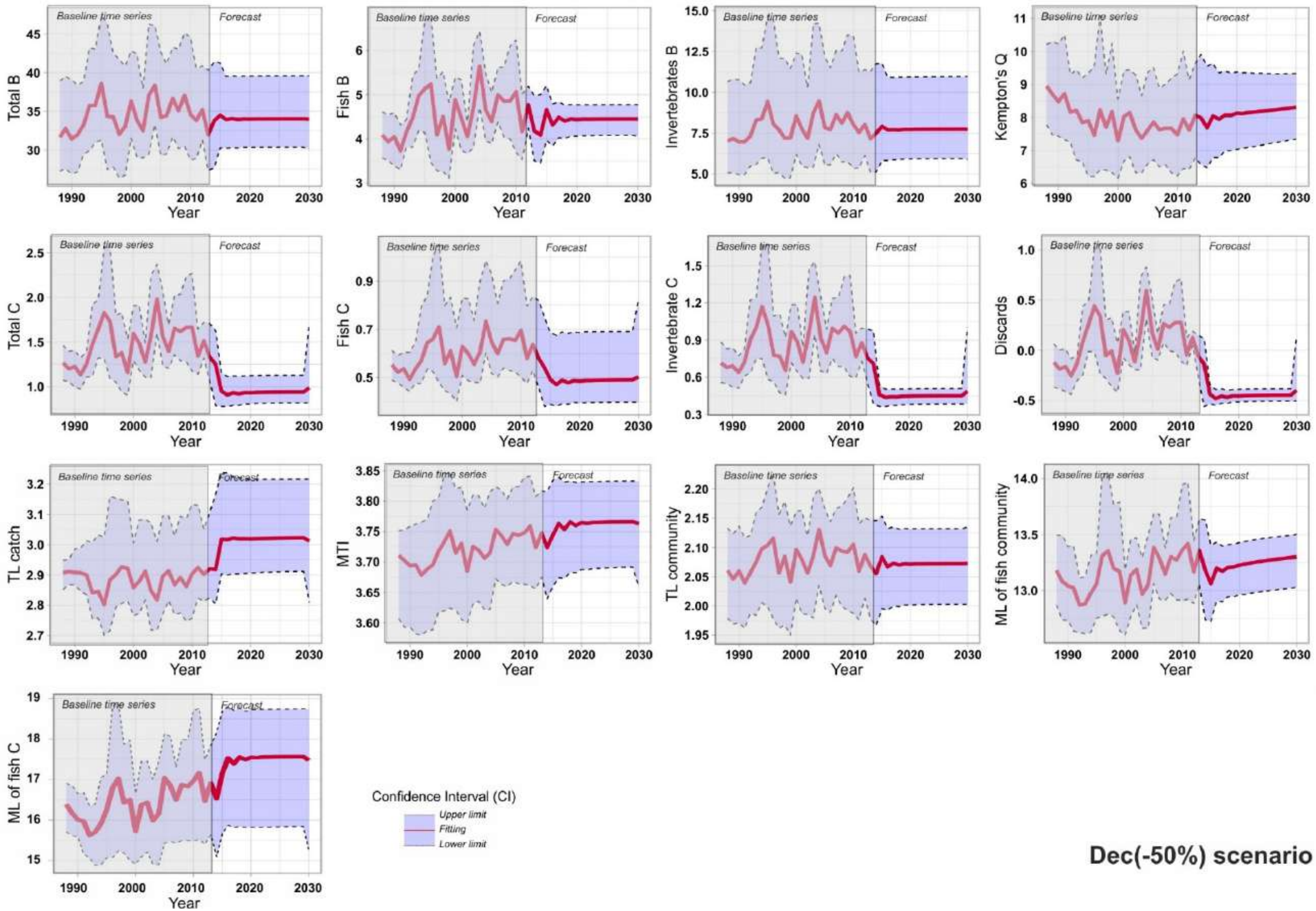


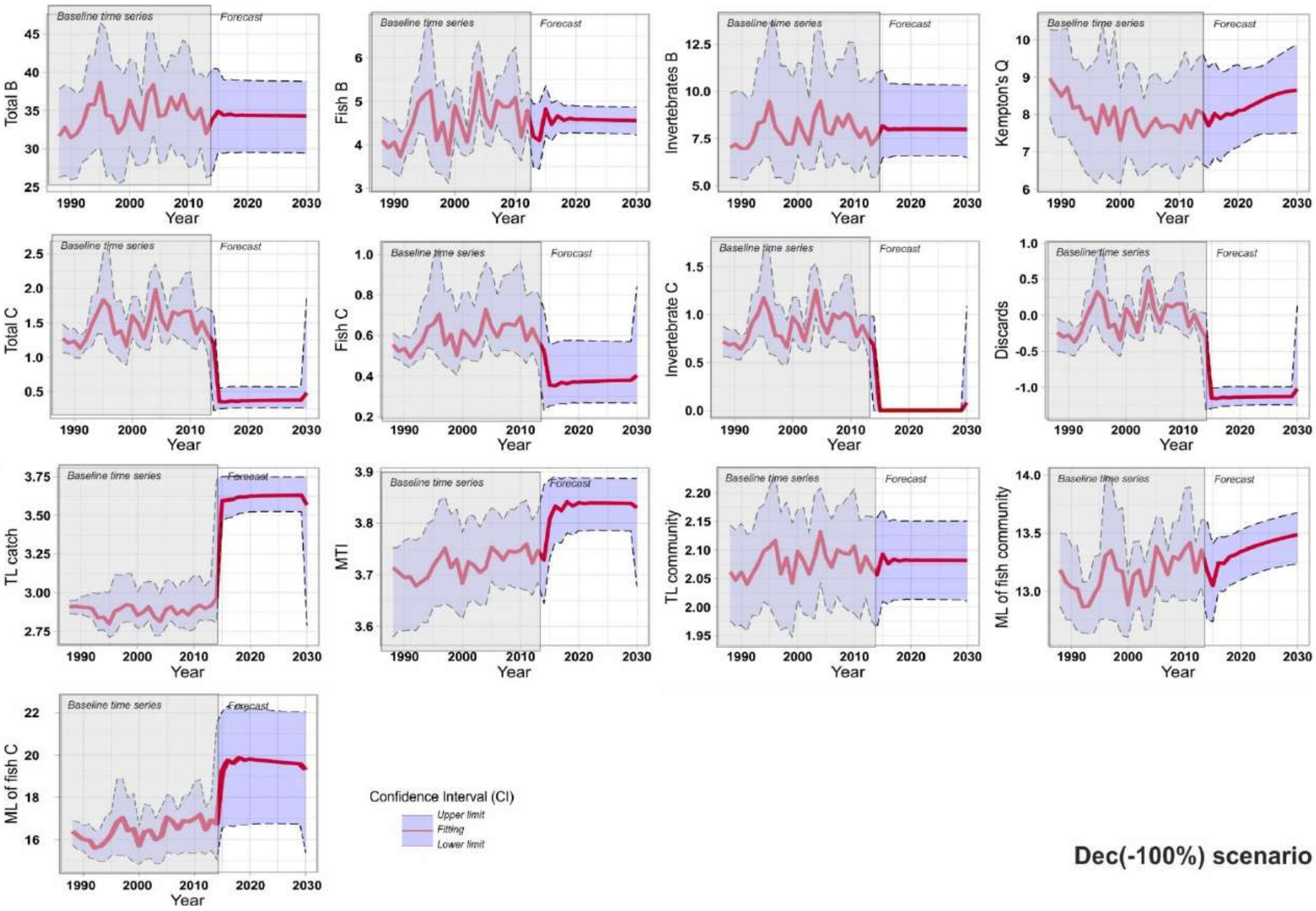
Clo2s scenario


Baseline time series
Forecast
Year
Total B
Fish B
Invertebrates B
Kempton's Q
Total C
Fish C
Invertebrate C
Discards
TL catch
MTI
TL community
ML of fish community
ML of fish C
Confidence Interval (CI)
Upper limit
Fitting
Lower limit
Dec(-10%) scenario


Baseline time series
Forecast
Total B
Fish B
Invertebrates B
Kempton's Q
Total C
Fish C
Invertebrate C
Discards
TL catch
MTI
TL community
ML of fish community
ML of fish C
Year
Confidence Interval (CI)
Upper limit
Fitting
Lower limit
Dec(-25%) scenario



Baseline time series
Forecast
Total B
Fish B
Invertebrates B
Kempton's Q
Total C
Fish C
Invertebrate C
Discards
TL catch
MTI
TL community
ML of fish community
ML of fish C
Year
Confidence Interval (CI)
Upper limit
Fitting
Lower limit
Dec(-50%) scenario


Baseline time series
Forecast
Total B
Fish B
Invertebrates B
Kempton's Q
Total C
Fish C
Invertebrate C
Discards
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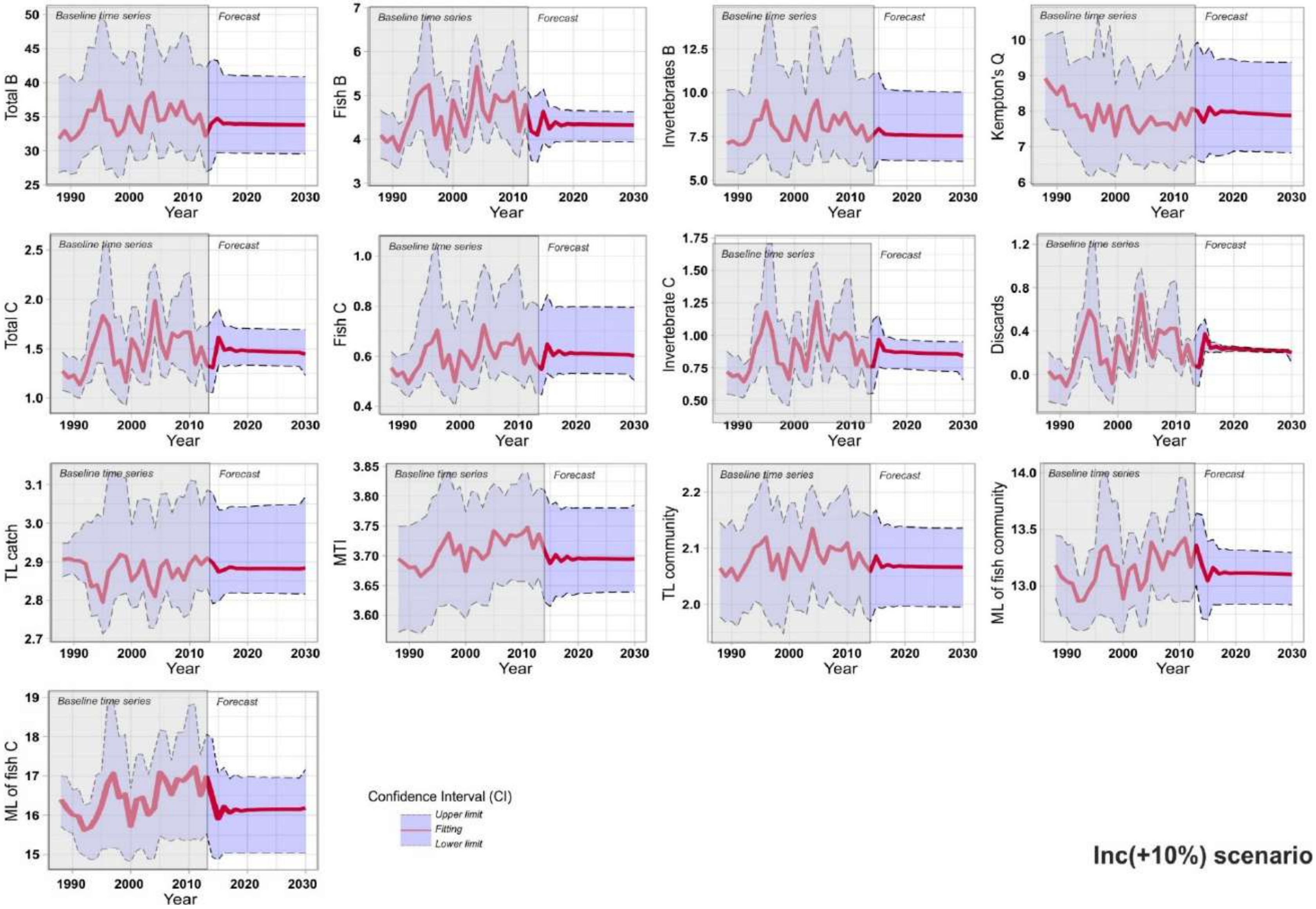
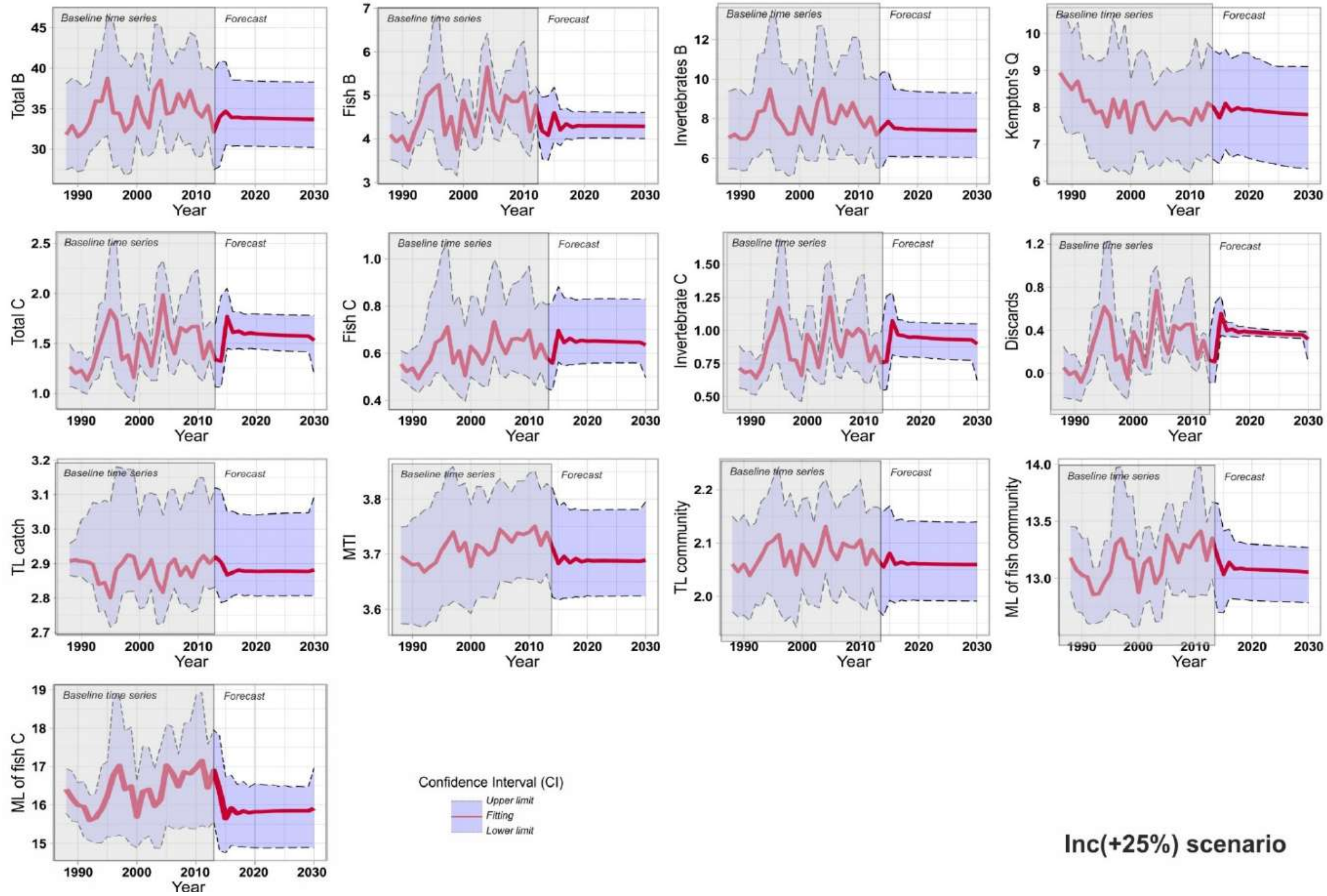
TL community
ML of fish community
ML of fish C
Year
Confidence Interval (CI)
Upper limit
Fitting
Lower limit
Dec(-100%) scenario



Inc(+10%) scenario



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Invertebrates B
Kempton's Q
Total C
Fish C
Invertebrate C
Discards
TL catch
MTI
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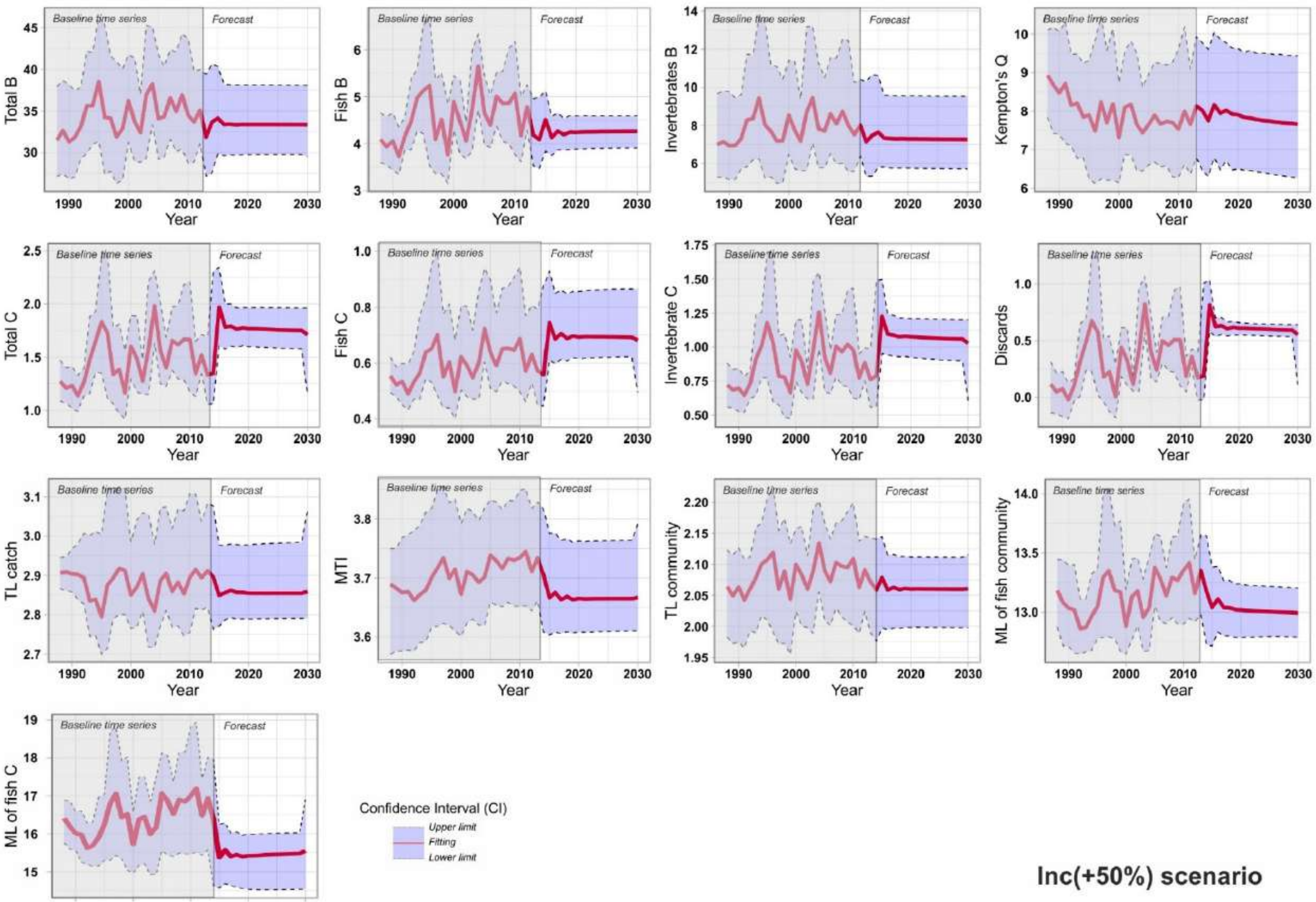
ML of fish community
ML of fish C
Year
Baseline time series
Forecast
Confidence Interval (CI)
Upper limit
Fitting
Lower limit
Inc(+25%) scenario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Forecast
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Total B
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Invertebrates B
Kempton's Q
Total C
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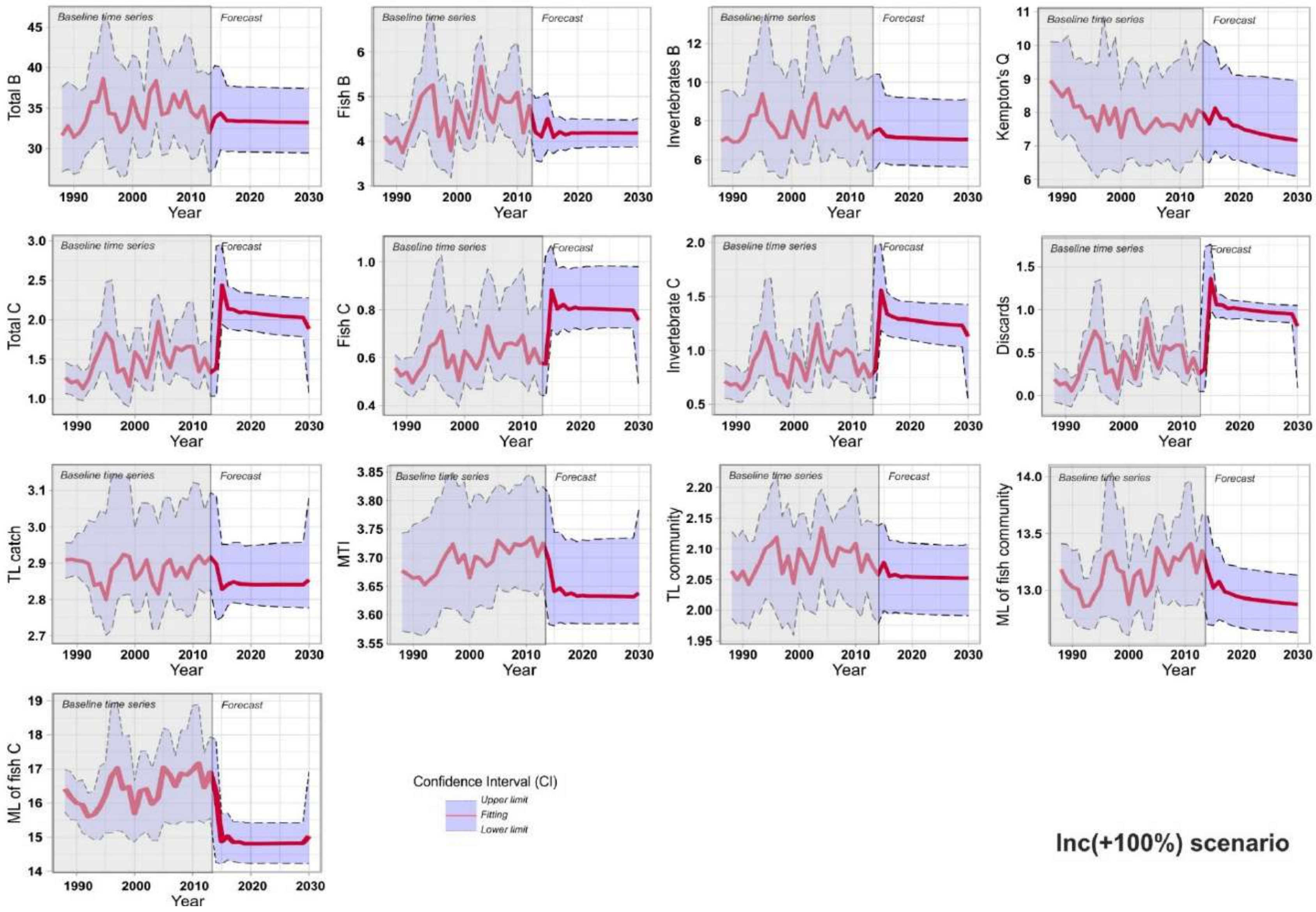
ML of fish community
ML of fish C
Confidence Interval (CI)
Upper limit
Fitting
Lower limit
Inc(+50%) scenario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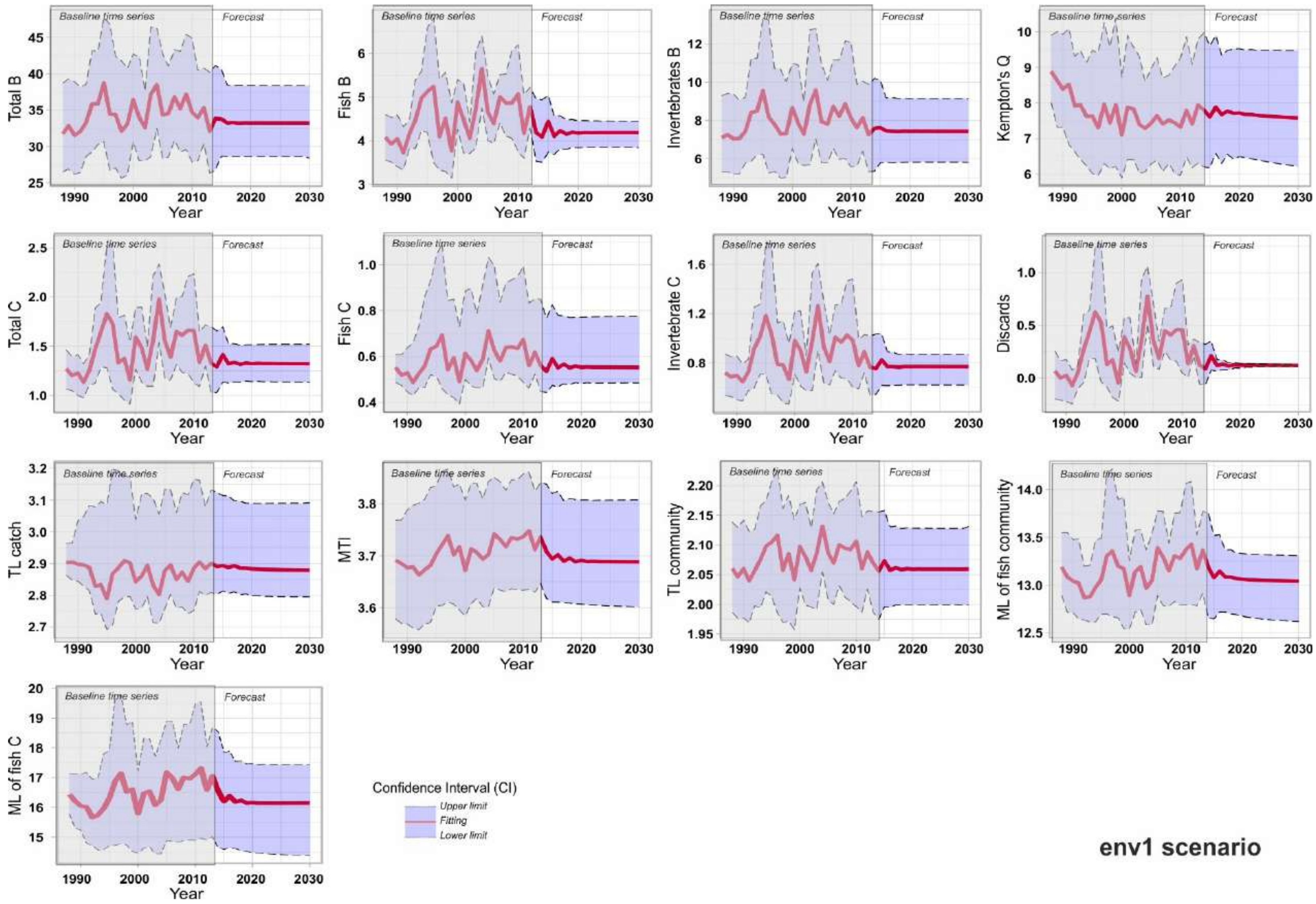
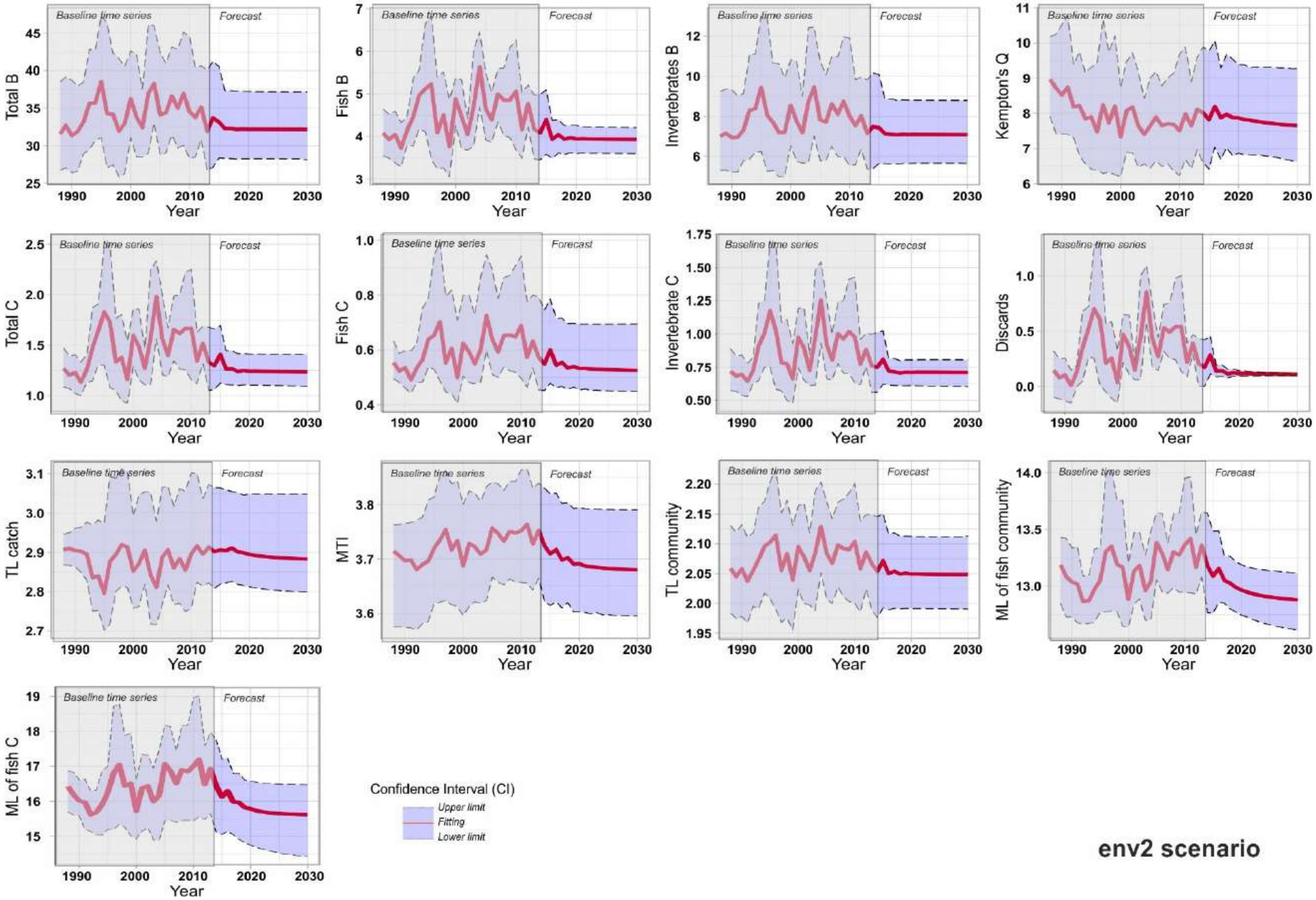
Baseline time series
Forecast
Year
Confidence Interval (CI)
Upper limit
Fitting
Lower limit
Inc(+100%) scenario



env1 scenario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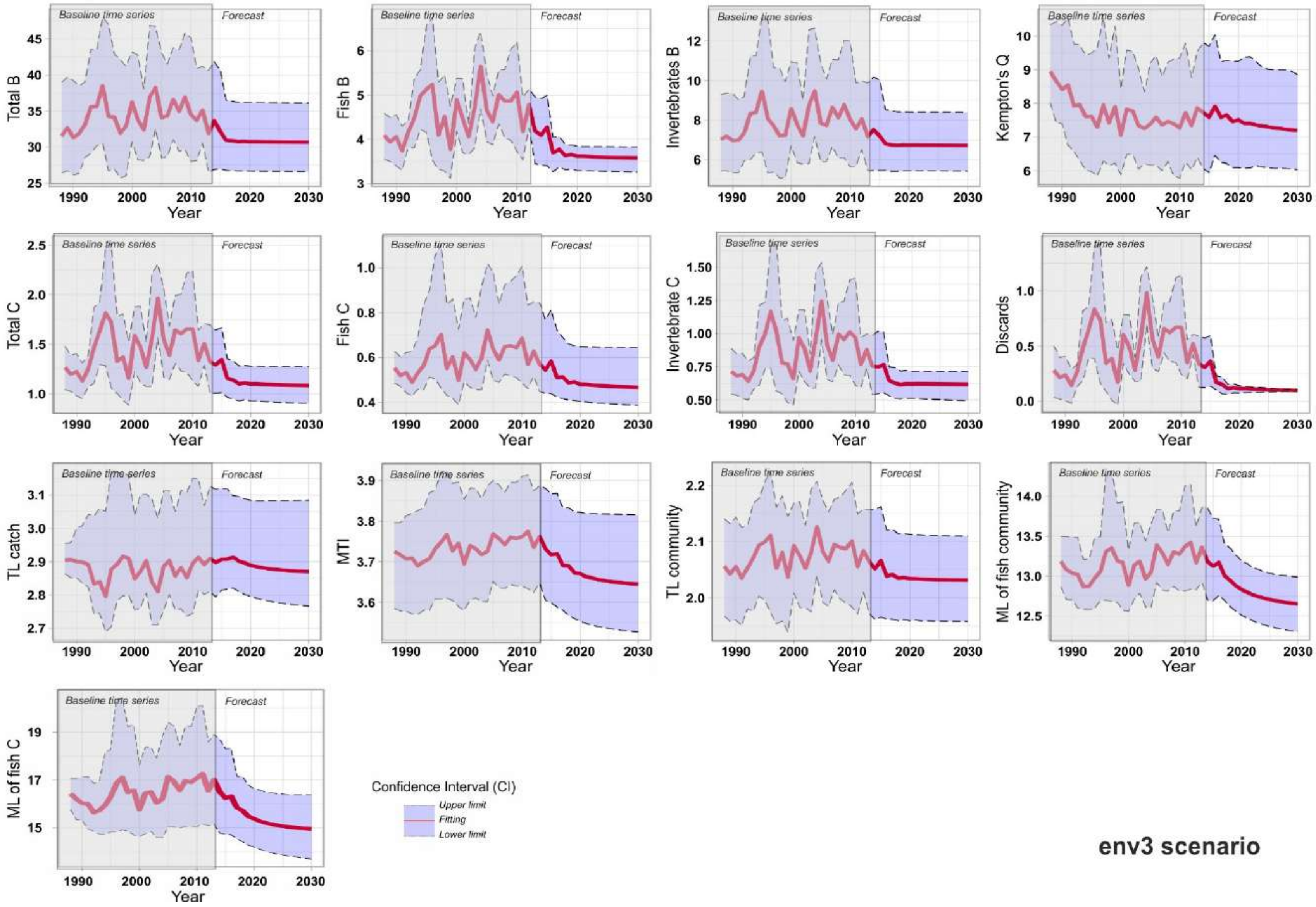
Fish B
Invertebrates B
Kempton's Q
Total C
Fish C
Invertebrate C
Discards
TL catch
MTI
TL community
ML of fish community
ML of fish C
Year
Baseline time series
Forecast
Confidence Interval (CI)
Upper limit
Fitting
Lower limit
env2 scenario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Fish B
Invertebrates B
Kempton's Q
Total C
Fish C
Invertebrate C
Discards
TL catch
MTI
TL community
ML of fish community
ML of fish C
Baseline time series
Forecast
Year
Confidence Interval (CI)
Upper limit
Fitting
Lower limit
env3 scenario

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CHAPTER 4. Vulnerability of marine resources affected by a small-scale tropical shrimp fishery in Northeast Brazil

The supplementary material follows the order according to the manuscript presented in the Chapter 4:

Supplementary material method

Detail about the estimations of the life history parameters used in the present study.

Frequency of length

Figure S1. Frequency of length of the main species caught by bottom trawl fishing

Total mortality

Figure S2. Linearized length converted catch curve to estimate the total mortality

Boundaries of scoring of the productivity attributes

Figure S3. Productivity attributes and rankings used to determine the vulnerability

Redundancy of pairs of life history traits

Figure S4. Bivariate relationships between pairs of life history traits

Difference of the methods to definition of the boundaries of attribute scores

Figure S5. Negative and positive difference in the estimates of productivity, susceptibility and vulnerability between the methods (quantile and k-means)

Productivity input data

Table S1. Input data for the attributes used to estimate the productivity

Susceptibility input data

Table S2. Input data for the attributes used to estimate the susceptibility

Supplementary material method

We presented here, detail about the estimations of the life history parameters used in the present study. When not available in the literature:

The Von Bertalanffy growth coefficient (k; cm.y^{-1}) was estimated using the empirical equation of Le Quesne and Jennings (2012) for Teleostei:

$$K = 2.15 \times L_{\infty}^{-0.46}$$

Where L_{∞} is the asymptotic length estimated from Froese and Binohlan (2000) based in the maximum reported total length to species:

$$\log_{10}(L_{\infty}) = 0.444 + 0.9841 \times \log_{10}(L_{\max})$$

The Size at first maturity (L_{50}; cm) was estimated followed by Binohlan and Froese (2009) based in the maximum reported total length to species:

$$\log L_{50} = -0.1189 + 0.9157 \times \log L_{max}$$

The Age maximum (A_{max}) was estimated from the empirical equation of Taylor (1960):

$$A_{max} = K + \left(\frac{2.996}{t_0}\right)$$

Where K is the Von Bertalanffy growth coefficient (described above) and t_0 is the theoretical age in years which the fish would have at length zero. t_0 was estimated by the empirical equation from Froese and Binohlan (2003):

$$\log_{10}(-t_0) = -0.3922 - (0.2752 \times \log_{10}(L_{\infty})) - (1.038 \times \log_{10}(K))$$

Ratio between fishing mortality and natural mortality (F/M)

The fishing mortality (F) was estimated by the difference between the total mortality (Z) and natural mortality (M). We used the average (M) from nine different empirical relationships by application developed by Jason cope (https://github.com/shcaba/Natural-Mortality-Tool). All relations, except Hamel_Amax are described below:

Then_nls, Then_lm and Then_VBGF by Then et al. (2014):

$$\log(M) = 1.717 - 1.01\log(t_{max})$$

$$M = 4.889{t_{max}}^{-0.916}$$

$$M = 4.118K^{0.73}L_{\infty}^{-0.33}$$

ZM_CA_pel and ZM_CA_dem by Alverson and Carney (1975); Zhang and Megrey (2006):

$$M = {}^{3K}\!/_{(e^{aKt_{max}} - 1)}$$

$$M = \frac{\beta K}{e^{(t_{mb}-t_0)} - 1}$$

Jensen_VBGF1 and Jensen_VBGF2 by Jensen (1996, 1997, 2001):

$$M = 1.5K$$

$$M = 0.21 + 1.47K$$

Pauly_lt by Pauly (1980):

$$\log M = -0.0066 - 0.279\log L_{\infty} + 0.6543\log K + 0.4634\log T$$

Where, tmax is maximum age, K and $L\infty$ are von Bertalanffy growth coefficient and asymptotic size respectively, and water temperature T.

While total mortality was estimated from the Length-based methods (e.g., Catch curve and Powell–Wetherall plot) see detail in (Pauly, 1983; Wetherall, 1986; Schwamborn, 2018). From Powell–Wetherall (P-W), Schwamborn (2018) developed a modified method based in "gamma" selection to minimize effect of subjective manual choice of data points for regression.

Intrinsic growth rate (*r)*

This parameter was estimated from the equation proposed by (Mertz, 1970), using the life table. The estimate of r from this simple approach generally leads to errors of at maximum 10% around the true value (Stearns, 1992).

The life table with survivors by age groups was calculated based on the estimates of the natural mortality (M), developed by Gislason et al. (2010).

$$M = 0.55KL_{\infty}^{1.44}exp^{-1.61logL}$$

Where, L corresponding to length referring to an age (t) (projected from 0 to 100) estimated from Von Bertalanffy (1938):

$$L = L_{\infty}\left[1 - e^{-k(t-t_0)}\right]$$

t_o is theoretical age when size equals zero.

From the length and mortality by age were estimated the survival probability each age to the next (S_p) and the fecundity (F_c) considering the probability of maturity by L_{50} values to each species. To estimate the fecundity, weight by age was used as proxy, where fertility is proportional to weight (L^3) assuming a sex ratio of 1:1 was assumed.

In turn, these values (S_p and F_c) were used to estimate the net reproductive rate (R_0) and the generation time (G), incorporating the age (t). Finally, the intrinsic growth rate (*r*) was obtained from the relation between (R_0) and (G):

$$r = \frac{\log R_0}{G}$$

Considering the limitations of the approach employed, *r* estimates here obtained are useful for the comparative purposes among the species considered, but their use as absolute isolated estimate, should be taken with caution.

Frequency of length

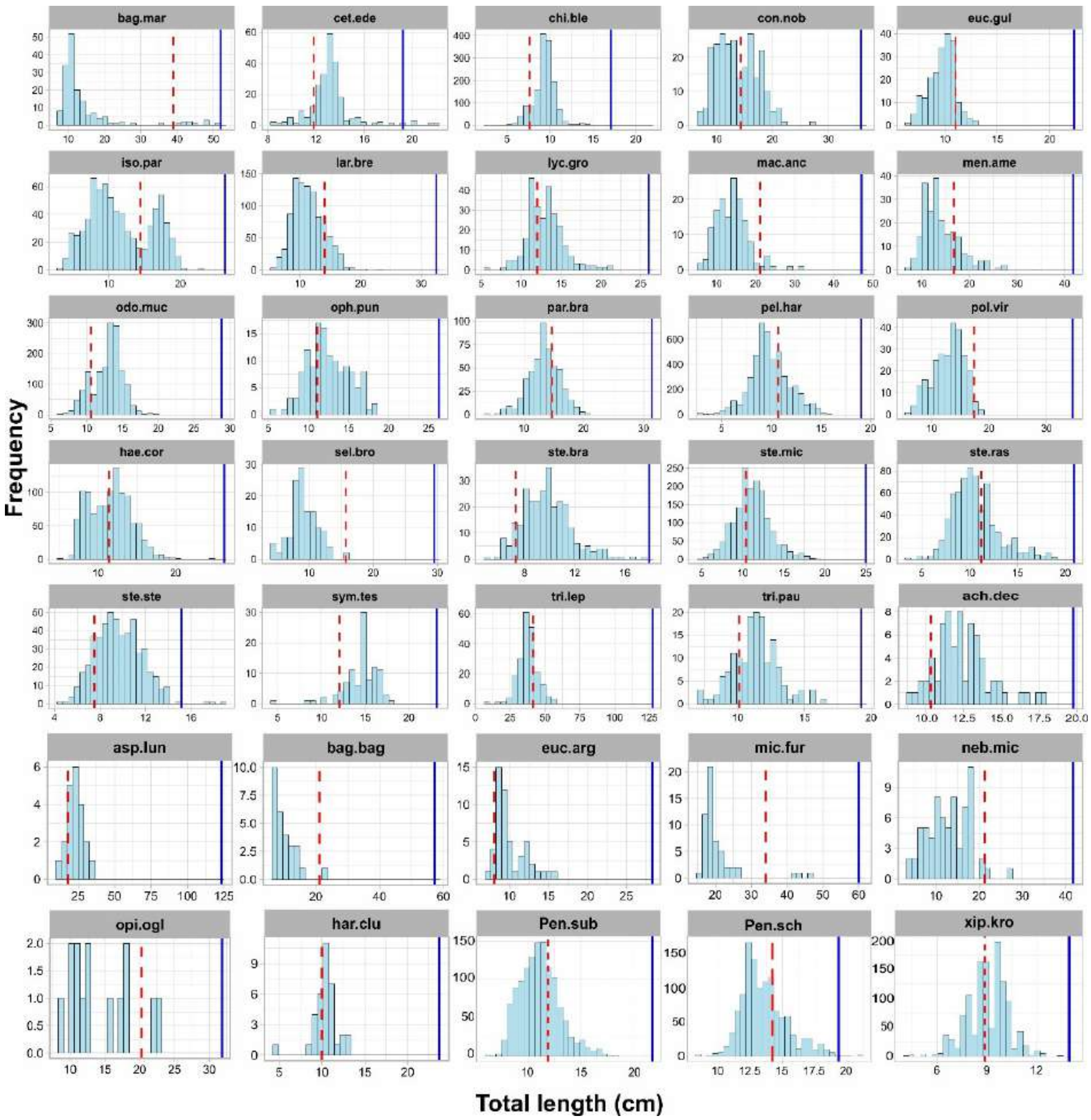


Figure S1. Frequency of length of the main species caught by bottom trawl fishing in BSIR, south of Pernambuco, Northeast Brazil. Blue solid line and red dashed line represents the asymptotic length (L_∞) and size at first maturity (L_{50}), respectively. Species code may be accessed from Table 3 in the manuscript.

Total mortality

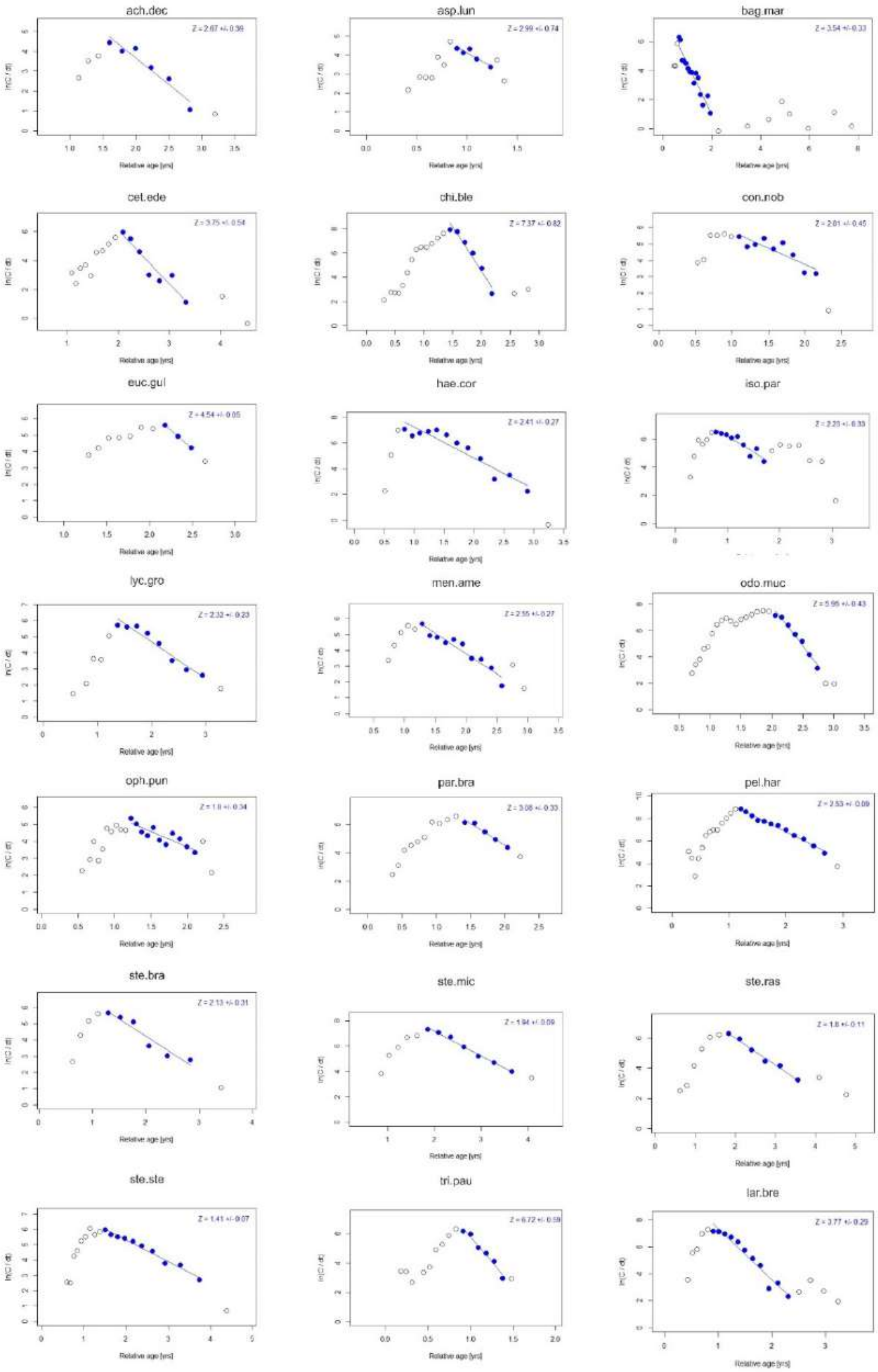
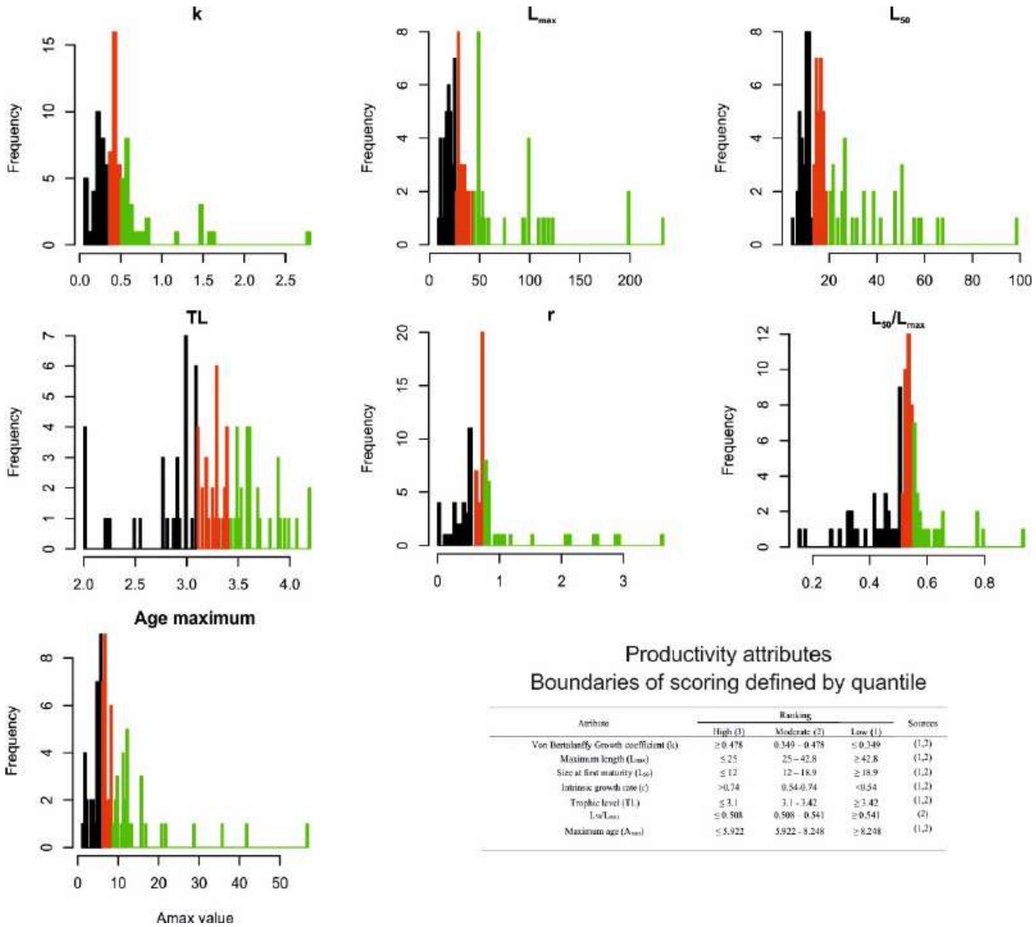
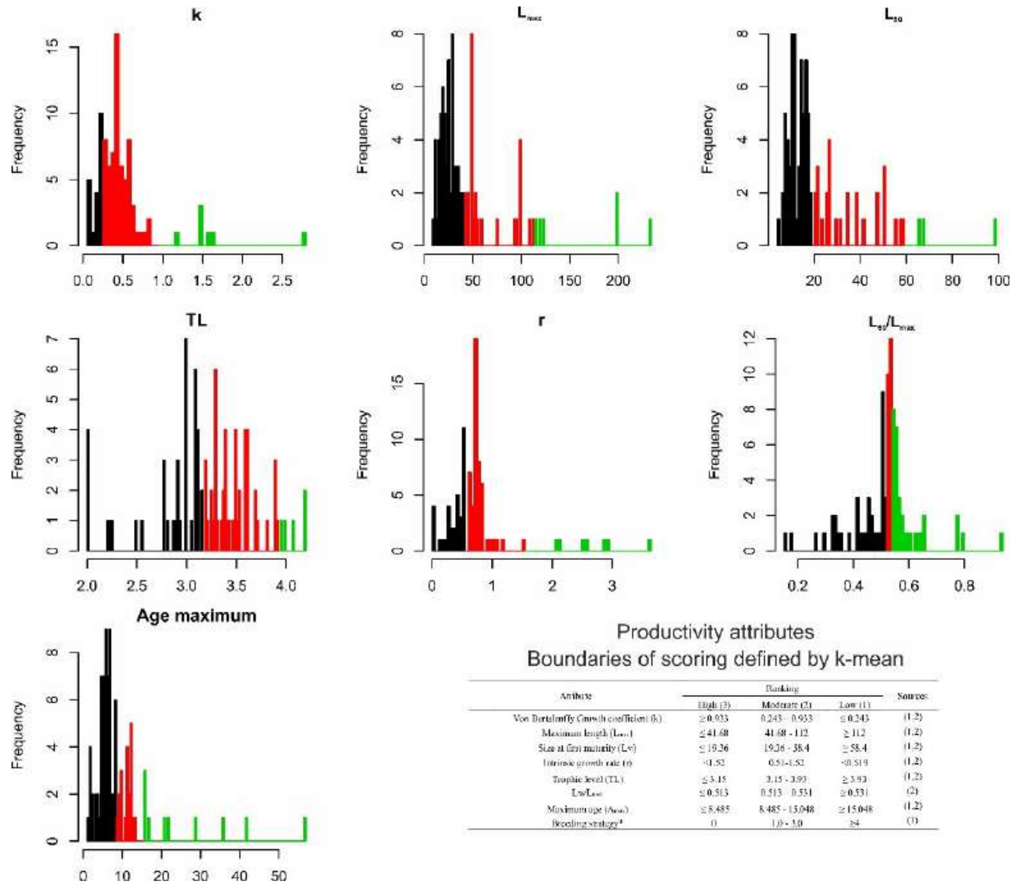


Figure S2. Linearized length converted catch curve to estimate the total mortality (Z ± SE) (Chapman and Robson, 1960; Pauly, 1983) of the fish species caught by bottom trawl fishing in BSIR, south of Pernambuco, Northeast Brazil. Species code may be accessed from Table 3 in the manuscript.

Boundaries of scoring of the productivity attributes



Attribute	Ranking			Sources
	High (3)	Moderate (2)	Low (1)	
Von Bertalanffy Growth coefficient (k)	≥0.478	0.349 - 0.478	≤0.349	(1,2)
Maximum length (L_{max})	≤25	25 – 42.8	≥42.8	(1,2)
Size at first maturity (L_{50})	≤12	12 – 18.9	≥18.9	(1,2)
Intrinsic growth rate (r)	>0.74	0.54-0.74	<0.54	(1,2)
Trophic level (TL)	≤3.1	3.1 - 3.42	≥3.42	(1,2)
L_{50}/L_{max}	≤0.508	0.508 – 0.541	≥0.541	(2)
Maximum age (A_{max})	≤5.922	5.922 - 8.248	≥8.248	(1,2)



Attribute	Ranking			Sources
	High (3)	Moderate (2)	Low (1)	
Von Bertalanffy Growth coefficient (k)	≥0.933	0.243 - 0.933	≤0.243	(1,2)
Maximum length (L_{max})	≤41.68	41.68 - 112	≥112	(1,2)
Size at first maturity (L_{50})	≤19.36	19.36 - 58.4	≥58.4	(1,2)
Intrinsic growth rate (r)	<1.52	0.51-1.52	<0.519	(1,2)
Trophic level (TL)	≤3.15	3.15 - 3.93	≥3.93	(1,2)
L_{50}/L_{max}	≤0.513	0.513 – 0.531	≥0.531	(2)
Maximum age (A_{max})	≤8.485	8.485 - 15.048	≥15.048	(1,2)
Breeding strategy[a]	0	1.0 - 3.0	≥4	(1)

Figure S3. Productivity attributes and rankings used to determine the vulnerability of species caught by bottom trawl fishing in BSIR, south of Pernambuco, Northeast Brazil. Boundaries of scoring defined by quantile and k-means methods.

Redundancy of pairs of life history traits

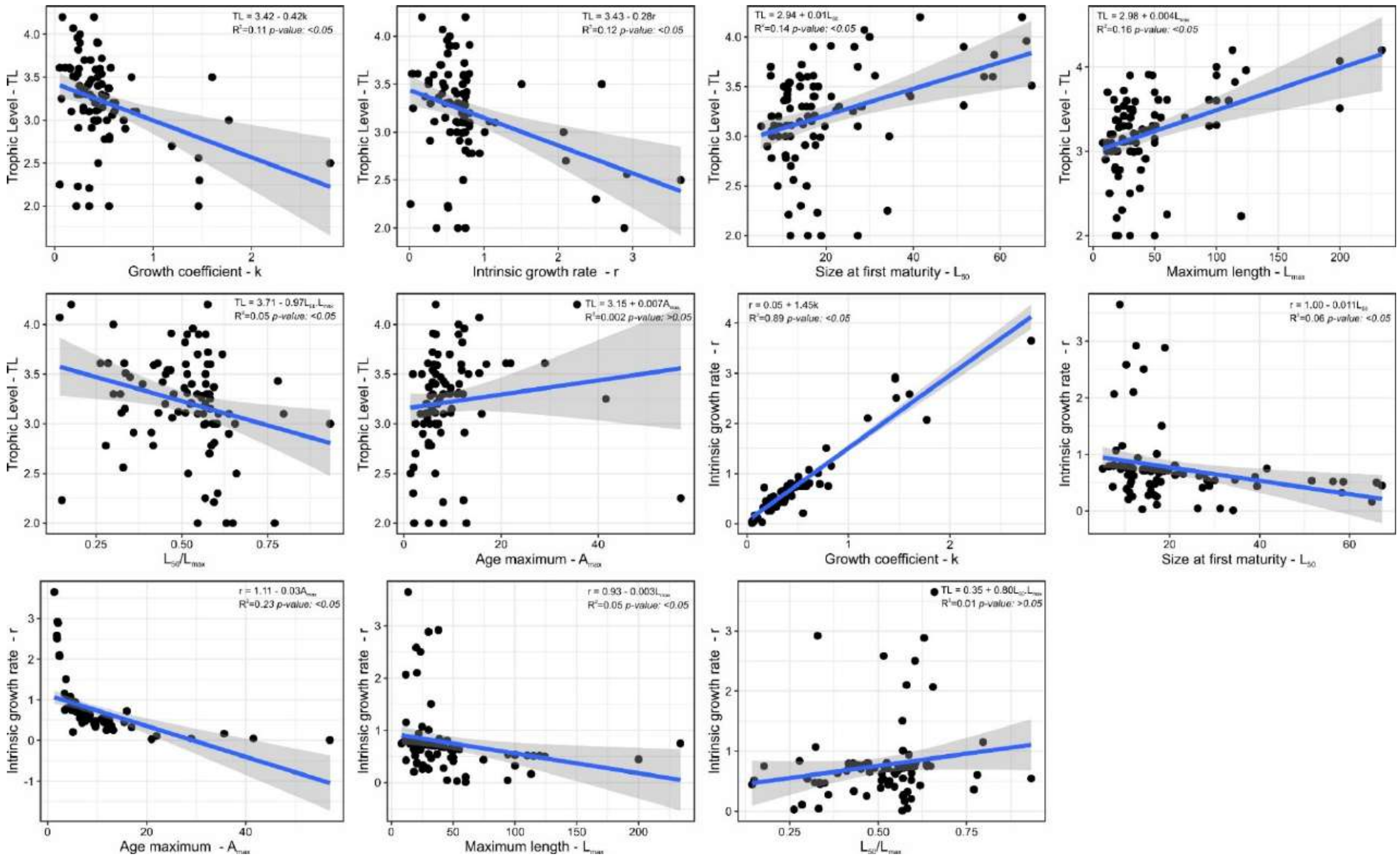


Figure S4. Bivariate relationships between pairs of life history traits. Units: L_{max} (cm), A_{max} (years), k (cm $year^{-1}$), L50 (cm), L_{50}/L_{max} (no unit), Trophic level (no unit), *r* (no unit).

Difference of the methods to definition of the boundaries of attribute scores

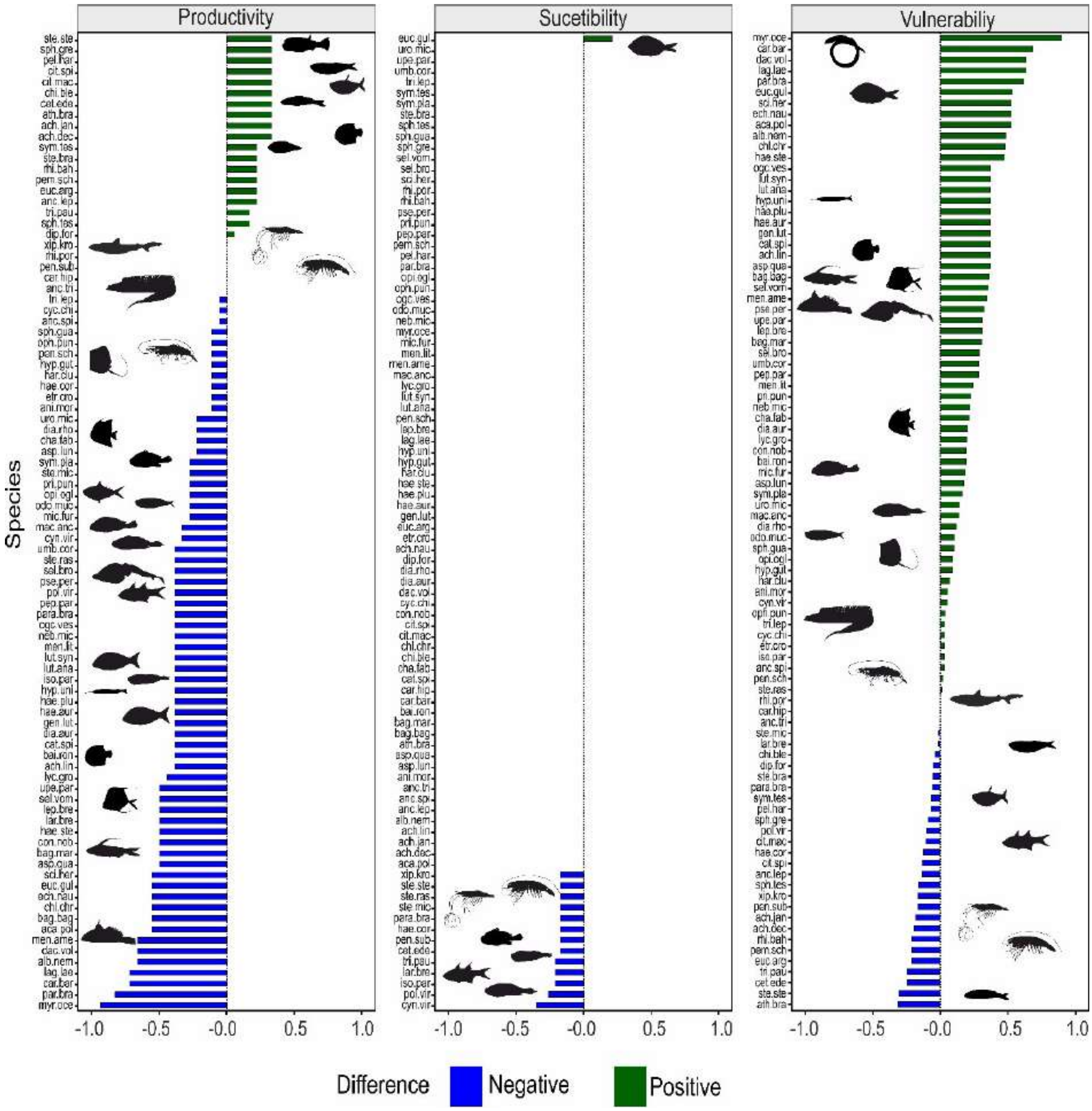


Figure S5. Negative and positive difference in the estimates of productivity, susceptibility and vulnerability between the methods (quantile and k-means) to definition of the boundaries of attribute scores. Species codes are described in Table 3 of the manuscript.

Productivity input data

Table S1. Input data for the attributes used to estimate the productivity of target and non-target species caught by bottom trawl fishing in Sirinhaém, south of Pernambuco, Northeast Brazil. Von Bertalanffy Growth coefficient (k); Maximum length (L_{max}); Size at first maturity (L_{50}); Intrinsic growth rate (r); Trophic level (TL); L_{50}/L_{max} and Maximum age (A_{max}). Values in red were estimated by equation: L∞ (logL∞=0.444+0.9841×logLmax by Froese and Binohlan (2000)); k (K=2.15 x L∞^(-0.46) by Le Quesne and Jennings (2012)); t0 (log-t0=-0.3922-(0.2752×logL∞)-(1.038×logK) by Froese and Binohlan (2003)); Amax (Amax=K+(2.996/t0) by Taylor (1960)); L_{50} (logL50=-0.1189+0.9157×logLmax by Binohlan and Froese (2009)).

Group	Order	Family	Specie	Cod.sp	min-max (mean-cm)	IUCN	Lmax	L∞	k	t0	Amax	L50	L50/Lmax	a	b	r	TL	Sources
fish	Tetraodontiformes	Ostraciidae	*Acanthostracion polygonius*	aca.pol	21.3-21.3 (21.3)	LC	50	51.99	0.35	-0.41	8.18	27.3	0.546	0.0282	2.83	0.65	2	1;133
fish	Pleuronectiformes	Achiridae	*Achirus declivis*	ach.dec	9-17.9 (12.39)	LC	18.7	19.75	0.55	-0.32	5.17	11.1	0.594	0.0102	3.25	0.75	3.37	2;55
fish	Pleuronectiformes	Achiridae	*Achirus lineatus*	ach.lin	9.5-16.3 (12.86)	LC	33.1	34.64	0.42	-0.37	6.75	18.7	0.565	0.0094	3.29	0.71	2.99	2;55
fish	Albuliformes	Albulidae	*Albula nemoptera*	alb.nem	20-21.9 (20.95)	LC	51	53.02	0.35	-0.41	8.25	27.8	0.545	0.0174	2.92	0.65	3.3	3;134
fish	Clupeiformes	Engraulidae	*Anchoa januaria*	ach.jan	-	LC	10.2	10.88	0.72	-0.28	3.9	6.5	0.637	0.0081	3.13	0.79	2.9	4;133;84
fish	Clupeiformes	Engraulidae	*Anchoa spinifer*	anc.spi	6.9-20.2 (10.5)	LC	24	25.25	0.49	-0.34	5.82	14	0.583	0.005	3.18	0.74	3.15	5;55;
fish	Clupeiformes	Engraulidae	*Anchoa tricolor*	anc.tri	8-11.6 (9.88)	LC	11.6	10.06	1.77	-0.21	2.4	7.6	0.655	0.007	3.02	2.07	3	6;55;85
fish	Clupeiformes	Engraulidae	*Anchoviella lepidentostole*	anc.lep	12.1-13 (12.6)	LC	11.8	14.3	0.83	-0.2	3.41	9.4	0.797	0.0054	3.14	1.15	3.1	7;13;133;86
fish	Perciformes	Haemulidae	*Anisotremus moricandi*	ani.mor	16.1-16.5 (16.28)	LC	15.1	31.5	0.44	-0.36	6.45	9.1	0.603	0.0174	3.01	0.74	3	8;133
fish	Siluriformes	Ariidae	*Aspistor luniscutis*	asp.lun	10.9-33.5 (21.07)	LC	120	123.06	0.23	-0.5	12.25	18	0.15	0.004	3.26	0.51	2.23	9;55;87
fish	Siluriformes	Ariidae	*Aspistor quadriscutis*	asp.qua	40.2-40.2 (40.2)	LC	50	51.99	0.35	-0.41	8.17	27.3	0.546	0.006	3.11	0.65	3.1	10;55
fish	Atheriniformes	Atherinopsidae	*Atherinella brasiliensis*	ath.bra	11.7-11.7 (11.7)	LC	16	16.94	0.58	-0.31	4.81	9.1	0.569	0.006	2.97	0.75	3.2	5;135;88
fish	Siluriformes	Ariidae	*Bagre bagre*	bag.bag	6.2-22.9 (10.05)	NT	55	57.11	0.33	-0.42	8.54	21.2	0.385	0.0045	3.09	0.64	3.4	5;55;89
fish	Siluriformes	Ariidae	*Bagre marinus*	bag.mar	7-50.5 (13.38)	DD	50	51.99	0.35	-0.41	8.17	39	0.78	0.0028	3.29	0.61	3.43	55;71
fish	Acanthuriformes	Sciaenidae	*Bairdiella ronchus*	bai.ron	6.6-11.2 (8.86)	LC	35	36.6	0.41	-0.38	6.92	15.8	0.451	0.005	3.33	0.7	3.2	11;55;90
fish	Carangiformes	Carangidae	*Carangoides bartholomaei*	car.bar	13-16 (14.4)	LC	100	102.85	0.26	-0.48	11.26	30	0.3	0.0298	2.71	0.54	4	5;55;91
fish	Carangiformes	Carangidae	*Caranx hippos*	car.hip	7.5-7.7 (7.6)	LC	124	127.1	0.23	-0.51	12.44	66	0.532	0.0126	2.97	0.5	3.96	5;55;92
fish	Carangiformes	Carangidae	*Cathorops spixii*	cat.spi	24.5-24.5 (24.5)	LC	30	31.45	0.44	-0.36	6.45	17.1	0.57	0.0079	3.02	0.72	3.4	10;133
fish	Clupeiformes	Engraulidae	*Cetengraulis edentulus*	cet.ede	8.4-22 (13.08)	LC	18.2	19.23	0.55	-0.32	5.11	11.8	0.648	0.004	2.72	0.75	2	2;55;72
fish	Moroniformes	Ephippidae	*Chaetodipterus faber*	cha.fab	6.1-14.5 (9.56)	LC	20.5	50.88	0.22	-0.75	12.87	15.8	0.771	0.00009	2.82	0.36	2	12;12;12

fish	Clupeiformes	Pristigasteridae	*Chirocentrodon bleekerianus*	chi.ble	2.5-14.5 (9.38)	LC	16.1	17.05	0.58	-0.31	4.82	7.6	0.472	0.002	3.41	0.81	3.06	13;55;81
fish	Carangiformes	Carangidae	*Chloroscombrus chrysurus*	chl.chr	1.7-16.5 (5.16)	LC	48.3	25.45	0.32	0.06	9.42	15.5	0.321	0.011	2.93	0.48	3.3	93;55;93
fish	Pleuronectiformes	Paralichthyidae	*Citharichthys macrops*	cit.mac	11.5-11.5 (11.5)	LC	20	21.1	0.53	-0.33	5.34	11.8	0.59	0.0062	3.2	0.75	3	3;136
fish	Pleuronectiformes	Paralichthyidae	*Citharichthys spilopterus*	cit.spi	4-21 (12.74)	LC	21	22.14	0.52	-0.33	5.46	11.7	0.557	0.005	3.2	0.75	3.28	13;55;94
fish	Perciformes	Haemulidae	*Conodon nobilis*	con.nob	6.6-26.9 (13.7)	LC	34.2	35.78	0.41	-0.37	6.85	14.3	0.418	0.0096	3.14	0.72	3.59	14;55;56
fish	Pleuronectiformes	Paralichthyidae	*Cyclopsetta chittendeni*	cyc.chi	7.5-7.5 (7.5)	LC	32	33	0.78	-0.16	3.68	18.2	0.569	0.0079	3.13	1.51	3.5	15;5;133
fish	Pleuronectiformes	Paralichthyidae	*Cynoscion virescens*	cyn.vir	8.3-30.5 (15.61)	LC	115	118.01	0.24	-0.5	12.01	58.6	0.51	0.0108	2.86	0.51	3.82	16;55
fish	Scorpaeniformes	Dactylopteridae	*Dactylopterus volitans*	dac.vol	-	LC	50	33.58	0.3	-0.57	9.42	27.3	0.546	0.0071	3.1	0.41	3.7	17;62;137
fish	Perciformes	Gerreidae	*Diapterus auratus*	dia.aur	8.2-19.5 (12.98)	LC	42.8	44.6	0.37	-0.39	7.6	17.6	0.411	0.01	3.09	0.67	2.91	18;55;74
fish	Perciformes	Gerreidae	*Diapterus rhombeus*	dia.rho	8.4-14.5 (10.79)	LC	42.3	26.25	0.24	-0.86	12.48	15.2	0.359	0.009	3.16	0.28	2.91	19;55;82
fish	Perciformes	Serranidae	*Diplectrum formosum*	dip.for	9.2-9.2 (9.2)	LC	30	20.4	0.7	-0.23	4.05	17.1	0.57	0.0091	3.1	1.01	3	20;3;133
fish	Syngnathiformes	Echeneidae	*Echeneis naucrates*	ech.nau	74.5-74.5 (74.5)	LC	74.5	60.3	0.25	-0.59	11.39	39.4	0.529	0.0028	3.15	0.44	3.4	21;133
fish	Pleuronectiformes	Paralichthyidae	*Etropus crossotus*	etr.cro	4.2-13.5 (10.79)	LC	20	17	1.6	-0.03	1.85	10.32	0.516	0.0073	3.09	2.59	3.5	22;63;55;95
fish	Perciformes	Gerreidae	*Eucinostomus argenteus*	euc.arg	6.6-15.6 (9.86)	LC	24.8	28.31	0.61	-0.24	4.67	8.03	0.324	0.008	3.15	1.07	3.11	23;55;96
fish	Perciformes	Gerreidae	*Eucinostomus gula*	euc.gul	6.7-12.8 (9.7)	LC	25.5	22.3	0.29	-0.69	9.64	11	0.431	0.007	3.3	0.33	3.11	24;64;55;97
invertebrate	Decapoda	Peneidae	*Farfantepenaeus subtilis*	far.sub	7-18.5 (11.15)	LC	20.5	21.62	1.19	-0.08	2.38	11.9	0.58	0.0102	2.87	2.1	2.7	140
fish	Perciformes	Haemulidae	*Genyatremus luteus*	gen.lut	8.5-17.3 (13.5)	LC	37	38.66	0.4	-0.38	7.11	34.5	0.932	0.0119	3.19	0.55	3	18;2;98
fish	Perciformes	Haemulidae	*Haemulon aurolineatum*	hae.aur	8.1-12.3 (10.35)	LC	25	24.2	0.23	-0.93	11.88	11.7	0.468	0.0148	3	0.26	3.54	25;65;133;99
fish	Perciformes	Haemulidae	*Haemulon plumierii*	hae.plu	9.4-9.4 (9.4)	DD	53	41.8	0.15	-0.08	21	13.9	0.262	0.0167	2.98	0.03	3.61	26;16;55;99
fish	Perciformes	Haemulidae	*Haemulon steindachneri*	hae.ste	12-14.7 (13.08)	LC	30	31	0.21	-1	13.26	17.1	0.57	0.0103	3.15	0.25	3.5	24;65;137
fish	Perciformes	Haemulidae	*Haemulopsis corvinaeformis*	hae.cor	5.2-25 (11.49)	LC	25	26.29	0.48	-0.35	5.92	11.45	0.458	0.0093	3.15	0.77	3.54	27;55
fish	Clupeiformes	Clupeidae	*Harengula clupeola*	har.clu	4.7-12.8 (10.37)	LC	18	23.68	0.43	-0.41	6.56	10.7	0.594	0.003	3.52	0.64	3	17;61;55
fish	Myliobatiformes	Dasyatidae	*Hypanus guttatus*	hyp.gut	61.5-93 (77.25)	LC	200	203.44	0.19	-0.57	15.5	67.2	0.336	0.0123	3.06	0.45	3.51	28;133;28
fish	Beloniformes	Hemiramphidae	*Hyporhamphus unifasciatus*	hyp.uni	15.9-17.6 (16.65)	NT	30	30.4	1.46	0.05	2.1	18.9	0.63	0.0984	1.69	2.89	2	5;55;25
fish	Acanthuriformes	Sciaenidae	*Isopisthus parvipinnis*	iso.par	3.4-22.7 (11.56)	LC	25	26.29	0.48	-0.35	5.92	14.4	0.576	0.0056	3.19	0.74	3.72	18;55;73
fish	Tetraodontiformes	Tetraodontidae	*Lagocephalus laevigatus*	lag.lae	4.9-7 (5.95)	LC	100	102.85	0.26	-0.48	11.26	51.6	0.516	0.0232	2.89	0.53	3.31	29;137
fish	Carangiformes	Sciaenidae	*Larimus breviceps*	lar.bre	5.5-23.2 (11.38)	LC	31	32.48	0.43	-0.37	6.54	14.04	0.453	0.0075	3.16	0.72	3.5	5;55;73

fish	Ophidiiformes	Ophidiidae	*Lepophidium brevibarbe*	lep.bre	13.3-13.3 (13.3)	DD	27.3	28.66	0.46	-0.35	6.17	15.7	0.575	0.0023	3.04	0.73	3.6	30;133
invertebrate	Decapoda	Peneidae	*Litopenaeus schmitti*	lit.sch	8.5-20.9 (13.57)	DD	23.5	19.3	1.47	-0.04	1.91	14.2	0.604	0.0092	2.94	2.51	2.3	139;139;139
fish	Perciformes	Lutjanidae	*Lutjanus analis*	lut.ana	21.6-21.6 (21.6)	NT	94	84.5	0.05	-1.8	29	31.22	0.332	0.0108	3.17	0.04	3.61	25;16;55;101
fish	Perciformes	Lutjanidae	*Lutjanus synagris*	lut.syn	7.2-10.9 (9.08)	NT	60	46.8	0.11	-1	22	17.1	0.285	0.0125	3.08	0.11	3.61	25;16;55;102
fish	Clupeiformes	Engraulidae	*Lycengraulis grossidens*	lyc.gro	5.5-21.5 (12.97)	LC	23.5	26	0.42	-0.69	6.44	12	0.511	0.004	3.22	0.63	3.11	31; 57;55;75
fish	Acanthuriformes	Sciaenidae	*Macrodon ancylodon*	mac.anc	5.3-32.1 (13.99)	LC	45	47.1	0.43	-0.33	6	21.13	0.47	0.0056	3.08	0.81	3.91	32;18;55;76
fish	Acanthuriformes	Sciaenidae	*Menticirrhus americanus*	men.ame	7.5-27.5 (14.04)	DD	50	41.8	0.29	-0.52	9.81	16.7	0.334	0.0045	3.28	0.48	3.15	33;18;55;77
fish	Acanthuriformes	Sciaenidae	*Menticirrhus littoralis*	men.lit	9-15.4 (12.22)	DD	48.3	50.25	0.35	-0.41	8.04	23	0.476	0.0083	3.04	0.66	3.3	16;133;78
fish	Acanthuriformes	Sciaenidae	*Micropogonias furnieri*	mic.fur	15-47 (20.77)	LC	60	53.1	0.05	-2.11	57	34.1	0.568	0.0056	3.19	0.01	2.25	34;58;55;79
fish	Anguilliformes	Ophichthidae	*Myrichthys ocellatus*	myr.oce	-	LC	110	112.96	0.24	-0.49	11.77	56.3	0.512	0.0015	2.9	0.52	3.6	35;133
fish	Acanthuriformes	Sciaenidae	*Nebris microps*	neb.mic	4.3-27.5 (13.02)	LC	40	41.74	0.39	-0.39	7.37	22.3	0.558	0.0094	3	0.68	3.26	36;55
fish	Clupeiformes	Pristigasteridae	*Odontognathus mucronatus*	odo.muc	6-19.9 (12.89)	LC	19.2	28.8	0.35	-0.49	8.07	11.4	0.594	0.0281	2.23	0.52	2.21	37;59;55
fish	Lophiiformes	Ophichthidae	*Ogcocephalus vespertilio*	ogc.ves	18.1-18.1 (18.1)	LC	30.5	31.97	0.44	-0.36	6.5	17.4	0.57	0.0302	2.61	0.72	3.4	38;137
fish	Acanthuriformes	Sciaenidae	*Ophioscion punctatissimus*	oph.pun	5.9-18.4 (12.48)	DD	25	26.29	0.48	-0.35	5.92	11.1	0.444	0.0062	3.28	0.77	3.42	11;55;74
fish	Anguilliformes	Clupeidae	*Opisthonema oglinum*	opi.ogl	9-23.1 (14.52)	LC	38	31.8	1.46	-0.06	1.99	12.5	0.329	0.0081	3.01	2.92	2.56	25;5;55;104
fish	Pleuronectiformes	Paralichthyidae	*Paralichthys brasiliensis*	para.bra	6.3-20.2 (11.55)	LC	100	102.85	0.26	-0.48	11.26	51.6	0.516	0.0018	3.56	0.53	3.9	39;55
fish	Acanthuriformes	Sciaenidae	*Paralonchurus brasiliensis*	par.bra	4.1-20.7 (13.35)	LC	30	31.45	0.44	-0.36	5.6	14.7	0.49	0.0023	3.47	0.72	3.12	40;11;55;73
fish	Clupeiformes	Pristigasteridae	*Pellona harroweri*	pel.har	2.6-16.5 (9.86)	LC	18	19.02	0.55	-0.32	5.08	10.7	0.594	0.0102	3.02	0.75	2.81	5;55;74
fish	Perciformes	Pempheridae	*Pempheris schomburgkii*	pem.sch	10-10 (10)	LC	15	15.9	0.6	-0.31	4.67	9.1	0.607	0.0159	2.95	0.81	3.1	38;133
fish	Perciformes	Stromateidae	*Peprilus paru*	pep.par	5.7-15 (7.11)	LC	30	31.45	0.44	-0.36	6.45	15.56	0.519	0.0152	3.05	0.72	2.5	38;137;105
fish	Perciformes	Polynemidae	*Polydactylus virginicus*	pol.vir	6.1-19 (12.94)	LC	33	34.54	0.42	-0.37	6.74	17.4	0.527	0.0065	3.13	0.71	3.21	42;55;74
fish	Scorpaeniformes	Triglidae	*Prionotus punctatus*	pri.pun	9-14.5 (11.33)	LC	45	52.7	0.07	-3.16	41.55	26.2	0.582	0.0116	2.96	0.05	3.25	43;66;137
fish	Rajiformes	Rhinobatidae	*Pseudobatos percellens*	pse.per	23-23 (23)	DD	100	109.31	0.16	-1.78	16.95	58.3	0.583	0.0031	3.06	0.32	3.6	44;5;133;107
fish	Clupeiformes	Clupeidae	*Rhinosardinia bahiensis*	rhi.bah	9.3-10.3 (9.8)	LC	8	8.57	0.8	-0.26	3.48	5.1	0.638	0.0111	2.89	0.75	3.1	45;55
fish	Carcharhiniformes	Carcharhinidae	*Rhizoprionodon porosus*	rhi.por	21.7-21.7 (21.7)	DD	113	136.4	0.08	-3.27	35.64	65	0.575	0.0021	3.1	0.17	4.2	46;67;133;108
fish	Siluriformes	Ariidae	*Sciades herzbergii*	sci.her	4-15.7 (9)	LC	94.2	96.98	0.26	-0.47	10.95	28.3	0.3	0.0059	3.11	0.54	3.3	48;55;74
fish	Carangiformes	Carangidae	*Selene brownii*	sel.bro	4.1-10 (5.95)	LC	29	29.5	0.23	-0.87	12.15	16.6	0.572	0.0123	3.03	0.28	3.3	24;5;55

fish	Carangiformes	Carangidae	*Selene vomer*	sel.vom	2.6-26.4 (7.21)	LC	48.3	31.5	0.43	-0.37	6.59	26.5	0.549	0.0167	2.93	0.62	3.9	24;18;55
fish	Tetraodontiformes	Tetraodontidae	*Sphoeroides greeleyi*	sph.gre	19.4-19.4 (19.4)	LC	18	19.02	0.55	-0.32	5.08	7.5	0.417	0.0217	2.87	0.8	2.78	5;55;83
fish	Tetraodontiformes	Tetraodontidae	*Sphoeroides testudineus*	sph.tes	9.2-9.2 (9.2)	DD	38.8	30	0.51	-0.31	5.57	10.8	0.278	0.0213	2.93	0.84	2.78	49;38;55;110
fish	Perciformes	Sphyraenidae	*Sphyraena guachancho*	sph.gua	6.9-34 (15.27)	LC	200	203.44	0.19	-0.57	15.5	28.8	0.144	0.0094	2.76	0.44	4.07	50;55;111
fish	Acanthuriformes	Sciaenidae	*Stellifer brasiliensis*	ste.bra	4.9-17.5 (9.87)	LC	17	17.98	0.57	-0.32	4.95	7.3	0.429	0.0096	3.03	0.8	3.61	13;55;68
fish	Acanthuriformes	Sciaenidae	*Stellifer microps*	ste.mic	5.1-19.5 (11.17)	LC	20.5	24.96	0.3	-0.63	9.66	10.4	0.507	0.0058	3.26	0.39	3.36	51;11;55;73
fish	Acanthuriformes	Sciaenidae	*Stellifer rastrifer*	ste.ras	3.8-19 (10.61)	LC	32.1	20.9	0.37	-0.49	7.6	11.2	0.349	0.005	3.36	0.47	3.47	13;55;74
fish	Acanthuriformes	Sciaenidae	*Stellifer stellifer*	ste.ste	4.4-18.8 (9.61)	LC	14.3	15.17	0.62	-0.3	4.57	7.5	0.524	0.0059	3.26	0.81	3.2	52;55;68
fish	Pleuronectiformes	Cynoglossidae	*Symphurus plagusia*	sym.pla	3-18.5 (10.28)	LC	25	26.29	0.48	-0.35	5.92	14.5	0.58	0.0067	3.21	0.74	3.3	36;55
fish	Pleuronectiformes	Cynoglossidae	*Symphurus tessellatus*	sym.tes	4.5-18.4 (14.53)	LC	22	23.18	0.51	-0.34	5.58	12.9	0.586	0.0033	3.29	0.94	2.78	13;55
fish	Pleuronectiformes	Achiridae	*Trichiurus lepturus*	tri.lep	10-58.4 (37.78)	LC	234	127.4	0.4	-0.98	6.53	41.6	0.178	0.0001	3.41	0.75	4.2	53;38;55;80
fish	Pleuronectiformes	Achiridae	*Trinectes paulistanus*	tri.pau	7-16.6 (11.26)	LC	18.2	19.23	0.55	-0.32	5.11	10.8	0.593	0.0082	3.33	0.21	3.37	13;55
fish	Acanthuriformes	Sciaenidae	*Umbrina coroides*	umb.cor	14.1-14.1 (14.1)	LC	35	36.6	0.17	-1.63	16	19.7	0.563	0.0066	3.2	0.72	3.1	24;18;137
fish	Perciformes	Mullidae	*Upeneus parvus*	upe.par	13.1-13.1 (13.1)	LC	30	31.45	0.44	-0.36	6.45	17.1	0.57	0.0044	3.31	0.55	3.9	35;137
fish	Myliobatiformes	Urotrygonidae	*Urotrygon microphthalmum*	uro.mic	10.7-10.7 (10.7)	DD	11.8	28.13	0.36	-1.39	6.96	7.3	0.619	0.0098	3.08	0.43	3.7	54;69;133
invertebrate	Decapoda	Peneidae	*Xiphopenaeus kroyeri*	xip.kro	4-13.5 (9)	DD	13.5	14	2.8	0.07	1.34	8.9	0.659	0.0069	2.91	3.65	2.5	141

1 (Menezes and Figueiredo, 1980) ; 2 (Joyeux et al., 2009); 3 (Robins and Ray, 1986) ; 4 (Franco et al., 2014b) ; 5 (Cervigón et al., 1992) ; 6 (Carvalho, 2014) ; 7 (Camara et al., 2001) ; 8 (Moura et al., 1999) ; 9 (Burgess, 2004) ; 10 (Carpenter, 2002) ; 11 (Chao, 1978) ; 12 (Soeth et al., 2019) ; 13 (Barreto et al., 2018) ; 14 (Pombo et al., 2014) ; 15 (Pauly, 1994) ; 16 (IGFA, 2001) ; 17 (da Costa et al., 2018) ; 18 (Cervigón, 1993) ; 19 (Elliff et al., 2013); 20 (Bubley and Pashuk, 2010) ; 21 (Bachman et al., 2018); 22 (Rábago-Quiroz et al., 2008); 23 (Silva et al., 2014); 24 (García and Duarte, 2006); 25 (Lessa et al., 2004); 26 (Vasconcelos-Filho et al., 2018); 27 (Eduardo et al., 2018); 28 (da Silva et al., 2018); 29 (Shipp, 1981); 30 (Robins et al., 2012); 31 (Goulart et al., 2007); 32 (Ikeda, 2003); 33 (Giannini and Paiva-Filho, 1992); 34 (Santos, 2015); 35 (Smith, 1997); 36 (Keith et al., 2000); 37 (Silva-Júnior, 2004); 38 (Claro, 1994); 39 (Carvalho-Filho, 1992); 40 (Dos S. Lewis and Fontoura, 2005); 42 (Motomura, 2004); 43 (Andrade, 2004); 44 (Caltabellotta et al., 2019); 45 (Whitehead et al., 1988); 46 (Lessa and Santana, 1998); 48 (Chacon et al., 1994); 49 (Pauly, 1991); 50 (Reiner, 1996) ; 51 (Sarmento, 2015); 52 (Dias et al., 2017); 53 (Al-Nahdi et al., 2009); 54 (Santander Neto, 2015); 55 (Viana et al., 2016); 56 (Lira et al., 2019); 57 (Kullander and Ferraris, 2003); 58 (Nakamura et al., 1986); 59 (Freire et al., 2009); 60 (Passos et al., 2012); 61 (Lieske and Myers, 1994); 62 (Roux, 1986); 63 (Hensley, 1995); 64 (Amador-del Ángel et al., 2015); 65 (Robins and Ray, 1986); 66 (Teixeira and Haimovici, 1989); 67 (Motta et al., 2014); 68 (Trindade-Santos and Freire, 2015); 69 (Uyeno et al., 1983); 71 (Lima et al., 2016); 72 (Souza-Conceição et al., 2005); 73 (Silva Júnior et al., 2015); 74 (Conceição, 2017); 75 (Mai and Vieira, 2013); 76 (Cardoso et al., 2018); 77 (Freitas et al., 2011); 78 (Braun and Fontoura, 2004); 79 (Santos et al., 2015); 80 (Barreto et al., 2017); 81 (Corrêa et al., 2005); 82 (Bezerra et al., 2001); 83 (Schultz et al., 2002); 84 (Esper, 1982); 85 (Silva Júnior et al., 2013); 86 (Giamas et al., 1985); 87 (Mishima and Tanji, 1983); 88 (Bervian and Fontoura, 1997); 89 (Véras and Da Silva Almeida, 2016); 90 (Torres Castro et al., 1999); 91 (Santos, 2012); 92 (García-Cagide et al., 1994); 93 (de Queiroz et al., 2018); 94 (Dias et al., 2005); 95 (Oliveira and Favaro, 2011); 96 (Leão, 2016); 97 (Mexicano-Cíntora, 1999); 98 (Gómez et al., 2002); 99 (Cardoso de Melo et al., 2020); 101 (Teixeira et al., 2010); 102 (Viana et al., 2015);104 (Simoni, 2019); 105 (Cerqueira and Haimovici, 1990); 107 (Rocha and Gadig, 2013); 108 (Mattos et al., 2001); 110 (Rocha et al., 2002); 111 (Akadje et al., 2019); 112 (Costa et al., 2016); 113 (Franco et al., 2014a); 114 (Pinheiro et al., 2006); 116 (Chaves et al., 2017); 117 (Juras and Yamaguti, 1989); 118 (Martins and Haimovici, 2000); 119 (Souza et al., 1988); 120 (Batista, 2012); 121 (Bervian and Fontoura, 2007); 124 (Alfaro-Martínez et al., 2016); 125 (Gomes et al., 1999); 126 (Freitas, 2017); 127 (Shinozaki-Mendes et al., 2013); 128 (Gaichas et al., 2017); 131 (Isaac-Nahum et al., 1988); 132 (López et al., 2015); 133 (Froese et al., 2014); 134 (Garcia et al., 1998); 135 (da Costa et al., 2014); 137 (Vianna et al., 2004); 138 (Vaz-dos-Santos and Rossi-Wongtschowski, 2013); 139 (Silva et al., 2018); 140 (Silva et al., 2015); 141 (Lopes et al., 2014)

Susceptibility input data

Table S2. Input data for the attributes used to estimate the susceptibility of target and non-target species caught by bottom trawl fishing in Sirinhaém, south of Pernambuco, Northeast Brazil. Frequency of occurrence (FO); total mortality (Z); fishing mortality (F); natural mortality (M); Spawning Potential Ratio (SPR) and percentage of adults in catches (%Adults).

Group	Order	Family	Specie	id.sp	FO	Z	F	M	F/M	SPR	% Adults	Vertical distribution	Guild	ref
fish	Tetraodontiformes	Ostraciidae	*Acanthostracion polygonius*	aca.pol	rare and Scarce			0.563				reef-associated	MS	30
fish	Pleuronectiformes	Achiridae	*Achirus declivis*	ach.dec	frequent and Scarce	2.67	1.596	1.074	1.486	0.416	0.841	demersal	ES	1
fish	Pleuronectiformes	Achiridae	*Achirus lineatus*	ach.lin	rare and Scarce			0.685				demersal	ES	1
fish	Albuliformes	Albulidae	*Albula nemoptera*	alb.nem	rare and Scarce			0.558				demersal	MS	28
fish	Clupeiformes	Engraulidae	*Anchoa januaria*	ach.jan	rare and Scarce			1.203				pelagic	MM	29
fish	Clupeiformes	Engraulidae	*Anchoa spinifer*	anc.spi	frequent and Scarce			0.968				pelagic	MM	1
fish	Clupeiformes	Engraulidae	*Anchoa tricolor*	anc.tri	rare and Scarce			2.64				pelagic	MM	3
fish	Clupeiformes	Engraulidae	*Anchoviella lepidentostole*	anc.lep	rare and Scarce			1.352				pelagic	MM	31
fish	Perciformes	Haemulidae	*Anisotremus moricandi*	ani.mor	rare and Scarce			0.717				demersal	MS	2
fish	Siluriformes	Ariidae	*Aspistor luniscutis*	asp.lun	rare and Scarce	2.99	2.618	0.372	7.038		0.857	demersal	MS	4
fish	Siluriformes	Ariidae	*Aspistor quadriscutis*	asp.qua	rare and Scarce			0.563				demersal	MS	4
fish	Atheriniformes	Atherinopsidae	*Atherinella brasiliensis*	ath.bra	rare and Scarce			0.969				pelagic	ES	1
fish	Siluriformes	Ariidae	*Bagre bagre*	bag.bag	rare and Scarce			0.538			0.036	demersal	MM	6
fish	Siluriformes	Ariidae	*Bagre marinus*	bag.mar	frequent and Higher Abundant	3.54	2.977	0.563	5.288		0.06	demersal	MM	5
fish	Acanthuriformes	Sciaenidae	*Bairdiella ronchus*	bai.ron	rare and Scarce			0.667				demersal	MM	1
fish	Carangiformes	Carangidae	*Carangoides bartholomaei*	car.bar	rare and Scarce			0.406				reef-associated	MS	7
fish	Carangiformes	Carangidae	*Caranx hippos*	car.hip	rare and Scarce			0.367				reef-associated	MS	1
fish	Carangiformes	Carangidae	*Cathorops spixii*	cat.spi	rare and Scarce			0.718				demersal	ES	1
fish	Clupeiformes	Engraulidae	*Cetengraulis edentulus*	cet.ede	frequent and Higher Abundant	3.75	2.67	1.08	2.472	0.403	0.63	pelagic	MM	1
fish	Moroniformes	Ephippidae	*Chaetodipterus faber*	cha.fab	rare and Scarce			0.368				reef-associated	MM	8
fish	Clupeiformes	Pristigasteridae	*Chirocentrodon bleekerianus*	chi.ble	frequent and Higher Abundant	7.37	6.23	1.14	5.465	0.281	0.89	pelagic	MS	9

fish	Carangiformes	Carangidae	*Chloroscombrus chrysurus*	chl.chr	rare and Scarce			0.55				pelagic	MS	1
fish	Pleuronectiformes	Paralichthyidae	*Citharichthys macrops*	cit.mac	rare and Scarce			0.871				demersal	MS	9
fish	Pleuronectiformes	Paralichthyidae	*Citharichthys spilopterus*	cit.spi	rare and Scarce			0.851				demersal	MM	1
fish	Perciformes	Haemulidae	*Conodon nobilis*	con.nob	frequent and Higher Abundant	1.8	1.126	0.674	1.671	0.129	0.58	demersal	MM	1
fish	Pleuronectiformes	Paralichthyidae	*Cyclopsetta chittendeni*	cyc.chi	rare and Scarce			1.219				demersal	MS	32
fish	Pleuronectiformes	Paralichthyidae	*Cynoscion virescens*	cyn.vir	frequent and Scarce			0.521				demersal	MM	10
fish	Scorpaeniformes	Dactylopteridae	*Dactylopterus volitans*	dac.vol	rare and Scarce			0.502				reef-associated	MS	10
fish	Perciformes	Gerreidae	*Diapterus auratus*	dia.aur	rare and Scarce			0.579				demersal	MM	1
fish	Perciformes	Gerreidae	*Diapterus rhombeus*	dia.rho	frequent and Scarce			0.467				demersal	MM	1
fish	Perciformes	Serranidae	*Diplectrum formosum*	dip.for	rare and Scarce			1.079				reef-associated	MS	32
fish	Syngnathiformes	Echeneidae	*Echeneis naucrates*	ech.nau	rare and Scarce			0.388				reef-associated	MM	1
fish	Pleuronectiformes	Paralichthyidae	*Etropus crossotus*	etr.cro	rare and Scarce			2.442				demersal	MM	11
fish	Perciformes	Gerreidae	*Eucinostomus argenteus*	euc.arg	rare and Scarce			0.938			0.788	reef-associated	MM	1
fish	Perciformes	Gerreidae	*Eucinostomus gula*	euc.gul	rare and Scarce	4.54	4.083	0.457	8.934		0.048	reef-associated	MM	1
invertebrate	Decapoda	Peneidae	*Farfantepenaeus subtilis*	far.sub	frequent and Higher Abundant	6.86	4.71	2.15	2.191	0.1233	0.42	demersal	MS	12
fish	Perciformes	Haemulidae	*Genyatremus luteus*	gen.lut	rare and Scarce			0.619				demersal	ES	1
fish	Perciformes	Haemulidae	*Haemulon aurolineatum*	hae.aur	rare and Scarce			0.371				reef-associated	MS	1
fish	Perciformes	Haemulidae	*Haemulon plumierii*	hae.plu	rare and Scarce			0.23				reef-associated	MS	32
fish	Perciformes	Haemulidae	*Haemulon steindachneri*	hae.ste	rare and Scarce			0.332				reef-associated	MS	13
fish	Perciformes	Haemulidae	*Haemulopsis corvinaeformis*	hae.cor	frequent and Higher Abundant	2.41	1.6265	0.7835	2.076	0.102	0.6	demersal	MS	1
fish	Clupeiformes	Clupeidae	*Harengula clupeola*	har.clu	frequent and Scarce			0.862			0.343	reef-associated	MS	1
fish	Myliobatiformes	Dasyatidae	*Hypanus guttatus*	hyp.gut	rare and Scarce			0.286				demersal	MS	1
fish	Beloniformes	Hemiramphidae	*Hyporhamphus unifasciatus*	hyp.uni	rare and Scarce			2.243				reef-associated	MM	1
fish	Acanthuriformes	Sciaenidae	*Isopisthus parvipinnis*	iso.par	frequent and Higher Abundant	2.23	1.4465	0.7835	1.846		0.6	demersal	MM	14
fish	Tetraodontiformes	Tetraodontidae	*Lagocephalus laevigatus*	lag.lae	rare and Scarce			0.393				pelagic	MM	15
fish	Carangiformes	Sciaenidae	*Larimus breviceps*	lar.bre	frequent and Higher Abundant	3.77	3.063	0.707	4.332		0.198	demersal	MM	14
fish	Ophidiiformes	Ophidiidae	*Lepophidium brevibarbe*	lep.bre	rare and Scarce			0.711				demersal	MS	-

invertebrate	Decapoda	Peneidae	*Litopenaeus schmitti*	lit.sch	frequent and Higher Abundant	4.26	2.89	1.37	2.109	0.1819	0.347	demersal	MS	16
fish	Perciformes	Lutjanidae	*Lutjanus analis*	lut.ana	rare and Scarce			0.149				reef-associated	MS	1
fish	Perciformes	Lutjanidae	*Lutjanus synagris*	lut.syn	rare and Scarce			0.219				reef-associated	MS	1
fish	Clupeiformes	Engraulidae	*Lycengraulis grossidens*	lyc.gro	frequent and Higher Abundant	2.32	1.546	0.774	1.997	0.213	0.519	pelagic	ES	17
fish	Acanthuriformes	Sciaenidae	*Macrodon ancylodon*	mac.anc	frequent and Higher Abundant			0.736			0.048	demersal	MM	14
fish	Acanthuriformes	Sciaenidae	*Menticirrhus americanus*	men.ame	frequent and Higher Abundant	2.55	2.142	0.408	5.25		0.15	demersal	MM	22
fish	Acanthuriformes	Sciaenidae	*Menticirrhus littoralis*	men.lit	rare and Scarce			0.572				demersal	MM	32
fish	Acanthuriformes	Sciaenidae	*Micropogonias furnieri*	mic.fur	frequent and Higher Abundant			0.134			0.055	demersal	MM	10
fish	Anguilliformes	Ophichthidae	*Myrichthys ocellatus*	myr.oce	rare and Scarce			0.388				reef-associated	MM	18
fish	Acanthuriformes	Sciaenidae	*Nebris microps*	neb.mic	frequent and Scarce			0.788			0.014	demersal	ES	19
fish	Clupeiformes	Pristigasteridae	*Odontognathus mucronatus*	odo.muc	frequent and Higher Abundant	4.97	4.385	0.585	7.496	0.136	0.814	pelagic	MS	20
fish	Lophiiformes	Ophichthidae	*Ogcocephalus vespertilio*	ogc.ves	rare and Scarce			0.712				reef-associated	MS	1
fish	Acanthuriformes	Sciaenidae	*Ophioscion punctatissimus*	oph.pun	frequent and Higher Abundant	1.8	0.8463	0.9537	0.887	0.362	0.591	demersal	MM	21
fish	Anguilliformes	Clupeidae	*Opisthonema oglinum*	opi.ogl	rare and Scarce			2.202			0.429	reef-associated	MM	1
fish	Pleuronectiformes	Paralichthyidae	*Paralichthys brasiliensis*	para.bra	rare and Scarce			0.406				demersal	MM	1
fish	Acanthuriformes	Sciaenidae	*Paralonchurus brasiliensis*	par.bra	frequent and Higher Abundant	2.89	1.941	0.949	2.045	0.157	0.43	demersal	MM	14
fish	Clupeiformes	Pristigasteridae	*Pellona harroweri*	pel.har	frequent and Higher Abundant	2.53	1.44	1.09	1.321	0.225	0.53	demersal	MS	32
fish	Perciformes	Pempheridae	*Pempheris schomburgkii*	pem.sch	rare and Scarce			1				reef-associated	MS	32
fish	Perciformes	Stromateidae	*Peprilus paru*	pep.par	rare and Scarce			0.718				pelagic	MS	32
fish	Perciformes	Polynemidae	*Polydactylus virginicus*	pol.vir	frequent and Higher Abundant			0.852			0.015	demersal	MM	1
fish	Scorpaeniformes	Triglidae	*Prionotus punctatus*	pri.pun	rare and Scarce			0.125				demersal	MS	1
fish	Rajiformes	Rhinobatidae	*Pseudobatos percellens*	pse.per	rare and Scarce			0.261				demersal	MS	33
fish	Clupeiformes	Clupeidae	*Rhinosardinia bahiensis*	rhi.bah	rare and Scarce			1.352				pelagic	ES	23
fish	Carcharhiniformes	Carcharhinidae	*Rhizoprionodon porosus*	rhi.por	rare and Scarce			0.132				reef-associated	MS	32
fish	Siluriformes	Ariidae	*Sciades herzbergii*	sci.her	rare and Scarce			0.417				demersal	ES	1
fish	Carangiformes	Carangidae	*Selene brownii*	sel.bro	frequent and Scarce			0.472			0.008	reef-associated	MS	1
fish	Carangiformes	Carangidae	*Selene vomer*	sel.vom	rare and Scarce			0.703				demersal	MS	1

fish	Tetraodontiformes	Tetraodontidae	*Sphoeroides greeleyi*	sph.gre	frequent and Scarce			1.091				reef-associated	ES	24
fish	Tetraodontiformes	Tetraodontidae	*Sphoeroides testudineus*	sph.tes	rare and Scarce			0.825				reef-associated	ES	1
fish	Perciformes	Sphyraenidae	*Sphyraena guachancho*	sph.gua	rare and Scarce			0.293				pelagic	MS	32
fish	Acanthuriformes	Sciaenidae	*Stellifer brasiliensis*	ste.bra	frequent and Higher Abundant	2.13	1.02	1.11	0.919	0.422	0.868	demersal	MM	1
fish	Acanthuriformes	Sciaenidae	*Stellifer microps*	ste.mic	frequent and Higher Abundant	1.94	1.335	0.605	2.207	0.174	0.6	demersal	ES	6
fish	Acanthuriformes	Sciaenidae	*Stellifer rastrifer*	ste.ras	frequent and Higher Abundant	1.8	1.045	0.755	1.384	0.234	0.256	demersal	MM	20
fish	Acanthuriformes	Sciaenidae	*Stellifer stellifer*	ste.ste	frequent and Higher Abundant	1.41	0.21	1.2	0.175	0.808	0.82	demersal	ES	26
fish	Pleuronectiformes	Cynoglossidae	*Symphurus plagusia*	sym.pla	rare and Scarce			0.783				demersal	MM	10
fish	Pleuronectiformes	Cynoglossidae	*Symphurus tessellatus*	sym.tes	frequent and Scarce			1.004			0.881	demersal	MM	10
fish	Pleuronectiformes	Achiridae	*Trichiurus lepturus*	tri.lep	frequent and Higher Abundant			0.661			0.23	demersal	MS	1
fish	Pleuronectiformes	Achiridae	*Trinectes paulistanus*	tri.pau	frequent and Higher Abundant	2.77	1.684	1.086	1.551	0.386	0.686	demersal	MM	1
fish	Acanthuriformes	Sciaenidae	*Umbrina coroides*	umb.cor	rare and Scarce			0.297				demersal	MS	32
fish	Perciformes	Mullidae	*Upeneus parvus*	upe.par	rare and Scarce			0.718				demersal	MS	32
fish	Myliobatiformes	Urotrygonidae	*Urotrygon microphthalmum*	uro.mic	rare and Scarce			0.603				demersal	MS	34
invertebrate	Decapoda	Peneidae	*Xiphopenaeus kroyeri*	xip.kro	frequent and Higher Abundant	10.4	6.8	3.6	1.889	0.2674	0.6	demersal	MS	27

1 (Vasconcelos Filho and Oliveira, 1999); 2 (Dias, 2007); 3 (Araújo et al., 2008); 4 (Denadai et al., 2004); 5 (Segura-Berttolini and Mendoza-Carranza, 2013); 6 (Barletta and Blaber, 2007); 7 (Santos, 2012); 8 (Riede, 2004); 9 (Passos et al., 2013); 10 (Paiva et al., 2009a); 11 (Oliveira and Favaro, 2011); 12 (Silva et al., 2015); 13 (Reis-Filho et al., 2010); 14 (Silva Júnior et al., 2015);15 (Denadai et al., 2012); 16 (Silva et al., 2018); 17 (Mai and Vieira, 2013); 18 (Monteiro-Neto et al., 2013); 19 (Mourão et al., 2014); 20 (Passos et al., 2013); 21 (Spach et al., 2004); 22 (Turra et al., 2012); 23 (Clark and Pessanha, 2015); 24 (Schultz et al., 2002); 26 (Rodrigues-Filho et al., 2011); 27 (Lopes et al., 2017); 28 (Lopes and Sampaio, 2002); 29 (Santos et al., 2019); 30 (Andrade et al., 2016); 31 (Brandão, 2018); 32 (Paiva et al., 2013); 33 (Favero, 2019); 34 (Almeida et al., 2000)

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DOCTORAT BRETAGNE LOIRE / SCIENCES DE LA MER ET DU LITTORAL

UBO Université de Bretagne Occidentale

Titre : L'évaluation de la pêche de la crevette en Pernambuco, au nord-est du Brésil : une approche écosystémique

Mots clés : Approche écosystémique de la pêche, Pêche à la crevette tropicale, Pêche au chalut à petite échelle, Prises accessoires, Brésil

Résumé : L'objectif principal de cette thèse est d'évaluer le contexte actuel et le potentiel futur impact de la pêche et des changements environnementaux dans le cadre de l'Approche Ecosystémique de la Pêche (AEP) sur l'écosystème côtier de Sirinhaém en tant qu'étude de cas pour le chalutage de crevettes à petite échelle dans le nord-est du Brésil, ainsi que de contribuer à la réflexion sur la mise en place d'éventuelles mesures de gestion Dans notre étude de cas et sans tenir compte des changements environnementaux, ne pas adopter de mesures de contrôle de l'effort pour les conditions actuelles de chalutage ne permet pas de causer des pertes importantes pour les espèces cibles. Une forte réduction de l'effort ou une limitation de la taille/des engins ne semblent pas être des mesures nécessaires, étant donné que, selon l'évaluation traditionnelle des stocks, les espèces cibles sont exploitées à des niveaux acceptables. Cependant, la diminution contrôlée de l'effort de chalutage jusqu'à 10% était plus favorable que la saison fermée qui ne présentait pas d'améliorations significatives en termes de fonctionnement de l'écosystème.En outre, en raison de l'extension de la zone de pêche, la gestion spatiale n'est peut-être pas très efficace dans une éventuelle gestion de la pêche. Les espèces non ciblées ne sont souvent pas prises en compte dans les mesures de gestion, étant donné leur importance socio-économique dans la région, elles doivent être mieux évaluées dans le cadre de l'AEP en tenant compte de l'effet sur l'ensemble de la dynamique trophique et de la durabilité des prises accessoires, essentielles pour la sécurité alimentaire. Les dispositifs de réduction des prises accessoires peuvent être une alternative, mais leur viabilité doit être mieux évaluée, principalement en termes socio-économiques. Indépendamment des mesures qui peuvent être appliquées, nous avons identifié que l'effet cumulatif des changements environnementaux et de la pêche, menace la durabilité de l'écosystème. Il faut donc en tenir compte dans toute mesure éventuelle.

Title : Evaluation of the shrimp fishery in the Pernambuco, Northeast of Brazil: An ecosystem approach

Keywords : Ecosystem Approach to Fisheries, Tropical shrimp fisheries, Small-scale trawl fisheries, Bycatch, Brazil

Abstract : The overall aim of this thesis is to assess the current framework and potential future impact of fishing and environmental changes under the scope of Ecosystem Approach to Fishery (EAF) on the Sirinhaém coastal as a case study for small-scale shrimp trawling in Northeastern Brazil, contributing to the reflection on the implementation of possible management measures. In our case study and without accounting the environmental changes, not adopting effort control measures for the current trawling conditions do not appear to cause major losses for target species. A high effort reductions or size/gear limitations did not appear to be necessary measures, considering that, according to the traditional stock assessment, the target species are being exploited at accepted levels. However, the controlled decrease trawling efforts up to 10% were promising than the closed season which did not present significant improvements in terms of ecosystem functioning. In addition, given the fishing area extension, spatial management maybe not very effective in a possible fisheries management. The non-target species often not considered in management measures, given the socio-economic importance in the region, they need to be better assessed under the EAF taking into the effect in whole trophic dynamic and the bycatch sustainability, essential for the food security. Bycatch Reduction Devices may be one alternative, but its viability needs better evaluate, mainly in terms of socio-economic. Regardless the measures that may be applied, we identified that the cumulative effect of environmental changes and fishing, threaten the sustainability of the ecosystem. Hence, should be considered in any eventual measures.