UNIVERSIDADE FEDERAL RURAL DE PERNAMBUCO
PRÓ-REITORIA DE PESQUISA E PÓS-GRADUAÇÃO
PROGRAMA DE PÓS-GRADUAÇÃO EM RECURSOS PESQUEIROS E AQUICULTURA

Interpretando os efeitos antrópico-ambientais na ictiofauna ao longo dos habitats costeiros e marinhos no Nordeste do Brasil

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Tese apresentada ao Programa de Pós-Graduação em Recursos Pesqueiros e Aquicultura da Universidade Federal Rural de Pernambuco como exigência para obtenção do título de Doutor.

Prof. Dr. Thierry Frédou

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Dedicatória

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Resumo

As alterações nos ecossistemas costeiros através da pesca ou outros impactos antrópicos, representam uma grande ameaça à conservação de muitas espécies que dependem da complementaridade e do funcionamento adequado dos diferentes ecossistemas costeiros (mangue, recifes...). Este estudo teve como objetivo documentar alguns impactos sofridos pela sucessão de habitats costeiros interconectados em Pernambuco. Deste modo, essa tese foi dividida em duas partes. A primeira parte abordou as atividades que tiveram impacto físico sobre a paisagem marinha conectada. Para este fim, foram coletados dados via satélite da paisagem costeira durante as últimas três décadas a fim de fornecer informações sobre a evolução da cobertura dos manguezais em três estuários de Pernambuco. Além disso, análises tróficas usando isótopos estáveis foram empregados para entender a variabilidade na conectividade ecológica entre habitats estuarinos e costeiros num contexto de mudanças antropogênicas crescentes nas morfologias estuarinas. Investigamos a influência desta morfologia nos fluxos entre estuários morfologicamente distintos e suas áreas costeiras adjacentes traçando a origem da matéria orgânica (costeira ou estuarina) subjacente à dieta da carapeba *Eucinostomus argenteus*. Os resultados desta primeira parte mostram que, embora as atividades antropogênicas (aquicultura, agricultura, urbanização e construção de portos) tenham causado desmatamento de mangues, há uma tendência de aumento de sua cobertura total em dois dos três estuários estudados nas últimas três décadas. Isso foi relacionado a salinização, permitida pelas mudanças climáticas e outras atividades antrópicas (porto, represas). Vimos que a morfologia dos estuários podia influenciar a dieta da carapeba fazendo com que a contribuição dos recursos costeiros na dieta aumente com a conexão com o mar (determinada por características morfológicas como a largura da boca). Assim, as atividades estabelecidas ao longo da costa de Pernambuco estão modificando a paisagem marítima ao remover habitats como os manguezais e estão afetando o funcionamento dos ecossistemas ao modificar as variáveis ambientais, influenciando os fluxos entre habitats conectados e induzindo a salinização. A segunda parte da tese foi relacionada ao impacto das frotas coexistentes que exploram esses ambientes conectados. Com esta finalidade, foram estudadas as capturas das principais artes de pesca operadas ao longo do gradiente estuário-plataforma continental. Nosso estudo mostrou que o complexo padrão de pesca das diferentes frotas coexistentes resulta na captura de espécies migratórias por várias artes de pesca ao longo do gradiente estuarino-plataforma e em distintos estágios da vida. As diversas artes de pesca capturaram majoritariamente compartimentos produtivos (indivíduos jovens e espécies de baixo nível trófico). Como as frotas coexistentes têm como alvo espécies comuns, os rendimentos destas pescarias estão inter-relacionados, o que torna necessária uma gestão multiecossistêmica. Nossos resultados sugerem que limitações de tamanho não seriam aplicáveis a estas frotas coexistentes, pois os benefícios de rendimento não serão obtidos pelas frotas que sofreriam essas limitações. Uma abordagem baseada em ecossistemas como o "*Balanced harvest*" (ideia de pescar uma variedade de espécies, estoques e tamanhos, em proporção à sua produtividade natural), consistente com os padrões de pesca existentes, poderia ser uma alternativa interessante para o manejo dessas frotas. Como a atividade pesqueira ocorre em habitats degradados, um planejamento espacial marinho seria então de extrema importância para a gestão dessas pescarias. Este planejamento deve integrar todas as atividades que possam causar danos aos habitats costeiros e marinhos pois o rendimento das pescarias requer o bom funcionamento destes ecossistemas conectados.

Palavras chaves: ecossistemas costeiros, impactos antrópicos, gestão pesqueira, interconectividade, planejamento espacial marinho.

Abstract

Changes in coastal ecosystems through fishing or other anthropic impacts represent a major threat to the conservation of many species that depend on the complementarity and proper functioning of different coastal ecosystems (mangroves, reefs, marine prairies). This study aimed to document some of the impacts suffered by the succession of interconnected coastal habitats in Pernambuco. Thus, this thesis was divided into two parts. The first part addressed activities that had a physical impact on the connected marine landscape. Hence, satellite data from the coastal landscape has been collected over the past three decades to provide information on the evolution of mangrove cover in three estuaries of Pernambuco. In addition, stable isotope methods have been employed to understand the variability in ecological connectivity between estuarine and coastal habitats in the context of increasing anthropogenic changes in estuarine morphologies. We investigated the influence of this morphology on the flows between morphologically distinct estuaries and their adjacent coastal areas by tracing the origin of organic matter (coastal or estuarine) underlying the diet of the silver mojarra *Eucinostomus argenteus*. The results of this first section show that although anthropic activities (aquaculture, agriculture, urbanization, and port construction) have caused mangrove swamps to be deforested, there is a tendency to increase their total coverage in two of the three estuaries studied over the past three decades. This has been related to salinization, allowed by climate change and other anthropic activities (port, dams). We observed that the morphology of the estuaries could influence the diet of the silver mojarra, causing the contribution of the coastal resources to the diet to increase with the connection with the sea (determined by morphological characteristics such as mouth width). Thus, activities established along the coast of Pernambuco are modifying the maritime landscape by removing habitats such as mangroves and are affecting the functioning of ecosystems by modifying environmental variables, influencing flows between connected habitats and inducing salinization. The second part of the thesis was related to the impact of coexisting fleets that exploit these connected environments. For this purpose, the catches of the main fishing gears operated along the estuary-shelf gradient were examined. Our study showed that the complex fishing pattern of the different coexisting fleets results in the capture of migratory species by various fishing gears along the estuarine-platform gradient and at different life stages. The various fishing gears have mostly caught productive compartments (young individuals and low trophic species). As the co-existing fleets target common species, the incomes of these fisheries are co-dependent, making multi-ecosystem management necessary. Our results suggest that size limitations would not be enforceable on these co-existing fleets, because the income benefits will not be obtained by the fleets that would suffer from these limitations. An ecosystem-based approach such as "Balanced harvest" (the idea of fishing a variety of species, stocks and sizes in proportion to their natural productivity), consistent with existing fishing patterns, could be an interesting alternative for managing these fleets. As fishing activity occurs in degraded habitats, marine spatial planning would then be of paramount importance for the management of these fisheries. This planning should integrate all activities that may cause damage to coastal and marine habitats because fisheries yields require the proper functioning of these connected ecosystems.

Key words: coastal ecosystems, human impacts, fisheries management, inter-connectivity, marine spatial planning.

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Sumário

1. Introdução

1.1. Contextualização da pesquisa

Os ambientes costeiros fornecem uma variedade de serviços ecossistêmicos, muitos dos quais são de importância fundamental para o bem-estar humano, para a saúde, a subsistência e a sobrevivência (BARBIER, 2015; BARBIER et al., 2011; SUTTON-GRIER; SANDIFER, 2019). Porém, a perda e degradação de habitats estuarinos, costeiros e marinhos, vem reduzindo a biodiversidade, ameaçando a multiplicidade de bens e serviços fornecidos pelos ecossistemas marinhos e diminuindo a resiliência do sistema a pressões futuras (DUARTE et al., 2015; PENDLETON et al., 2012). Esta modificação no habitat é causada por uma ampla gama de atividades e pressões antrópicas, incluindo práticas de pesca destrutivas (por exemplo, arrasto de fundo), sobrepesca, aquicultura, disseminação de espécies invasoras, eutrofização, exploração de petróleo, poluição e gás em larga escala (CRAIN et al., 2009; LU et al., 2018; TIANO et al., 2019). Este contexto de mudanças globais levanta questões sobre como os seres humanos irão lidar com a escassez dos recursos dos quais a sociedade depende para seu bom funcionamento (MOORE, 2016).

Consequentemente, dedicou-se especial atenção à crise pesqueira, que consiste no esgotamento gradual dos recursos marinhos, seja devido à poluição, à sobrepesca ou à degradação do ecossistema (PAULY, 2019). As alterações nos ecossistemas através da pesca ou outros processos ligados a antropização, representam uma grande ameaça à conservação de muitas espécies marinhas particularmente aquelas que estão em perigo (WILSON et al. 2008; WILSON et al. 2010). Acredita-se que a exploração tenha causado 55% das extinções marinhas, enquanto a degradação do habitat explica mais 37%, enfatizando a importância de considerar estes dois processos tanto para a conservação como para a sustentabilidade (DULVY; SADOVY; REYNOLDS 2003; DULVY 2006).

Como os ecossistemas não funcionam de forma independente, muitas vezes há interações importantes e fundamentais entre os ambientes terrestres, marinhos e de água doce (BEGER et al., 2010). Deste modo, as atividades costeiras terrestres podem modificar ou destruir os habitats costeiros e marinhos, causando o escoamento de sedimentos, nutrientes, toxinas e poluentes, e até mesmo alterar o fluxo de correntes e marés (DU; YANG; FENG, 2016; LECHÊNE et al., 2018; WETZEL et al., 2012). Assim, o surgimento da ecologia da paisagem marítima, "seascape ecology", acarretou na necessidade de considerar os habitats costeiros como um único continuum, e não como biomas separados (BERKSTRÖM; JÖRGENSEN; HELLSTRÖM, 2013; GREEN; SMITH; UNDERWOOD, 2012). Esta mudança conceitual permite uma gestão mais integrada da zona costeira (BEGER et al., 2010). Em particular, abordar a conectividade entre habitats é essencial para a gestão pesqueira porque os habitats costeiros fornecem funções ecológicas complementares que são críticas para os recursos marinhos vivos (SHEAVES, 2009).

Tal abordagem é essencial para proporcionar uma melhor compreensão da pesca tropical de pequena escala que é caracterizada por sua complexidade multidimensional (multi-espécies, multi-arte de pesca, multi-frotas, multi-ecossistemas). Assim, as interações técnicas (onde vários peixes são capturados juntos na mesma rede ou operação de pesca) e as interações multiespecíficas (onde a abundância da população influencia uns aos outros através da predação e competição) dificultam a administração adequada dessas pescarias mistas (DOLDER; THORSON; MINTO, 2018; THORPE; JENNINGS; DOLDER, 2017). Como a produtividade depende das espécies, a resiliência à pesca dependerá das estratégias de reprodução, crescimento e outras características específicas. Por exemplo, peixes de grande porte, longa vida, maturidade tardia e baixa taxa de reprodução são mais vulneráveis à pesca (CHEUNG; PITCHER; PAULY, 2005; ELSLER et al., 2019; LUCENA-FRÉDOU et al., 2017). A compreensão das estratégias de cada espécie é necessária para uma gestão adequada dessas pescarias multiespecíficas (FLETCHER et al., 2016). Além disso, a integração da variabilidade espacial é crítica para seu manejo (ADAMS et al., 2018; REES; KNOTT; DAVIS, 2018) particularmente porque essas pescarias exploram vários habitats com características diferentes. Uma abordagem que envolva esses dois aspectos é essencial para descobrir os limites apropriados para implementar uma abordagem ecossistêmica (RINDORF et al., 2017) especialmente para a pesca tropical multi-ecossistema que envolve a pesca de espécies migradoras. A distribuição destas espécies está relacionada com sua ontogenia e, portanto, sua extensão espacial depende de parâmetros individuais de crescimento, bem como de fatores ambientais (LEA et al., 2018).

De fato, as espécies migradoras representam uma ligação entre os habitats costeiros, pois ao longo de seu ciclo de vida, elas precisam usar várias partes do continuum costeiro (BERKSTRÖM et al., 2013; OLDS et al., 2016). Assim, estas espécies dependem da complementaridade e do funcionamento adequado dos diferentes ecossistemas costeiros (mangue, coral, algas marinhas) para atender às suas diversas necessidades (BERKSTRÖM et al., 2013; OLDS et al., 2016, 2018). Assim, os jovens de muitas espécies migratórias marinhas dependem de áreas de manguezais que oferecem condições adequadas para seu crescimento (ARCHAMBAULT et al., 2018). Estas espécies utilizam os manguezais como abrigo porque sua estrutura complexa pode protegê-los contra predadores (NANJO et al., 2014). Portanto, qualquer perda significativa de manguezais terá consequências significativas na biodiversidade e abundância de peixes (ESTOQUE et al., 2018; NIP; WONG, 2010). Em particular, este habitat melhora o rendimento da pesca costeira ao servir de berçário para espécies comerciais que migram dos manguezais para o mar aberto (HUTCHISON; SPALDING; ZU ERMGASSEN, 2014; SEARY et al., 2020). Assim, os manguezais são um componente chave na sucessão de habitats costeiros e marinhos interligados, dos quais depende o rendimento da pesca tropical e, portanto, a subsistência essencial das comunidades pesqueiras (CARRASQUILLA-HENAO; JUANES, 2017; MANSON et al., 2005; REIS-FILHO; HARVEY; GIARRIZZO, 2019).

Apesar de sua importância socioeconômica, o funcionamento dessas pescarias multidimensionais é pouco estudado (CHUENPAGDEE et al., 2019). Sua marginalização como pesca de baixo rendimento, baixa tecnologia e uso de arte de pesca pouco seletivos, em desacordo com as políticas de gestão de longo prazo, contribuem para a sua desvalorização no mundo industrial (KOLDING e VAN ZWIETEN 2011; KOLDING, JACOBSEN, et al. 2016). Como a pesca em pequena escala nos países em desenvolvimento é destinada principalmente a um mercado onde as preferências de tamanho não são essenciais (KOLDING et al., 2016), uma abordagem de gestão diferente daquela aplicada para pescarias industriais poderia ser implementada. Uma das propostas sugeridas é a colheita balanceada (“Balanced harvest”), ou seja, "pescar a maior variedade possível de espécies, estoques e tamanhos em um ecossistema, em proporção à sua produtividade natural, de modo que o tamanho relativo e a composição das espécies sejam mantidos" (GARCIA et al., 2012) mas essa abordagem nunca foi aplicada.

No Brasil, essas pescarias de pequena escala são responsáveis por mais de 90% do emprego no setor da pesca e cerca de 600.000 pescadores estavam diretamente envolvidos em atividades de pesca em 2011 (BARANGE et al., 2018). Estas pescarias brasileiras são inadequadamente administradas de modo que vários estoques são super explorados e não são sistematicamente avaliados (GASALLA; ABDALLAH; LEMOS, 2017). No entanto, eles são essenciais para a segurança alimentar, uma vez que o consumo per capita de frutos do mar aumentou de 6,66 para 9,75 kg entre 2005 e 2010 no Brasil (BARONE et al., 2017). Assim, estas pescarias são uma fonte crucial de proteína mais particularmente no Nordeste, visto que 47% dos pescadores artesanais do país estão registrados nesta região (GOMES DE ALENCAR et al., 2019).

A pesca na região nordeste é formada por múltiplas frotas, que têm diferentes métodos, poder e áreas de pesca (FRÉDOU; FERREIRA; LETOURNEUR, 2009; HAZIN et al., 1998). Contudo, por serem pescarias multiespecíficas que exploram ambientes conectados, a pressão da pesca de uma frota pode ter repercussões no rendimento de outra. De fato, essas frotas coexistentes provavelmente exploram o mesmo recurso devido ao grande número de espécies marinhas que migram entre os diferentes habitats costeiros e marinhos. Além disso, estas pescarias dependem de habitats tropicais produtivos que sofreram degradação, especialmente devido ao desenvolvimento urbano, aquicultura e atividades portuárias (GUIMARÃES et al., 2010; MUNIZ et al., 2005). Ademais, o nordeste do Brasil é a segunda região com o maior percentual de população (compreendendo quase 30% da população nacional) (IBGE, 2020). Isto implica uma demanda importante para a oferta de alimentos, principalmente do setor pesqueiro, já que 60% dos consumidores regulares de pescado brasileiro vivem na região Norte-Nordeste. (SONODA et al. 2012). O Estado de Pernambuco é um dos estados mais populosos da região nordeste, com 9,6 milhões de habitantes (IBGE, 2020). Existem 34 comunidades ribeirinhas em Pernambuco que se sustentam da pesca e da coleta de moluscos e crustáceos (LESSA et al., 2009). Cerca de 13.000 pescadores estão envolvidos na pesca artesanal de pequena escala (LESSA et al., 2009).

Há, portanto, a necessidade de um estudo que investigue o impacto nesses ambientes, decorrente da pesca ou de outras pressões antrópicas. A costa de Pernambuco compreende um mosaico de ambientes costeiros que constituem uma complexa paisagem marítima ao longo do gradiente estuário-plataforma continental. De fato, o conhecimento sobre a conectividade entre os ambientes marinho e de transição é primordial para um manejo sustentável dos recursos desses biomas. Desta forma, esse estudo pretende investigar como diferentes pressões antropogênicas interagem com as diversas características dos habitats conectados. Este estudo está dividido em duas partes complementares. A primeira parte focará nos impactos antropogênicos sobre a paisagem marinha regional, com interesse nas atividades que impactam fisicamente a paisagem marinha conectada. Com esta finalidade, um primeiro artigo científico trata da influência das atividades antropogênicas na evolução espaço-temporal da cobertura dos manguezais nos últimos 30 anos. Um segundo artigo examina como as características morfológicas influenciam fortemente o fluxo de matéria orgânica entre os habitats, o que é uma questão particularmente relevante dadas as crescentes alterações antropogênicas nos estuários. Em seguida, uma segunda parte trata da gestão da pesca nesses ambientes conectados. Assim, um terceiro artigo caracteriza o impacto de um importante tipo de pesca que ocorre nos manguezais. E finalmente o quarto artigo discute a aplicabilidade de uma abordagem ecossistêmica para as pescarias mistas de Pernambuco que operam em vários habitats marinhos conectados. Por fim, são apresentadas as considerações finais do estudo, contendo uma síntese das informações abordadas nestas duas componentes para realçar a necessidade de uma gestão da pesca mais holística que integre a conectividade dos habitats marinhos que vem sendo degradados.

1.2. Objetivos do trabalho

Este estudo teve como objetivo avaliar alguns dos impactos sofridos pela sucessão de habitats costeiros em Pernambuco e sua relação com a exploração dos recursos marinhos e a atividade pesqueira na região. Para isso, essa tese foi dividida em duas partes. A primeira com um interesse nas atividades que impactaram fisicamente a paisagem marinha conectada. E a segunda analisa o impacto de frotas coexistentes que exploram esses ambientes conectados. Nossa hipótese é que cada um dos impactos tem implicações diferentes para o funcionamento do conjunto de habitats interconectados.

Parte 1 - Impactos da degradação do habitat ligada as atividades antrópicas estabelecidas nos ambientes costeiros.

O primeiro artigo intitulado "Coastal land use in Northeast Brazil: mangrove coverage evolution over three decades" fornece informações sobre a evolução da cobertura dos manguezais em três dos principais estuários de Pernambuco, durante um período de três décadas. Estes três sistemas têm sofrido com diferentes formas de uso do solo. Nossa principal hipótese é a de que cada tipo de alteração da paisagem (expansão portuária, aquicultura, urbanização e agricultura) ameaça os manguezais e tem um impacto estrutural quantificável. Este estudo foi o primeiro a aplicar a técnica de "grid" para investigar mudanças na cobertura dos manguezais em escala local. Nosso objetivo era de identificar e quantificar de maneira precisa as modificações espaço-temporal no manguezal e relacioná-las com as atividades antrópicas sofridas localmente.

O objetivo do segundo artigo intitulado "Importance of estuary morphology for ecological connectivity with their adjacent coast: a case study in Brazilian tropical estuaries" foi compreender a variabilidade na conectividade ecológica entre os habitats estuarinos e costeiros. Este artigo está inserido em um contexto onde há mudanças antropogênicas crescentes nos estuários, notadamente em suas morfologias. Investigamos os fluxos entre três estuários morfologicamente distintos e suas áreas costeiras adjacentes, rastreando a origem da matéria orgânica (costeira ou estuarina) subjacente à dieta de *Eucinostomus argenteus*. Nossa hipótese foi que as disparidades geomorfológicas entre os estuários, proporcionando-lhes diferentes graus de conexão com o mar, influenciam a origem da matéria orgânica que sustenta a dieta de *E. argenteus.*

1.2.1. Parte 2: Impactos das frotas coexistentes que exploram esses ambientes conectados.

No terceiro artigo "impact of intertidal fishing in mangroves", investigamos as particularidades da pesca entremarés nos manguezais e suas consequências em termos de gestão. Com esta finalidade, descrevemos a variabilidade espacial e temporal das amostras em termos de composição da família de peixes. Também estudamos o impacto da pesca de camboa, em três espécies com diferentes parâmetros de crescimento, a fim de estudar a variabilidade espaço-temporal das capturas em termos de número de indivíduos por amostra, tamanhos e ontogenia, e fornecer informações sobre a seletividade desta arte para cada espécie.

No quarto artigo intitulado "Co-existing fleets harvesting migrant species across several heavily exploited tropical habitats: a complex pattern compatible with Balanced harvest", investigamos como uma pescaria multiespecífica no nordeste do Brasil pode afetar o complexo ciclo de vida das espécies migratórias ao longo de um gradiente estuarino. O objetivo era de focar em espécies modelo que realizam migrações ontogenéticas através do gradiente estuário-plataforma continental e são relevantes para a pesca em todas as suas fases. Portanto, estas espécies habitam todos os ambientes explorados pelas pescarias de pequena escala na região, de modo que seu manejo requer uma visão multi-ecossistêmica. A seletividade complementar das artes de pesca deveria permitir uma ampla gama de tamanhos e espécies a serem coletadas, proporcionando um padrão de pesca compatível com uma abordagem *"Balanced Harvest"*. Assim, este trabalho (i) descreve as capturas relativas por tipos de artes de pesca, espécies e sua ontogenia, (ii) caracteriza o padrão de pesca de pequena escala sobre espécies migratórias e (iii) discute a aplicabilidade do *balanced harvest* para o manejo de espécies migratórias em pescarias tropicais.

Parte 1 - Impactos da degradação do habitat ligada as atividades antrópicas estabelecidas nos ambientes costeiros.

Coastal land use in Northeast Brazil: mangrove coverage evolution over three decades.

Artigo científico publicado na revista Tropical Conservation Science (ISSN: 1940-0829).

2. Coastal land use in Northeast Brazil: mangrove coverage evolution over three decades.

2.1. Abstract

It is essential to monitor both the mangrove coverage and the encroachment of anthropogenic activities to assess the evolution of these highly valuable and threatened ecosystems. Using the grid technique, Landsat images of three estuaries in Pernambuco, northeast Brazil, were analyzed to track changes in land use over the past three decades. This study is the first time the grid technique has been used as a precise method to quantify and localize the mangrove coverage changes at local scale. We found that the technique was a synthetical and cost-effective way of observing land-use changes over the study period, and its precision was evident for local-scale study. An increasing trend of mangrove coverage was observed and can be related to the salinization. This increase in salinization is a result of anthropogenic activities and climate change. We also found that differences in geometrical properties influenced the spatiotemporal patterns of mangroves. To mitigate adequately the negative impacts induced by anthropization and climate change, the characteristics of mangrove forest configuration and human activities should therefore both be considered.

Keywords : mangrove, Landsat, grid technique, anthropic threats, land use, remote sensing.

2.2. Introduction

Mangroves are of great socioeconomic and environmental importance globally. They provide raw materials and food, and contribute greatly to coastal protection, erosion control, water purification, carbon sequestration, tourism, recreation, and education (Barbier et al., 2011). They also participate to the maintenance of fisheries by providing nursery habitats and increasing marine productivity. This makes them of central importance to local culture and traditional knowledge (Walters et al., 2008). These forests of salt-tolerant species allow the development of complex food webs

(Valiela et al., 2001), and because mangrove ecosystems carry out essential ecological functions, any significant loss of mangrove forests will have significant consequences on these food webs as well as on biodiversity and abundance.

Yet, mangrove wetlands are being threatened by the growth of anthropogenic activities. Moreover, climate change, the increase of CO_2 emissions, rising sea levels and alterations in precipitation regimes are having a great impact on these ecosystems (McKee et al., 2012; Ward and Larcombe, 2010). Indeed, more than one third of the world mangrove disappeared in the last 60 years and since the beginning of the 21st century, mangroves continue to be lost at a rate of about 0.2% of their total area per annum (Hamilton and Casey, 2016). Despite these threats, only 6.9% of mangroves worldwide come under any form of protected area legislation (Giri et al., 2011). Mangrove forest management is thus a great challenge for biodiversity preservation, particularly in the estuaries where human activities are encroaching (Alongi, 2002; De Souza-Machado et al., 2016; Kennish, 2002).

Mangrove vegetation grows in a relatively narrow fringe between the land and the sea, between latitudes 25°N and 30°S (Valiela et al., 2001).The fact that mangroves are mostly found in developing regions of Asia, Africa and South America adds additional challenges to their monitoring (Li, Mao, Shen, Liu, & Wei, 2013; R. O. Santos, Lirman & Serafy, 2011). There is also a lack of precision over the definition of mangrove forests, and this contributes to a lack of clarity regarding the state of mangrove forests globally. For example, mangroves are often defined as intertidal communities of trees, but this can sometimes also refer to muddy sediments and other herbaceous vegetation. Therefore, it is impossible to establish whether the worldwide extent of mangrove ecosystems is 100 000 km^2 or 200 000 km^2 (Wilkie and Fortune, 2003).

The Brazilian coast has the second largest mangrove area in the world, distributed between the Oiapoque River on the French Guyana border (04°30’N), and Sonho Beach in the state of Santa Catarina (28°53’S) (Kjerfve and Lacerda, 1993). There was approximatively 10 124 km^2 of mangrove in Brazil in 1983, but by 2000 the estimated area had shrunk to 9 630 km^2 (Giri et al., 2011; Herz, 1991). In the last three decades, the Brazilian mangrove has been severely impacted by harbor expansions, the development of tourism infrastructure, and the growth of agriculture and industry, especially along the northeastern and southeastern coasts (Schaeffer-Novelli et al., 2000). The destruction of mangrove areas for the installation of shrimp farms has been identified as a particularly important factor for mangrove loss in the northeast region of Brazil (Guimarães et al., 2010).

The Brazilian state of Pernambuco has significant mangrove coverage because its geographical position and low altitude is conducive to mangrove growth (Silva, Galvíncio, Brandão Neto, & Morais, 2015). Along the 187 km of coast, there are fifteen estuaries with mangroves. This habitat is important for fauna biodiversity (e.g. fish, birds, mollusks and crustaceans) in the area. Indeed, more advanced larval stages were observed in the mangroves in Pernambuco compared to adjacent habitats, indicating strategic ontogenic migration to this zone (Silva-Falcão et al., 2013).

Biodiversity and fisheries are closely related, and many anthropogenic activities depend on a healthy mangrove ecosystem. There are 34 riverine communities in Pernambuco subsisting on fishing and gathering mollusks and crustaceans (Lessa et al., 2009b). Some 13 000 fishermen are involved in small-scale artisanal fishing (Silva et al., 2015). Collecting data describing the condition of mangrove coverage in the area should, therefore, be considered a top research priority, since an entire socio-economic system depends on it.

To understand mangrove transformation trends, historical data on land use is required. Integrating Geographic Information Systems (GIS) and remote sensing enables the generation of data that can accurately record landcover changes (Rawat and Kumar, 2015). Maps generated through this approach illustrate the ecological properties of the studied areas and can thus be used as tools to guide environmental managers and policy makers (Schmidt et al., 2004). Maps of anthropic activities must be used to implement concrete conservation strategies (Tulloch et al., 2015). Vulnerable areas need to be identified to enable efficient planning and prioritisation of conservation measures (Groves et al., 2002). Several vegetation indices have shown the benefits of separating the detected vegetation into small grids that allow meaningful spatial and temporal comparisons of vegetation dynamics and enable research to be conducted with the precision necessary to characterize local variations (Huete et al., 2002).

The aim of this study was to provide information on the evolution of mangrove cover in three of the main estuaries of Pernambuco, over a period of three decades. These three systems have suffered from different forms of land exploitation and we expected them to show different trends in their vegetation patterns. Our main hypothesis was that each type of landscape alteration (harbor expansion, aquaculture, urbanization and agriculture) threatens the mangroves and has a quantifiable structural impact. The grid technique has never been applied to investigate mangrove coverage changes at a local scale and this study is the first to test if this application is effective. It is assumed that dividing mangroves into small cells will aid with the precise identification and quantification of drastic spatiotemporal modifications.

2.3. Methods

2.3.1. Study area

Three estuaries – the Santa Cruz channel, Sirinhaém and Suape – were chosen because of the differences in their geomorphology and in the human activities being conducted within their areas (Figure 1).

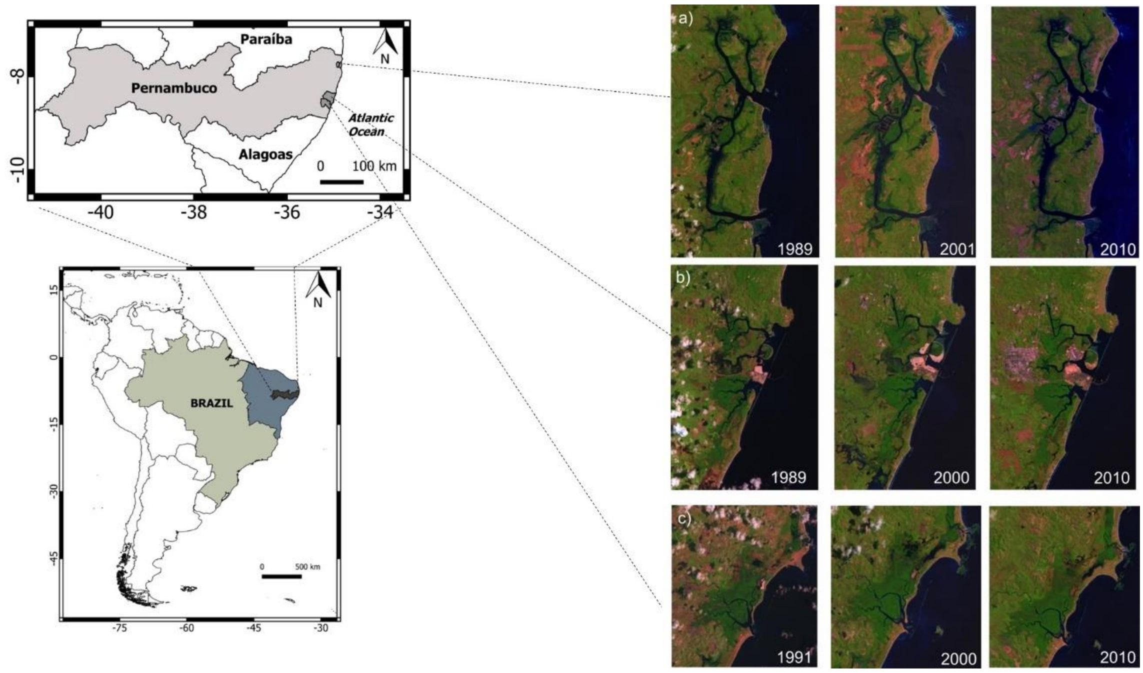


Figure: 1 Coastal land use in Northeast Brazil: mangrove coverage evolution over three decades

The Santa Cruz Channel is a 20 km long coastal estuarine complex located in the north of Pernambuco and comprising a U-shaped area of 22 km^2. The Santa Cruz Channel circles Itamaracá Island and has two accesses to the Atlantic Ocean (Schwamborn et al., 2001). The mangrove composed by *Rhizophora mangle* (red mangle), *Avicennia shaueriana* (black mangle) and *Laguncularia racemosa* (white mangle) (Medeiros et al., 2001) is distributed all along the channel, particularly on its eastern borders (Gomes et al., 2002). Situated at the center of the channel, the city of Itapissuma is Pernambuco's main fishing hub and one of the regions best-known tourism destinations (Quinamo, 2006). Aquaculture, artisanal fishing, industry and agriculture are all important anthropogenic activities in the area (De Moura & Candeias, 2009) .

The second estuary, in the Suape estuarine complex, has a watershed of 3,800 km^2 and is located approximately 40 km south of the city of Recife, between 8°15'–8°30'S and 34°55'–35°05'W (Koening et al., 2003; Souza and Sampaio, 2001). The region has a flattened geomorphological shape, medially divided by the outcrops of the fractured volcanic masses of the cape of Santo Agostinho (M. A. Santos & Costa, 1974; Silva et al., 2015). Before the construction of a port in 1979, four rivers–the Massangana, Tatuoca, Ipojuca and Merepe–drained into the Suape estuary, itself partly isolated from the ocean by an extensive sandstone reef-line (Silva et al., 2004). The construction of the port altered the geomorphological and hydrodynamic conditions of these rivers (Muniz et al., 2005). Indeed, landfills blocked the flow of both the Ipojuca and Merepe rivers, causing the retention and accumulation of water, which led to the inundation of the mangroves. The mangroves of the Suape complex are characterized by four genera belonging to three families: *R. mangle, Laguncularia racemosa (L.) c.f. Gaertn., A. germinans, Avicennia schaueriana Stapf & Leechman ex Moldenke, and Conocarpus erectus L*, as well as a variety of peripheral and generalist species (Silva et al., 2015).

The mangrove forest of the third study area, situated in Barra de Sirinhaém district in the southern coast of Pernambuco, is characterized by *R. mangle*, *L. racemosa*, and by species from the genera *Avicennia* (Maciel and Giuseppe Chaves Alves, 2009). In addition to artisanal fishing, agroindustry, services and trade are important socio-economic activities in the zone. A sugarcane plantation was established there in the 19th century and Sirinhaém plays an important role in national sugar production.

2.3.2. GIS analysis

In order to obtain the grid cells for our analysis (Figure 2) the following methods were used:

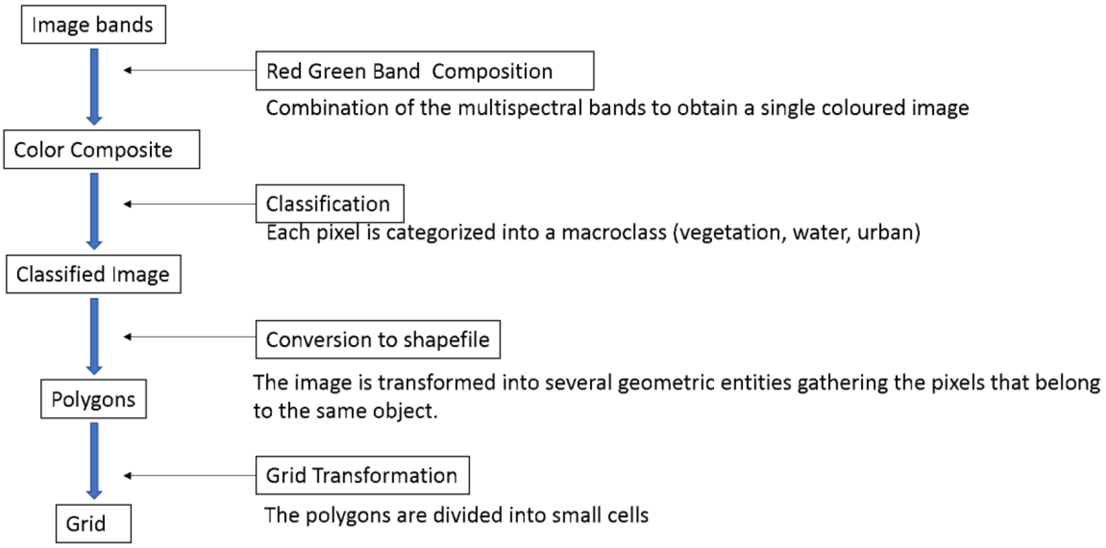
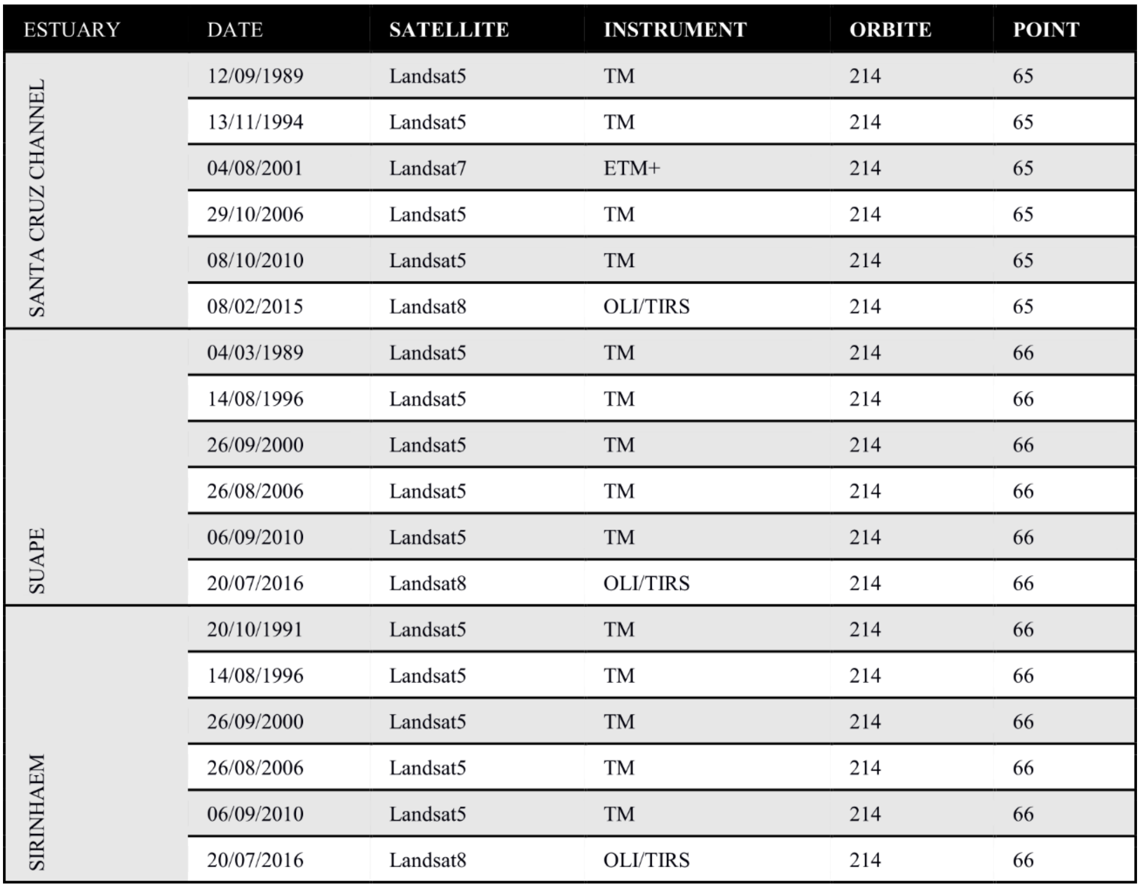


Figure 2: Flowchart of the GIS analysis

Satellite images

Satellite images of the three estuaries over the last three decades were analyzed to describe the evolution of the mangrove area. Optical images between 1989 and 2016 from the following satellites and sensors were used: LANDSAT-5/TM; LANDSAT-7/ETM+, LANDSAT-8/OLI/TIRS (Table 1).

Table 2-1: Characteristics of the Collected Images.



ESTUARY	DATE	SATELLITE	INSTRUMENT	ORBITE	POINT
SANTA CRUZ CHANNEL	12/09/1989	Landsat5	TM	214	65
	13/11/1994	Landsat5	TM	214	65
	04/08/2001	Landsat7	ETM+	214	65
	29/10/2006	Landsat5	TM	214	65
	08/10/2010	Landsat5	TM	214	65
	08/02/2015	Landsat8	OLI/TIRS	214	65
SUAPE	04/03/1989	Landsat5	TM	214	66
	14/08/1996	Landsat5	TM	214	66
	26/09/2000	Landsat5	TM	214	66
	26/08/2006	Landsat5	TM	214	66
	06/09/2010	Landsat5	TM	214	66
	20/07/2016	Landsat8	OLI/TIRS	214	66
SIRINHAEM	20/10/1991	Landsat5	TM	214	66
	14/08/1996	Landsat5	TM	214	66
	26/09/2000	Landsat5	TM	214	66
	26/08/2006	Landsat5	TM	214	66
	06/09/2010	Landsat5	TM	214	66
	20/07/2016	Landsat8	OLI/TIRS	214	66

These images were acquired from the Landsat collection of the United States Geological Survey site (https://www.usgs.gov/). Different compositions of Red (R), Green (G), and Blue (B) bands were realized to create false color images: Landsat-5 (bands 547), Landsat-7 (bands 543) and Landsat-8 (bands 654) (Figure 1).

Image processing

To map different anthropogenic activities and identify features in the landscape, a semi-automatic (or supervised) image processing classification technique was applied to the satellite images which enabled the identification of landcover through its spectral signature (Congedo, 2016). Macro-classes of objects with similar spectral signatures were determined in order to create manual training spectral signatures to overlay pixels belonging to the same landcover class. Two algorithms were then applied to assign each pixel to a class: the minimum distance and the maximum likelihood. The minimum distance algorithm calculates the Euclidean distance between spectral signatures of image pixels and training spectral signatures and assigns, for each pixel, the closest class of spectral signature (Mather and Tso, 2016).

The maximum likelihood algorithm was used to categorize a pixel using the probability distributions for the landcover classes, following Bayes' theorem (Munyati, 2004).

The supervised classification was carried out in QGIS (QGIS 2016) using the Semi-Automatic Classification Plugin (Congedo, 2016). Ten macro-classes were established (water, urban area, mud bank, mangrove, vegetation, cloud, exposed soil, cloud shadow, port, and aquaculture). For each estuary, maximum likelihood and minimum distance algorithms were applied. Then the accuracy of the classification was checked by looking at the satellite image to see if the class attributed to most of the pixels was valid. Moreover, the image classes were compared with a high-resolution satellite image from a similar date (http://www.google.com/earth/download/ge/). The algorithm that gave the most pertinent result was chosen: maximum likelihood for the Santa Cruz Channel and minimum distance for Suape and Sirinhaém. As the images were taken during different tide conditions, the mud flats that were exposed at low tide were classified as water in order to obtain the same pattern for high tide images.

Polygon processing

The classified raster images were converted into polygons and exported to a shapefile format. Manual cleaning was then applied using QGIS editing tools. The polygons that did not belong to the pre-established class were removed using the satellite image of the corresponding year as a visual reference. This hand editing method is often necessary after a supervised classification (Guimarães et al., 2010; Li et al., 2013).

Spatiotemporal patterns of changes in mangrove coverage

Using Themamap software (https://themamap.greyc.fr/), the mangrove polygons were converted into a grid to precisely accurately identify the areas that have undergone change. The use of a grid allows changes to be monitored at a local scale, and enables observations of how, when and where these changes occurred, at a spatial resolution finer than polygons. For each year the grid and the mangrove polygons were intersected, and a unique grid for all the years was obtained with a binary code representing the presence/absence of mangrove in each cell. The grid cells were 0.1 km by 0.1 km which was considered to be an appropriate unit given our study scale and the resolution of the satellite image used for the polygons. The spatiotemporal patterns of coverage mangrove modifications were analyzed mapping the mangrove that was lost, gained and remained stable over the three decades

Anthropogenic encroachment on mangroves

The encroachment of anthropogenic activities on mangroves was investigated and the polygons corresponding them were mapped. Anthropogenic activities considered to constitute the most direct threat to the mangrove forest were quantified: aquaculture, urbanization, the commercial port and agriculture. The vegetation and the exposed soil polygons in a radius of 1 km around mangrove polygons were used as a proxy to represent agricultural expansion. Industries were classified as urban because their spectral signatures were similar. To determine which activity had the greatest impact on mangrove, the intersection between the mangrove area at the beginning of the study (1990) and the area of each anthropic activity at the end of the study (2015) was calculated. The polygons generated by this intersection enabled us to estimate the overlap between the anthropic activities and the initial mangrove, being able to quantify mangrove substitution.

2.4. Results

2.4.1. Spatiotemporal patterns of mangrove coverage

Measuring the mangrove areas for each estuary in each image (Figure 3) helped assessing the spatiotemporal variations along the last three decades.

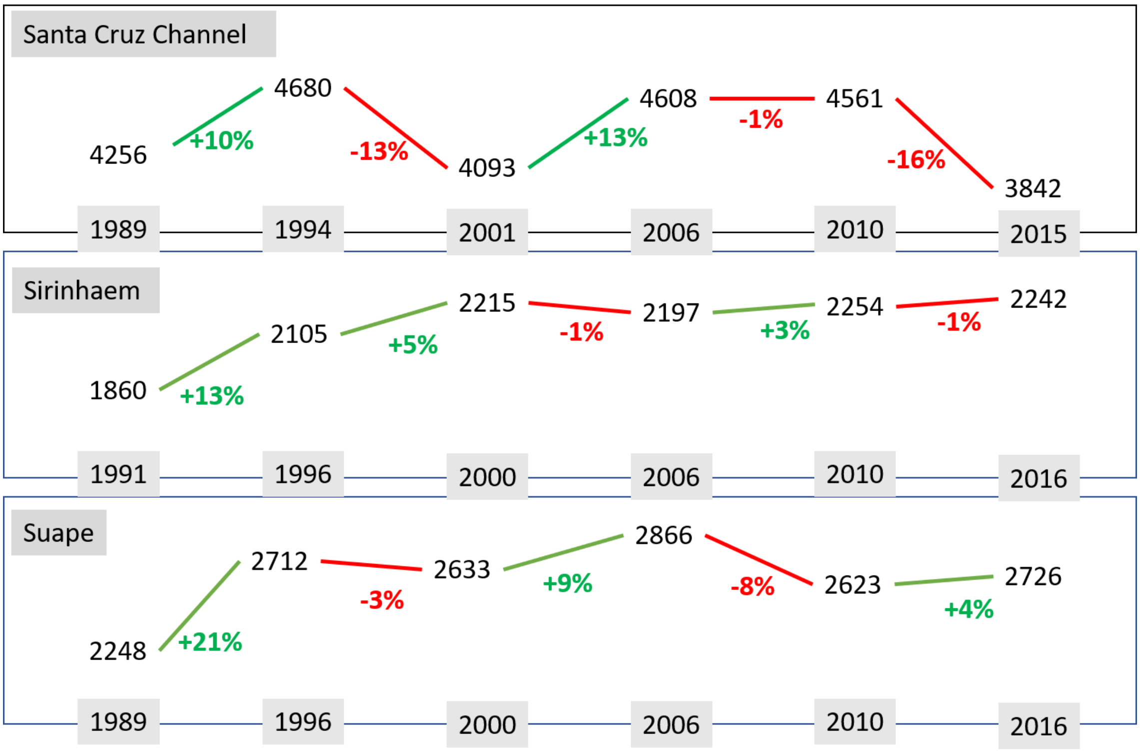


Figure 3: Slopechart of the mangrove areas of the three estuaries for each collected year and the evolution in percentage between two

The Santa Cruz Channel mangrove area increased between 1989 and 1994 and then decreased between 1994 and 2001. A stable period is noticeable between 2006 and 2010, followed by a decrease between 2010 and 2015. An increase of the Sirinhaém mangrove coverage can be observed between 1991 and 2000. After 2000, the coverage area remained relatively stable. A sharp increase in Suape mangrove was observed between 1989 and 1996. After this, fluctuations were minor, but an increase between 2000 and 2006 can be seen, followed by a decrease between 2006 and 2010.

The mangrove areas that were lost, gained and those which remained stable were mapped in order to see if the variations were localized in preferential areas (Figure 4).

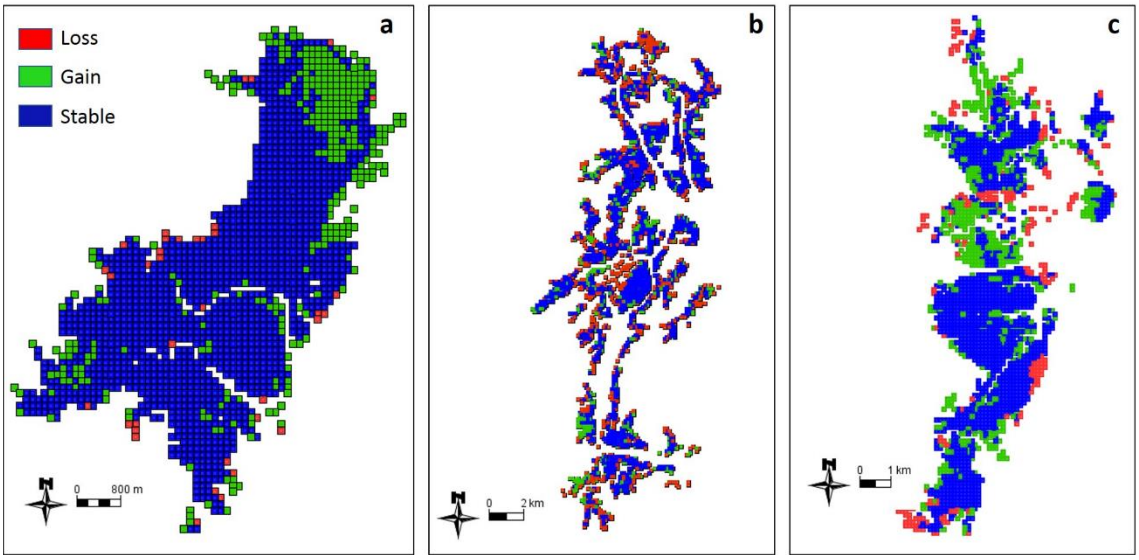


Figure 4: Maps of the mangrove areas that appeared, disappeared, and remained stable in (a) Sirinhae´m, (b) the Santa Cruz Channel, and Suape.

The three estuaries exhibit different spatial patterns in terms of mangrove coverage changes over the three decades. Mangrove appearances and disappearances in Sirinhaém occurred in the outermost mangrove ecosystem, and mostly in the north. The appearances consist in a large patch of mangrove in the northeast and few dispersed cells whereas the disappearances are insignificant over the three decades.

By contrast, the appearances in the Santa Cruz Channel are scarce and border the stable mangrove. A relevant area of mangrove disappeared in the center of the estuary and a substantial mangrove withdrawal is observable on the north and south edges.

Concerning Suape, the appearances of mangrove occurred mainly in the north and to a lesser extent in the south. A vast central area of new mangrove is noticeable whereas significant areas of mangrove were lost on the coastal side of the estuary.

2.4.2. Human encroachment on mangrove forests

Maps of human activity and mangrove spatial occupation over the last three decades were produced (Figure 5), enabling us to analyze the human impact on mangrove forests. Three years were chosen in each estuary to show landscape modification over time.

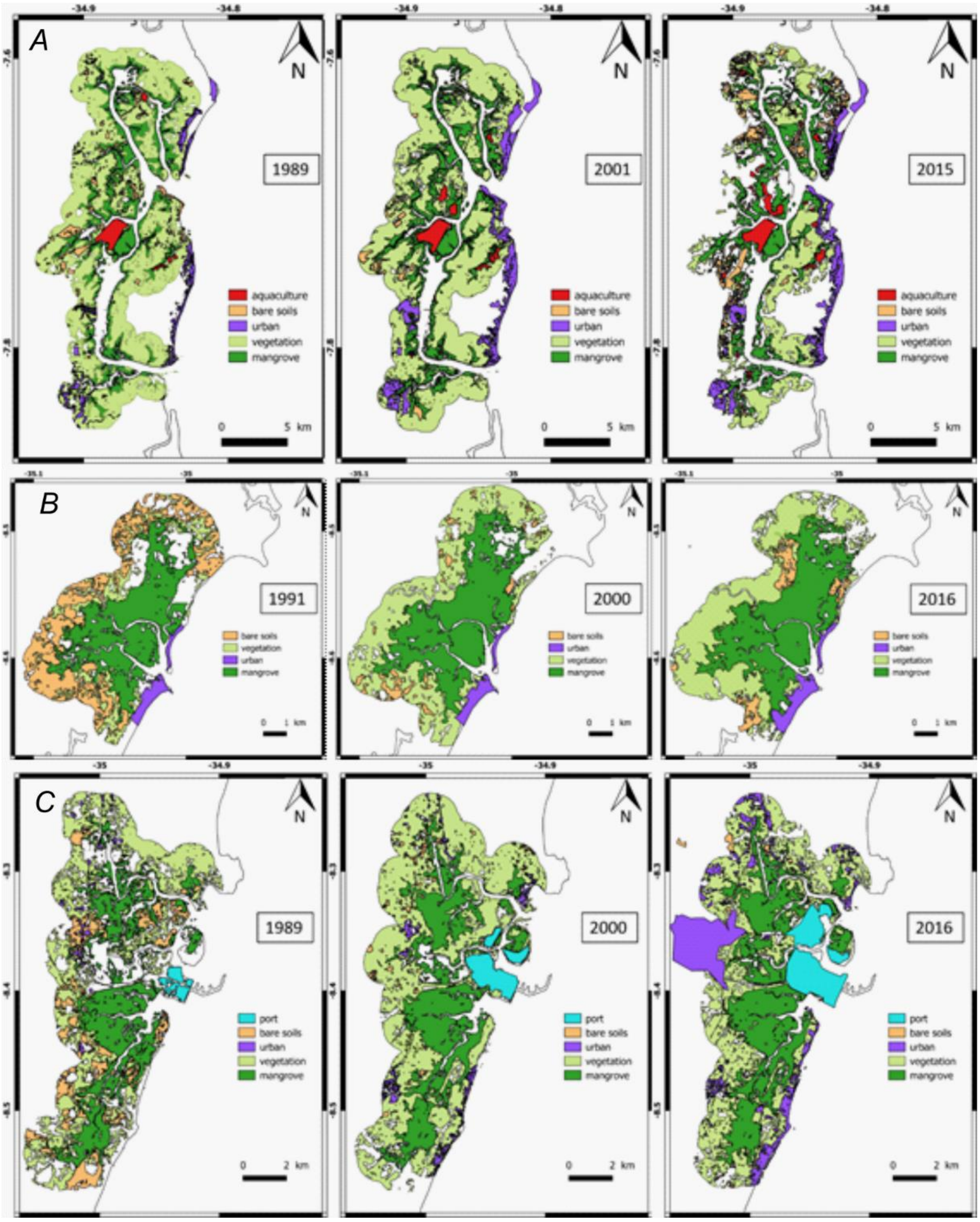


Figure 5: Evolution of mangrove areas and anthropic activities spatial occupation over the past three decades in (a) the Santa Cruz.

These images enabled a visualization of the impact of three main human activities that encroached on mangrove: aquaculture in the Santa Cruz Channel, the port in Suape, and agriculture in Sirinhaém. For the Santa Cruz Channel (Figure 5a), the conversion of mangrove into aquaculture has occurred in several zones distributed over the whole estuary but is more apparent in two major areas in the center. In Sirinhaém (Figure 5b), the entire edge of the mangrove forest is bordered by agriculture activity, leaving only a compact core of mangrove without direct exposure to this activity. In Suape, the encroachment of human activity is confined to a specific area in the central northern part of the estuary, where there has been gradual deforestation due to the construction of the industrial port (Figure 5c).

However, even if the other human activities didn't replace the mangrove, a general trend of anthropization is observable in the three estuaries and is more pronounced in Suape and the Santa Cruz Channel. Indeed, the urban area has increased significantly in both estuaries during these years and the Suape industrial complex, situated in the west of the mangrove forest, represents the anthropic activity which extended the most over the studied period.

The areas of the different land uses were measured over the three decades in the three estuaries (Table 2) in order to quantify the evolution of both the human activities and the mangrove coverage. It appears that the only estuary which suffered a mangrove reduction is the Santa Cruz channel with a 10% decrease of its mangrove area whereas in both Suape and Sirinhaém, a 20 % mangrove coverage increase occurred. An expansion of the human activities settled in the three estuaries is noticeable.

The aquaculture area, in the Santa Cruz Channel, increased by 92 % over the three decades, with the expansion beginning most noticeably after 1994. This activity had the greatest impact on mangrove in the estuarine complex, spreading over 3 % of the initial mangrove. The area dedicated to aquaculture increased sharply after 1994 but has remained stable since 2010, while the mangrove coverage gradually declined (Table 2). Furthermore, an increase of the urban area by 58 %, in this estuary, was accompanied by a decrease in vegetation and an increase of bare soils.

Meanwhile, in Sirinhaém 3 % of the mangrove was removed for the benefit of agriculture (2% by other vegetation and 1% by bare soils). Thus, the total area of exposed soils underwent an abrupt reduction after 1991 and was replaced by vegetation, while the total mangrove coverage increased. After this period, the areas of exposed soil, vegetation and mangrove remained relatively stable.

During the period of the study, the port in Suape substituted 2% of the initial mangrove. The most significant expansion of the port area occurred between 1996 and 2000, increasing from 159 ha to 355 ha along with a reduction of the mangrove. Meanwhile, the urban/industrial area sharply raised by 1494%, becoming the most significant human activity in the area. Nevertheless, even if Suape was the estuary where the anthropization was the most striking, the mangrove area still increased 21% from 2248 ha to 2726 ha.

Table 2: Areas in hectares of each land use for the three estuaries, their total evolution over the three decades and the proportion of mangrove that was removed for the benefit of each land use.

Estuary	**Land use**	**1990**	**2000**	**2015**	**Total change**		**Mangrove removal**
					ha	**%**	
Santa Cruz Channel	Aquaculture	345	474	662	+ 317	+ 92%	3%
	Bare soils	812	279	2486	+ 1674	+ 77%	0%
	Mangrove	4256	4093	3842	- 414	- 10%	-
	Urban	4690	7413	7432	+ 2742	+ 58%	0%
	Other Vegetation	12276	13796	5399	- 6877	- 56%	0%
Sirinhaém	Bare soils	803	212	249	- 554	- 69%	1%
	Mangrove	1860	2215	2241	+ 381	+ 20%	-
	Urban	145	145	199	+ 54	+ 37%	0%
	Other Vegetation	1323	275	2091	+ 768	+ 58%	2%
Suape	Port	118	355	683	+ 565	+ 479%	2%
	Bare soils	489	153	47	- 442	- 90%	0%
	Mangrove	2248	2633	2726	+ 478	+ 21%	-
	Urban	79	299	1220	+ 1141	+ 1454%	0%
	Other Vegetation	2916	4576	3521	+ 605	+ 21%	0%

2.5. Discussion

Protection measures for a mangrove area can only be pertinent if it is known how, where, and when mangrove coverage has changed. This study is the first application the grid technique for local scale monitoring of mangrove and it was effective for synthesizing changes in mangrove cover which can help implement conservation strategies. It is a parsimonious way to map the spatiotemporal patterns as they can be seen in two maps, one for the appearances and one for the disappearances. Unlike the gain-loss maps generated through the polygons overlapping technique, the grid technique enables to assess the modification rates for the overall period. Moreover, grids are more precise than polygons as the unit of study consists in smaller cells. They are therefore more adaptable for local scale monitoring. The ability to alter cell sizes also allows a good way to see modifications at different scales, which can then be attributed to processes operating at different scales. Further research is needed to test whether or not this technique can effectively determine scales of threats.

The grid technique is appropriate for spatially representing landscape processes and has previously been used for integrated river basin models (Rathjens et al., 2015). The precision of the technique, coupled with in-situ data allows to model several spatial aspects of animal or vegetation population structure (Royle et al., 2018). However, few studies are using this approach.

On another level, the distribution of different biotic elements is key to establishing connectivity, which is an omnipresent theme in current studies (Olds et al., 2017a). The grid technique could therefore bring powerful insights to seascape ecology as the hypothesis of coastal habitats being separate is giving way to the concept of an interconnected mosaic of habitats (Nagelkerken et al., 2015).

Mangrove coverage in the three estuaries fluctuated noticeably over the three decades of the study. An overall increase can be noted in Suape and Sirinhaém whereas, after a similar initial increase, the Santa Cruz channel mangrove has decreased during the last 5 years. Mangroves in all three estuaries are strongly influenced by salinization and the phases of mangrove increase can be linked to this. Salinization can stimulate mangrove growth by enabling its development in areas that were previously colonized only by freshwater plants (Teh et al., 2008). Wetland salinization is a process occurring at a global scale (Herbert et al., 2015a; Nachshon et al., 2014; Williams, 2001) and is related to climate change (Meiggs and Taillefert, 2011). During droughts, alterations in soils properties enable mangrove landward expansion (Rogers et al., 2006). Sea level rise also causes salt water intrusion into the estuaries (Church and White, 2006). In regions close to our study sites, such as the Metropolitan region of Recife, a 5.6 mm/year increase in sea level was observed between 1946 and 1988 (Neves and Muehe, 1995), which was above the mean world annual sea level rise at that time (Hay et al., 2015).

Furthermore, an increase in salinity can also be directly linked to anthropogenic activities (Cañedo-Argüelles et al., 2013a). This secondary salinization occurred in the Santa Cruz Channel between 1989 and 1994 due to extensive damming leading to the spread of mangroves (Lacerda and Marins, 2002). Another example of human-induced salinization is the establishment of the port in Suape, which altered the geomorphological and hydrodynamic conditions of the area, particularly in the Ipojuca River estuary (Koening et al., 2003). Muniz et al. (2005) sampled parameters including salinity, pH, dissolved oxygen, transparency, and temperature from three fixed stations in this estuary. They compared summer and winter before (1978) and after (1986–87) the port's construction, to analyze the effects of port-induced changes. A general increase of all the chemical parameters was observed post construction, with an increase in salinity having the greatest ecological impact. This increase is due to the Ipojuca river being diverted toward the ocean, causing a reduction in river-flow speed, sedimentation, and a diminution in the circulation and depth. As a result, the mouth of the river has been transformed into a coastal lagoon where evaporation and consequently salinity were high. This increase in salinity is likely to have favored mangrove development and enhanced the appearance of a recovery zone (Souza and Sampaio, 2001).

Because spatial patterns in each estuary are different, conservation measures must also take into account specific localized conditions within each of the three zones. Differences in mangrove dynamics between the estuaries can be explained by differences in anthropogenic activities and by the morphological shapes of the estuaries.

The Santa Cruz Channel is the only estuary which has suffered a loss in mangroves over the three decades. It is thus important to understand the evolution of land use in this estuary. The mapping anthropogenic activities demonstrated that aquaculture between 1994 and 2001 was significant in the center of the estuary. It adversely affected mangroves not just through direct encroachment leading to deforestation but also by induced physicochemical changes (Naylor et al., 2000). Toxic effluent and the alteration of sediment rates are two possible reasons for mangrove disappearance in zones that were not directly exposed to aquaculture.

The mangrove diminution observed in the Santa Cruz Channel between 2010 and 2015 could be attributed to pollution. Mercury contamination was already registered in the area as a consequence of the effluent from a chlorine and caustic soda plant which settled in 1963 and released between 22 and 35 tons of mercury by 1987 (Meyer, 1996). However, high mercury concentrations were encountered recently in the sediment of the Santa Cruz Channel (De Moura & Candeias, 2009). The values were higher than those found in 1981 by the Compania de Tecnologia de Saneamento Ambiental. Those results indicate recent inputs or a recent movement of the accumulated mercury toward the Santa Cruz Channel. As mercury is accumulated in sediment, its transport depends on environmental factors. As a consequence, the aquatic system can remain contaminated for many years after any mercury input in the area (De Moura & Candeias, 2009). Yet, mercury can prevent the survival of mangrove seedlings (Huang and Wang, 2010) and can also affect mangrove growth by causing a depletion of nutrients, since high concentrations threaten the micro-organisms that guarantee the biological conversion of minerals (Ravikumar et al., 2007).

The port construction is the main driver in the spatial dynamic of the Suape mangroves, whether through direct deforestation, salinization and sedimentation that would explain the fast recovery of the mangrove. The other drastic anthropic landscape modification during the studied period, is the growth of urban areas and the concomitant removal of other vegetation. Indeed, urban area is currently the predominant land use around the mangrove. This constitutes an additional chemical threat, as more and more urban effluent will be discharged near the mangroves. A recovery area in the center and the instability of the coastal zone were observed, emphasizing that the modifications were localized. This configuration enables priority zones for protective measures to be determined.

Mangrove forest shapes could also, in part, explain the differences observed in mangrove spatiotemporal coverage patterns as the stability of a forest, and its susceptibility to external pressures, depends on its geometrical features (Hessburg et al., 2005). Mangrove appearances and disappearances in the Santa Cruz Channel were distributed all along the estuary and formed several small aggregates. This is possibly due to its geometrical properties, which presented as elongated shapes, that could increase its vulnerability to salinization, resulting in the appearance and disappearance of areas suitable for mangrove growth. This highlights a lack of stability in this zone and suggests that the protection of targeted zones might be inappropriate: conservation measures need to be implemented at the estuary scale.

Mangrove stability in Sirinhaém could also be related to mangrove structural characteristics. Patch size and proximity to mangrove cells influence mangroves' ability to resist habitat loss since where the core forest is sizable, the stand as a whole is less vulnerable (Harper et al., 2007). Hence, the important patch size of Sirinhaém mangrove can confer a resistance to anthropogenic threats because the main part of the forest is not exposed–unlike in a more parceled landscape. The compactness of Sirinhaém's mangroves thus explains their overall growth, despite surrounding threats, since the survival of the vegetation is a function of the density of the initial forest (Kéfi et al., 2007).

2.6. Implications for conservation

In this study, the spatiotemporal patterns of changes in the mangrove cover were mostly determined by anthropic activity. Aquaculture, agriculture, urbanization and harbor construction cause deforestation, release toxic effluents inhibiting mangrove growth and induce salinization enhancing mangrove development. Despite of all these threats, our results showed a trend of mangrove growth in two of the three estuaries over the last three decades. Therefore, it seems that their resilience to disturbance was linked to their geometrical properties and to a favorable change of salinity conditions.

Indeed, salinization, mostly induced by port construction, damming and sea-rise level, had a positive effect on mangrove. Nevertheless, it could endanger the nursery function of mangrove since freshwater or estuarine fish can be adversely affected by increases in salinity, and this physiological stress could force them to migrate toward less saline areas (Nielsen et al., 2003). Salinization, could also constitute an ecological threat, decreasing inorganic processes of nitrogen removal, reducing carbon storage and increasing the release of toxic sulphides which impact nutrient cycles and climate regulation (Herbert et al., 2015). Since the secondary salinization that occurred in our study area, could have an impact at individual, population, community and ecosystem levels (Cañedo-Argüelles et al., 2013), mitigations measures should be contemplated.

Moreover, before the implementation of any conservation measures or, indeed, any further anthropogenic activities, the geometrical properties of the mangrove should be considered in order to evaluate future impacts. Although our study among others discussed the importance of geometrical features, few works on wetlands have focused on their implications for conservation planning to date (Boström et al., 2011). Mangrove spatial configurations (patch size and patch dispersion) can also determine faunal distribution (Pittman et al., 2004), therefore alterations to those features could impact biodiversity.

As these habitats are important for artisanal fisheries, our results can be used to elaborate concrete conservation strategies. Our mapping of the landscape modifications can indeed be useful, since the anthropization had a relevant impact on the mangrove patterns. Aquaculture, agriculture and port construction were established as three anthropogenic modifications that directly threaten the mangrove. Moreover, the expansion of the activities that don't encroach on the mangrove was quantified and in Suape, the urbanization was identified as a potential threat. Avoiding the spatial expansion of these activities should be a priority.

The grid technique was an effective method to investigate mangrove coverage changes at a local scale. Its precision helped to follow adequately spatiotemporal cover changes and relate them to anthropogenic activities. Hence, this method appears to be useful for conservation planning and future remote sensing work at local scale.

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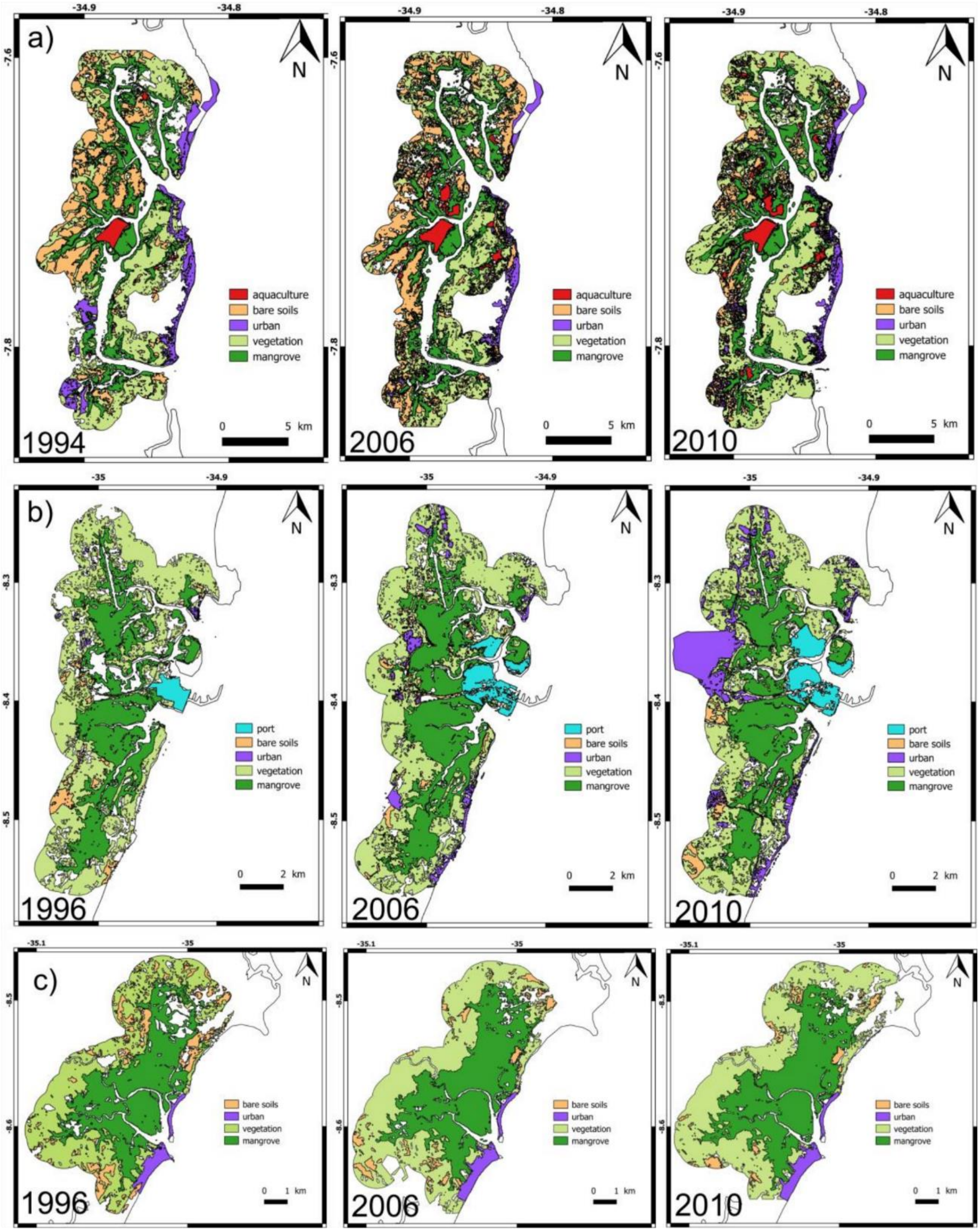
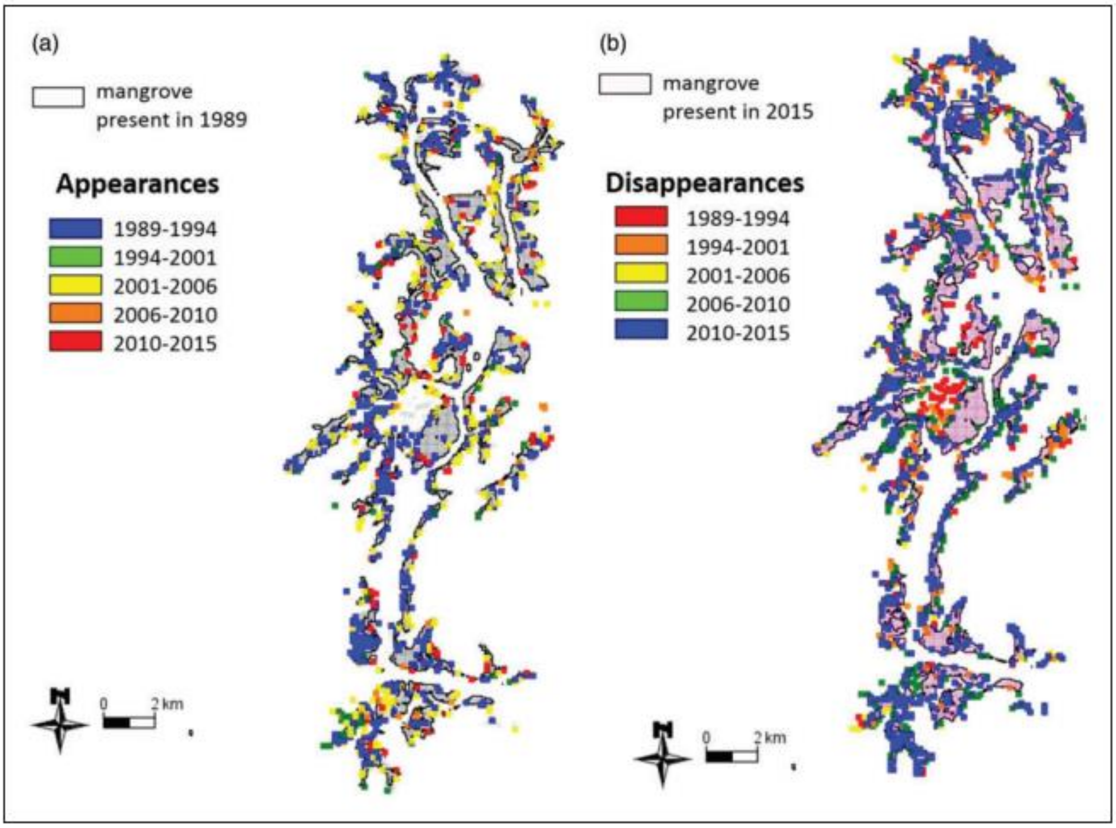
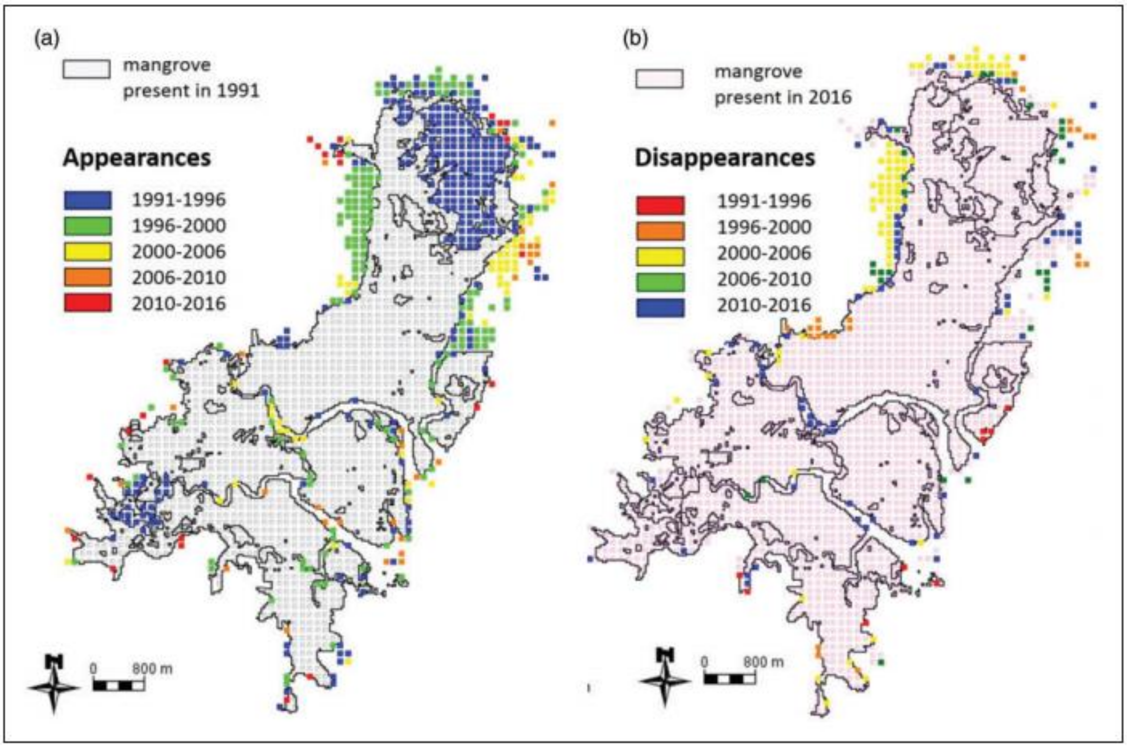
Supplementary material:



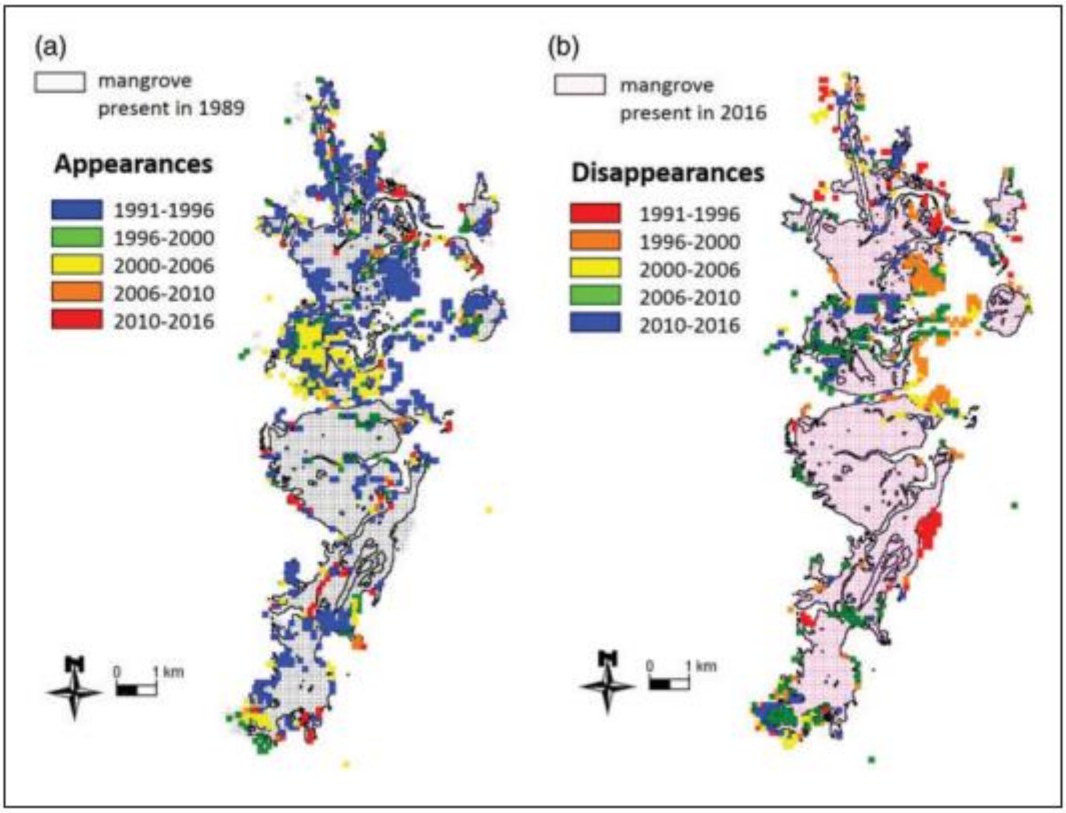
Supplementary material 1: Evolution of mangrove area and anthropic activities spatial occupation for the images that were not presented in the results in (a) the Santa Cruz Channel, (b) Suape, and (c) Sirinhaem.



Supplementary material 2: Maps of the mangrove areas that (a) appears and (b) disappears for each period in the Santa Cruz Channel



Supplementary material 3: Maps of the mangrove areas that (a) appears and (b) disappears for each period in Sirinhaém.



Supplementary material 4: Maps of the mangrove areas that (a) appears and (b) disappears for each period in Suape.

Importance of estuarine morphology for coastal connectivity: a case study in Brazilian tropical estuaries

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3. Importance of estuary morphology for ecological connectivity with their adjacent coast: a case study in Brazilian tropical estuaries.

3.1. Abstract

Coastal habitats are interlinked by ecological connectivity, defined as the exchange of organic matter or organisms between habitats. The degree of this connectivity will depend in particular on the geometric properties of the landscape. The increasing anthropogenic alterations in estuaries therefore raises the need to understand how their morphological characteristics influence fluxes between habitats. We used stable isotopes to investigate the variability of ecological connectivity between three estuaries and their adjacent coastal areas by tracking the origin of the organic matter (estuarine vs coastal) underlying the diet of the migrant species *Eucinostomus argenteus*. The chosen estuaries were geomorphologically distinct, exhibiting, in particular, differences in their degree of connection to the sea, corresponding to the morphological features (shape, mouth width) controlling key physico-chemical variables in this habitat (e.g. salinity). The sampling of the basal food sources contributing to the food web was performed in the three estuaries and in their adjacent coasts. The variability in stable isotope ratios between estuaries was examined for both fish and sources of organic matter. In the three estuaries, Bayesian models were applied for each season to quantify the relative contribution of sources from estuarine and coastal environments supporting the diet of the silver mojarra. The share of coastal organic matter increased with the degree of sea connection, indicating that the properties of the seascape can regulate the intensity of interactions between ecosystems. Variations in ecological connectivity are likely to affect the functioning of ecosystems as they influence trophic pathways and energy flows between adjacent habitats. Morphological modifications could thus significantly disturb ecosystems by altering the structure of food web, thereby affecting certain ecosystem services such as the availability of living marine resources.

3.2. Introduction

The emergence of seascape ecology brought the need to consider coastal habitats as a single continuum rather than as separate biomes (Green et al., 2012; Berkström et al., 2013). This conceptual change aimed at a more integrated management of the coastal zone (Beger et al., 2010). In particular, addressing the connectivity between habitats is essential for fisheries management because coastal habitats provide complementary ecological functions that are critical for marine living resources (Sheaves, 2009). Coastal habitats should therefore not be considered as isolated patches in which different sub-populations occur, since a comprehensive view reflects population dynamics more accurately (Burgess et al., 2014).

Ecosystem interactions can be primarily subdivided into biological, chemical and physical interactions (Ogden, 1997). The connectivity degree between coastal habitats will depend in particular on the geometric properties of the landscape (Olds et al., 2017). More specifically, landscape characteristics affect ecological connectivity which consists of the Interactions between ecosystems through the movement of organisms and the exchange of nutrients and organic matter that are involved in ecological processes within these systems (Nagelkerken, 2009). Previous work demonstrated that variability in the geometric characteristics of the seascape, such as the distance between habitats, can affect the intensity of flows between habitats (Mumby et al., 2004; Berkström et al., 2013).

Estuaries include a wide range of different transitional water bodies with distinct landscape characteristics (Flemming, 2011). Thereby, the classification of estuaries requires the consideration of physico-chemical variables such as hydrodynamics and bathymetry, as they determine not only the dynamics and structure of the sediments, but also the nature of the primary producers (Elliott & McLusky, 2002; Whitfield & Elliott, 2011). Consequently, the morphology of estuaries and, in particular, the set of geometrical variables controlling the degree of sea water dilution in the estuary (like depth and mouth width), could have an influence on ecological connectivity. This morphological variability, which can be viewed as their degree of connection to the sea, is likely to enhance or prevent flows with adjacent coastal habitats. In addition, estuarine morphologies suffer from increasing anthropic alterations, such as the construction of polders, harbours or dykes which can drastically change the sedimentary dynamics and the composition of estuarine biological communities (Wetzel et al., 2012; Du et al., 2016; Lechêne et al., 2018). It is thus relevant to investigate how estuarine morphological characteristics influence interactions between habitats. The implications of this variability in terms of ecosystem functioning need to be taken into account before deciding to modify seascape features or implement restoration measures through eco-engineering (Elliott et al., 2016).

Understanding ecological connectivity involves studying energy pathways in food webs and their intrinsic trophic relationships in order to assess community structure and functional role of species living in the ecosystems (Pasquaud et al., 2010; França et al., 2011). Hence, estuarine food webs can be described by characterizing trophic relationships, sources of organic matter and energy flows between system components (Pasquaud et al., 2008). In these complex and dynamic ecosystems, food webs can be supported by the production of various local primary producers, as well as the transport of organic matter from adjacent coastal and riverine areas (Choy et al., 2009; Selleslagh et al., 2015). However, identifying the origin of the organic matter at the base of estuarine fish food web can be difficult, in particular because conventional methods like gut content analysis provide only a snapshot of the diet at a given time (Pasquaud et al., 2008 ; Pasquaud et al., 2010; França et al., 2011). Moreover, gut analyses do not enable a comprehensive food web analysis to reliably track the source of the organic matter underlying the diet of high trophic level consumers that do not feed directly on primary producers.

Stable isotope methods produce estimates of trophic position that can both capture complex trophic interactions and track energy flows between habitats (Carvalho et al., 2017; Whitney et al., 2018; Gonzalez et al., 2019). Since a consumer's stable isotope ratios reflect the values of its food sources, trophic position assessment is possible if the differences in isotopic composition between an animal and its prey, i.e. trophic level enrichment, are known (Caut et al., 2009). Indeed stable isotopes of nitrogen (δ15N) and carbon (δ13C) have been successfully used to study ecosystem functions and food webs (Le et al., 2018). δ15N can constitute a proxy for the trophic position of an organism since it increases considerably with the trophic level (Post, 2002; Caut et al., 2009). On the other hand, δ13C differs substantially between primary producers (Post, 2002; Herzka, 2005), providing an overview of the origin of organic matter (Fry, 2006) .

In estuaries, the origin of organic matter has already been studied with stable isotopes and a high contribution of marine basal sources has been highlighted in some studies

(Pasquaud et al., 2008; Selleslagh et al., 2015). Nevertheless, other studies conducted in estuaries have shown that in situ primary production can outweigh other food sources and contribute significantly to fish growth (Lobry et al., 2008). Therefore, knowledge on the extent of ecological connectivity between estuaries and their adjacent coastal areas is essential for designing the scale of conservation measures, particularly in the case of migratory species using multiple habitats (Vasconcelos et al., 2010; Reis-Santos et al., 2018).

Migratory species constitute a good proxy to study ecological connectivity between two habitats since their movements represent linkages across the seascape (Selleslagh et al., 2015). In this study, we chose *Eucinostomus argenteus* Baird & Girard (1855) to investigate the intensity of the flows between habitats. The silver mojarra is a member of the Gerreidae family comprising species with complex life strategies that are important for artisanal fisheries in north-eastern Brazil (Pinto et al., 2013). *E. argenteus* is one of the main Gerreidae in the region and can be classified as an estuarine-dependent species, as its juveniles are found in great abundance in the estuarine habitats (Potter et al., 2013) where they reside since their reduced size at this ontogenetic stage (less than 13.5 cm) (Bouchon-Navaro et al., 2006) does not allow them to make large migrations (Franco et al., 2012). In those habitats, they were classified as second-order consumers, feeding opportunistically as well on microcrustaceans (amphipods, copepods, tanaidaceous, ostracods) as on detritus with variations in the proportion and frequency of different items according to their ontogeny and food availability (Chi-Espínola et al., 2018). Although adults of these species are more abundant in adjacent coastal areas (Ramos et al., 2016), they use estuaries as a feeding ground, predating mostly on Bivalvia siphons and polychaetes (Vasconcellos et al., 2018) or as a reproductive ground (Chaves and Bouchereau, 2000). The degree of ecological connectivity between estuaries and adjacent coast would be reflected by the dominant origin of the carbon sources (coastal vs estuarine origin) on which the species diet is based.

The objective of this study is to understand the variability in ecological connectivity between estuarine and coastal habitats. We are thus investigating the flows between three morphologically distinct estuaries and their adjacent coastal areas by tracking the origin of the organic matter (coastal or estuarine) underlying the diet of *E. argenteus*. Our hypothesis is that geomorphological disparities between estuaries influence the origin of the organic matter that supports the *E. argenteus* regime, based on the assumption that flows are enhanced in the estuaries most connected to the sea.

3.3. Material and methods

3.3.1. Study area

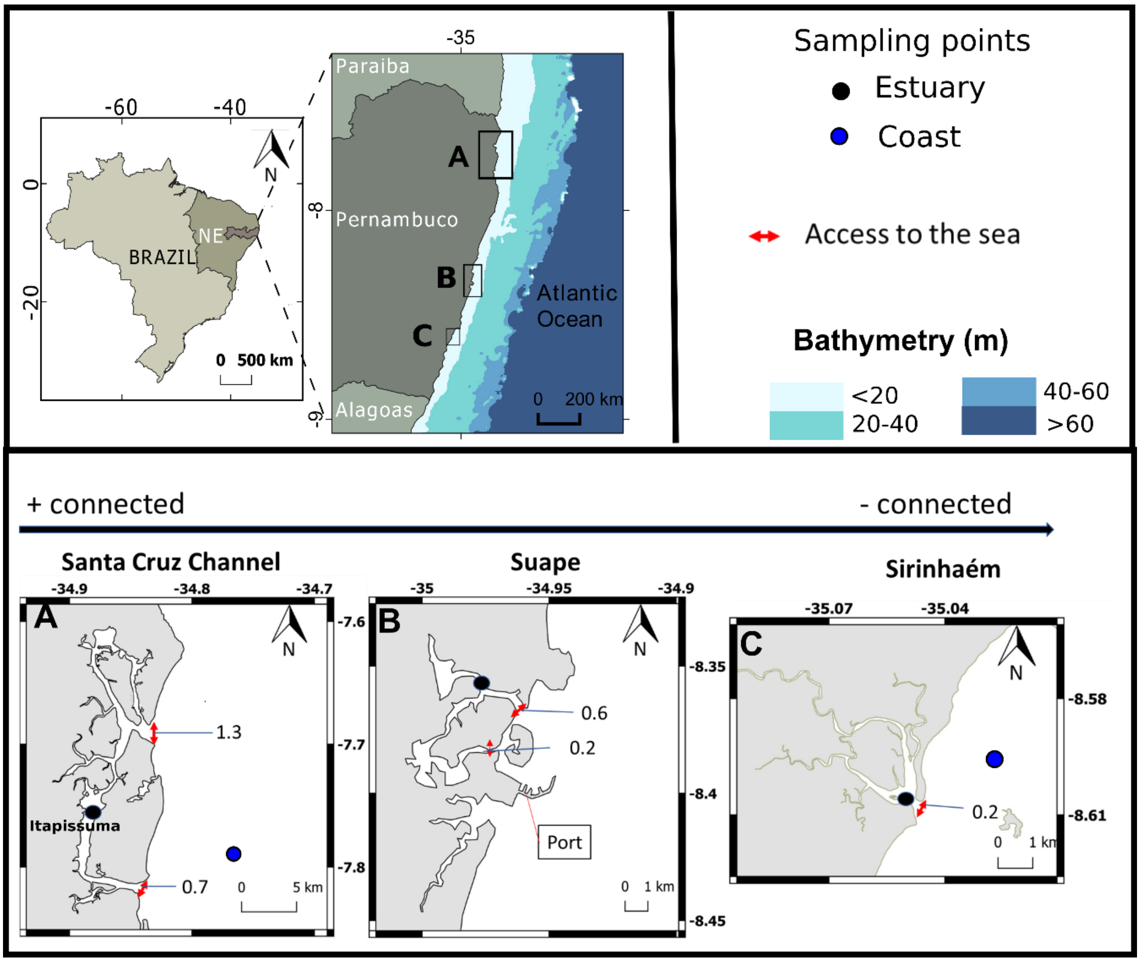

Figure 1: Map of the area under study showing the three estuaries (Santa Cruz Channel, Suape and Sirinhaém) and their adjacent coast where sampling was performed in the state of Pernambuco, Brazil. The widths of the mouths are indicated in kilometers for each estuary to show their geomorphological disparities, providing them with different degrees of connection with the sea

The area under study encompasses estuarine areas and shallow coastal waters of the continental shelf of the state of Pernambuco, Brazil (Figure 1). This narrow continental shelf (35 km on average),is characterized by shallow and warm waters (typically 26-29°C), soft slope, shelf break between 50 and 60 m, high salinity (>36) and a sedimentary cover composed of carbonatic and terrigenous sediments (Manso et al., 2003; Assunção et al., 2020). Three estuaries, the Santa Cruz Channel, Suape and Sirinhaem, were chosen because of their diverse morphological characteristics, which differ in particular in the geometric variables controlling the degree of dilution of the sea water in the estuary (such as the depth and width of the mouth) (Table 1, Figure 1). This different degree of connection to the sea is manifested notably by a difference in salinity (Table 1).

Table 1: Summary of morphological characteristics and anthropogenic activities of the three estuaries along the coast of Pernambuco, Brazil.

	+ connected	**Estuary**	- connected
Characteristics	**Santa Cruz**	**Suape**	**Sirinhaem**
Type	Ria	Coastal lagoon	Coastal plain
Estuary (km²) †	73.5	29.7	18.7
Vegetated area (km²) †	48.0	23.1	17.0
Watershed area (km²) †	25.5	6.6	1.7
Mean depth (m)	7.5 (North) /3.0 (South)	3.1	2.6
Mouth width (km) †	1.3/0.7	0.6/0.2	0.2
Salinity (mean± SD) ‡	31.2 ± 2.9	17.8± 2.4	9.6 ± 3.7
Pluviometry (mm; mean± SD)	1517 ±122	1869±367	2053±699
Activity	Aquaculture, industrial and domestic waste	Industrial harbour, industrial and agriculture waste	Industrial domestic and agriculture waste
Reference	(Medeiros & Kjerfve, 1993; Guimarães et al., 2010; Medeiros et al., 2001; Silva et al., 2011)	(CPRH, 2001; Borges, 2011; Silva et al., 2011)	(CPRH, 2001; Silva et al., 2011)

† Data obtained with the software ArcGis 10.1 based on satellite images from LANDSAT 7. ‡ Personal data collected during surveys in 2015. Type: geomorphological classification on the type of estuary according to Pritchard's classification. Estuary: estuary's area as the sum of mangrove and watershed areas. Salinity: Salinity at mid-depth of the water column. Pluviometry: Annual mean between 2014-2018 measured in stations near sampling points (APAC, 2020). Activity: existent anthropogenic activity with potential impact on the estuary.

The Santa Cruz Channel is located on the north coast and forms the largest estuarine complex in Pernambuco. It displays a U-shaped surface area of 22 km and has two accesses to the Atlantic Ocean (Silva et al., 2011). These two mouths are 0.7 km and

1.3 km wide, respectively, providing it with a high degree of connection with the open sea, resulting in high salinity (Table 1). In addition, the city of Itapissuma is the main fishing centre of Pernambuco, and also one of the most touristic destinations of the region. (Quinamo, 2006). Aquaculture, artisanal fishing, industry and agriculture are all important anthropogenic activities in the area (Moura and Candeias, 2009).

The Suape estuary, situated in the south coast, is a coastal lagoon with a flattened geomorphological shape (Santos and Costa, 1974). The geomorphological and hydrodynamic conditions were altered by the construction of the port in 1979 (Muniz et al., 2005). The generated landfills blocked the flow of two of the four rivers that previously drained into the Suape estuary. Suape is now formed by two main rivers that flow into a large brackish lagoon and has a restricted connection to the ocean.

Also located in the south littoral of Pernambuco, Sirinhaem is the smallest estuary among the three and presents the narrowest connection to the sea (CPRH 2001). This enclosed lagoon exhibits low salinity and low depth (Table 1). Artisanal fishing, agroindustry and trade are important socio-economic activities in the zone (Valença et al., 2010).

3.3.2. Sampling

Fish and baseline

E. argenteus was captured in the three estuaries above described, in shallow areas (Figure 1). The specimens were collected in 2015 from March to August and from September to January, which represent the local rainy and dry seasons, respectively (Medeiros et al., 2001) with the help of local fishermen using artisanal fishing gears. All gears characteristics are detailed in the Supplementary Table 1. After identification, all fish were measured to the nearest millimetre (Total length) and weighed to the nearest gram. In addition, a long-lived primary consumer (mangrove oyster *Crassostrea rhizophorae*) was collected to represent the baseline for estimating the trophic position of *E. argenteus*.

Organic matter sources

Organic matter sources were sampled in the estuaries, to provide a background of stable isotope ratios at the base of *E. argenteus* diet. Since mojarras are characterized by their highly protrusible mouth which they use to feed on items they remove from the sediment (Chen et al., 2007), sources associated to the benthic system were selected. Thus, macroalgae (*Sargassum sp*. and *Ulva sp*.), microphytobenthos, rotten mangrove leaves and particulate organic matter in the sediment (SOM) were collected along with seagrass when present in the area.

The SOM was also collected from the adjacent coast (shallow coastal waters) of the Santa Cruz Channel and Sirinhaem to measure the influence of coastal sources. However, as an industrial port is settled in the coast in front of the Suape estuary, the sampling of the coastal SOM could not be realized due to the restricted access to the area. We thus chose to use the coastal SOM isotopic ratios of Sirinhaem as a proxy for Suape considering the vicinity of the two areas.

The rotten mangrove leaves, seagrass and macroalgae were picked up manually during low tide. The SOM was collected from the first 2 mm layer of sediment while microphytobenthos (i.e. benthonic microalgae) was sampled from the sediment surface at low tide and extracted in the laboratory.

3.3.3. Isotope analysis

For stable isotopic analysis, a fragment of dorsal white muscle from each fish was extracted and cleaned with distilled water to remove any remaining scales or bones. The isotope analysis of the basal sources was performed on the whole collected sample. The material was dried in an oven at 60°C for 48 hours. Then, dried samples were grinded into a homogeneous fine powder with a mortar and pestle. The SOM samples were divided into two subsamples. One was acidified to remove inorganic carbon before the δ^{13}C analysis and the other was used for δ^{15}N analysis (Pinnegar and Polunin, 1999). Analysis of the carbon and nitrogen isotope ratios were performed at the Pôle de Spectrométrie Océan (PSO - IUEM, Plouzané, France) with an elemental analyzer (Thermo Flash 2000, interface Thermo ConFio IV) interfaced to a mass spectrometer (Thermo Delta V+).

The isotope ratio for carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) were calculated from the relation between the sample isotopic value and a known standard:

$\delta^{13}C$ or $\delta^{15}N = [(R\text{sample} / R\text{standard})-1] \times 1000$.

In this equation, R represents the ratio $^{13}C/^{12}C$ or $^{15}N/^{14}N$. The R standard is relative to international standards, the Air and V-PDB (Vienna PeeDee Belemnite) for nitrogen and carbon, respectively. The precision of the analysis was monitored through a known standard (Thermo – Acetanilide) every six samples and defined as ± 0.11‰ (standard error) and ±0.07‰ for carbon and nitrogen, respectively.

3.3.4. Statistical analyses

All the statistical analyses were performed with the software R version 3.5.2 (R Core Team, 2019). Mann Whitney tests were carried out to determine if there were any differences in $\delta^{13}C$ and $\delta^{15}N$ between the estuarine and coastal SOM. Spearman ranking tests correlation tests were performed to examine the relationship between fish $\delta^{13}C$ or $\delta^{15}N$ and total fish length in each estuary separately and for all estuaries together. Mann Whitney tests were applied to check whether the $\delta^{13}C$ and $\delta^{15}N$ of the fish varied seasonally in each estuary.

The trophic positioning of *Eucinostomus argenteus* was determined using the tRophicPosition package ("tRophicPosition: Bayesian Trophic Position Calculation with Stable Isotopes" (Quezada-Romegialli et al., 2018).The trophic position estimate is influenced by the baseline's representativeness of the target consumer food chain (Post, 2002). In view of the zoobenthivorous diet of the species studied, we proposed an estimate based on a long-lived filter feeder as a reference base (*C. rhizophorae*).

Kruskal-Wallis tests together with Dunn's post hoc tests were used to see if there were differences of $\delta^{13}C$ and $\delta^{15}N$ for the fish and the baseline (*C. rhizophorae*) among the different estuaries.

Moreover, the Standard Elliptical Area (SEA) method was applied to compare the isotopic composition of the fish caught in each estuary. The SEA encompasses about 40% of the data (mean and standard deviation of covariables $\delta^{13}C$ and $\delta^{15}N$) and provides an appropriate method for comparing the isotope niche (variability in isotope ratios) of groups belonging to several communities by eliminating extreme values that would bias the analysis if only the amplitude was considered (Jackson et al., 2011). The SEAc (corrected standard ellipse area) was calculated to overcome the underestimation for small sample sizes (Jackson et al. 2011). The SEAc was estimated for each estuary and season. These analyses were performed with the Stable isotope Bayesian analysis in R (SIBER) package (Jackson et al., 2017).

Bayesian stable isotope mixing models, using the MixSiar package (Stock & Semmens, 2015) were built to estimate the relative contributions of basal sources to *E. argenteus* diet in each estuary. MixSiar allows the inclusion of covariates to better understand the variance among both consumer and source tracer values (Stock et al., 2018). As estuarine environments are highly dynamic, the factor season was added for both the fish and the sources. The use of separate models for each season would have increased the residual error (Stock et al., 2018). The input data were ratios of carbon and nitrogen from fish along with the respective mean and standard deviations of sources for a given estuary and season. When sources were not significantly different between seasons, the annual average for both the rainy and dry seasons was used. Mixing models are highly influenced by the selection of a proper trophic enrichment factor (TEF) (Parnell et al., 2010). The TEF consists in the isotopic fractionation from source to consumer (Post, 2002). Since this TEF varies with both the diet and the habitat (Bunn, Leigh, and Jardine 2013), we chose a TEF proper to apply between estuarine zoobenthivorous fish (which is a second-degree consumer) and its basal sources of organic matter (primary producers): 2.0 ± 0.6 ‰ and 5.6 ± 1.5 ‰ for $\delta\, ^{13}C$ and $\delta\, ^{15}N$, respectively (see Kostecki et al., 2012; Selleslagh et al., 2015). As the $\delta^{15}N$ of fish in each estuary were not significantly different between seasons (Supplementary Table 2), the same TEF was applied for both seasons assuming that *E. argenteus* fed at the same trophic levels during the dry and rainy seasons. The signatures of the sources after application of TEF were represented on the same two-dimensional space as the isotope ratios of the consumers in order to verify a geometric prerequisite for the execution of Mixsiar. The mixing models were only run when the consumer isotope ratios were within the mixing polygon bounded by the base source signatures representing the trophic limits of the system (Phillips and Gregg, 2003).

3.4. Results

3.4.1. Fish isotopic variability and trophic position

A total of 46 samples of *E. argenteus,* 25 *C. rhizophorae* samples and 112 of basal food sources had their $\delta^{13}C$ and $\delta^{15}N$ isotope values analysed (Supplementary Table 2 and 3). Estuarine fish size ranged from 3.6 cm (in Suape) to 16.6 cm (in Sirinhaem) (Figure 2). The size of the fish was not correlated to their $\delta^{15}N$ nor their $\delta^{13}C$ values when considering all estuaries together (Supplementary Table 4). However, fish size in the Santa Cruz Channel was positively correlated with their $\delta^{15}N$ values and the fish size in Suape were negatively correlated with their $\delta^{13}C$ (Supplementary Table 4, Figure 2).

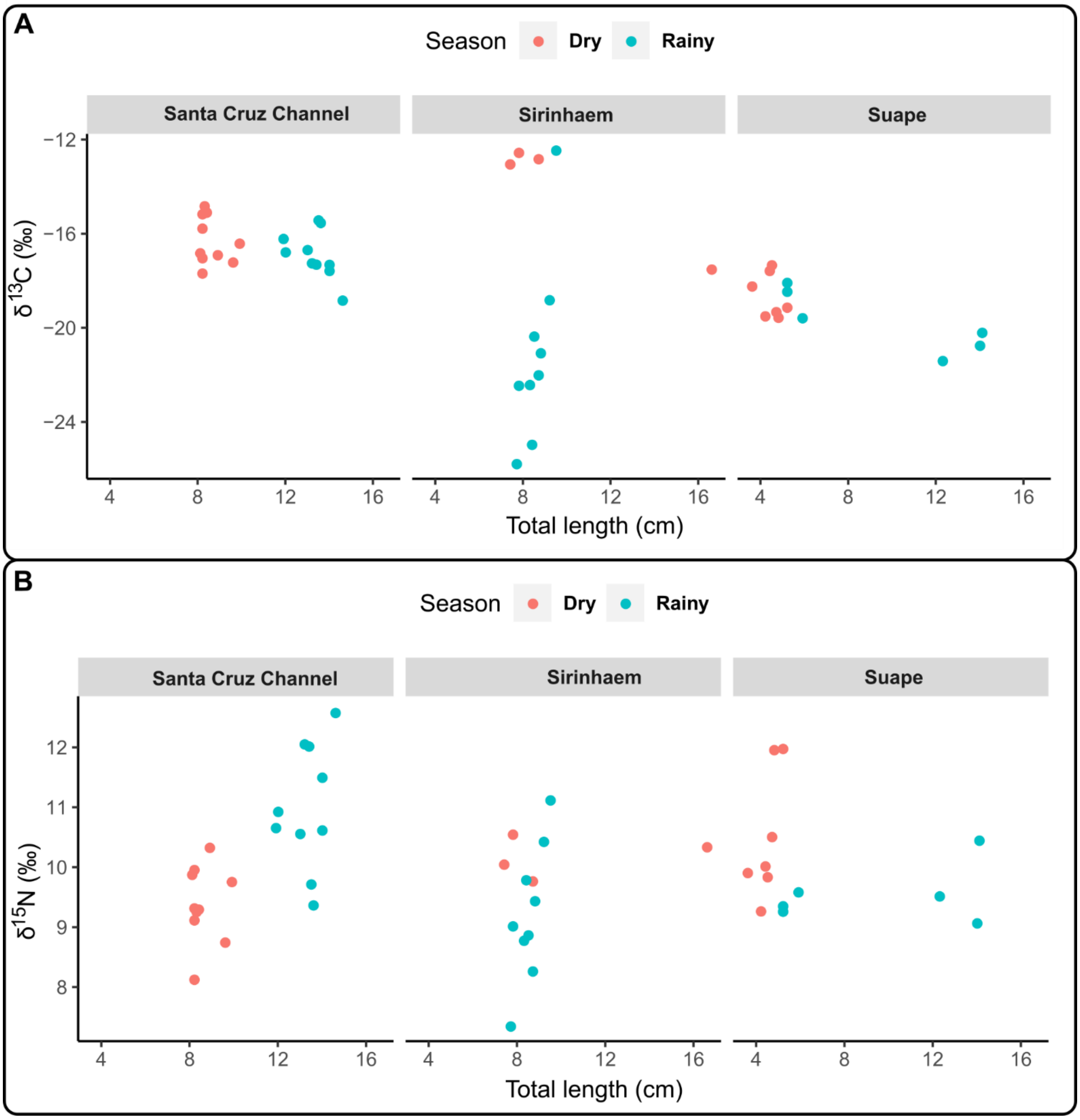

Figure 2: Biplot showing A) the δ^{13}C and B) the δ^{15}N of fish according to their total length in in each estuary (Santa Cruz Channel, Sirinhaem and Suape) during both seasons (dry and rainy).

Seasonal variations in terms of δ^{15}N were only observed for fish captured in the Santa Cruz Channel (Supplementary Table 5). Seasonal differences in δ^{13}C were detected only for Sirinhaem (Supplementary Table 5, Figure 2, 3). In addition, Sirinhaem was the only estuary where the ellipses of both seasons did not overlap. However, the overlap between the rainy and dry ellipses was small for Suape and the Santa Cruz Channel.

Sirinhaem ellipse during the rainy season completely overlaps the Suape ellipse for the rainy season and overlapped an important part of Suape ellipse for the dry season.

Sirinhaem fish collected during the rainy season had the most depleted $\delta^{13}C$, followed by the Suape fish in both seasons (Figure 3) while the $\delta^{13}C$ of the Santa Cruz Channel fish caught in both seasons exhibited more enriched $\delta^{13}C$ values, comparable to those of the coastal SOM. Fish caught in Sirinhaem during the dry season had the highest values of $\delta^{13}C$. The most elevated values of SEAc were observed for Sirinhaem during the rainy season (10.1‰²). All the other estuaries /seasons values exhibited low SEAc (between 1.9 and 3.7‰²).

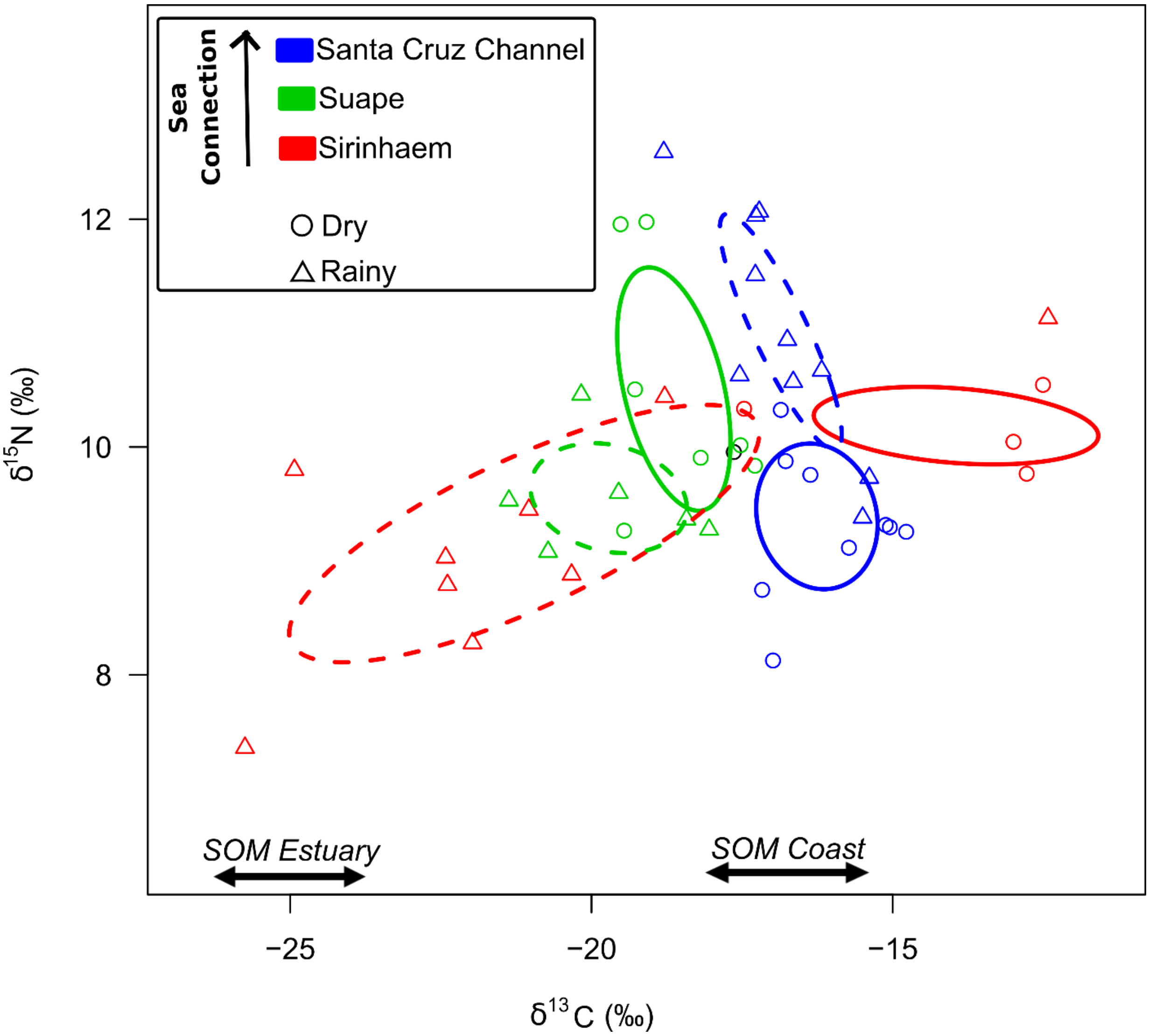

Figure 3: *Fish standard ellipses of the three estuaries during both dry (solid line) and rainy (dashed line) seasons. The black points represent the fish sampled in coastal areas which are plotted to highlight the differences in the degree of connection to the sea among the three estuaries (Santa Cruz Channel, Suape and Sirinhaem). The δ13C values of the SOM collected in estuaries (-25 ± 1.2 ‰) and at the coast (-17 ±1.6 ‰) are shown in the graph to give an indication of the δ13C values at the base of each environment.*

The $\delta^{15}N$ values of the fish among estuaries were not significantly different (Kruskal-Wallis chi-squared = 1.9052, p-value > 0.05) whereas $\delta^{13}C$ values varied depending on the estuaries (Kruskal-Wallis chi-squared = 14.458, df = 2, p-value < 0.05) and the Santa Cruz Channel was significantly different from Suape and Sirinhaem (Dunn's test, p-value < 0.05).Likewise, the *C. rhizophorae* $\delta^{15}N$ did not exhibit differences among estuaries (Kruskal-Wallis chi-squared = 5.6029, df = 2, p-value > 0.05) but their $\delta^{13}C$ were different (Kruskal-Wallis chi-squared = 15.382, df = 2, p-value > 0.05). The trophic positions of *E. argenteus* were similar among the estuaries: 2.78 ± 0.17 in Sirinhaem, 2.75 ± 0.14 in Suape and 2.66 ± 0.11 in the Santa Cruz Channel.

3.4.2. Sources contributions to *E. argenteus* diet

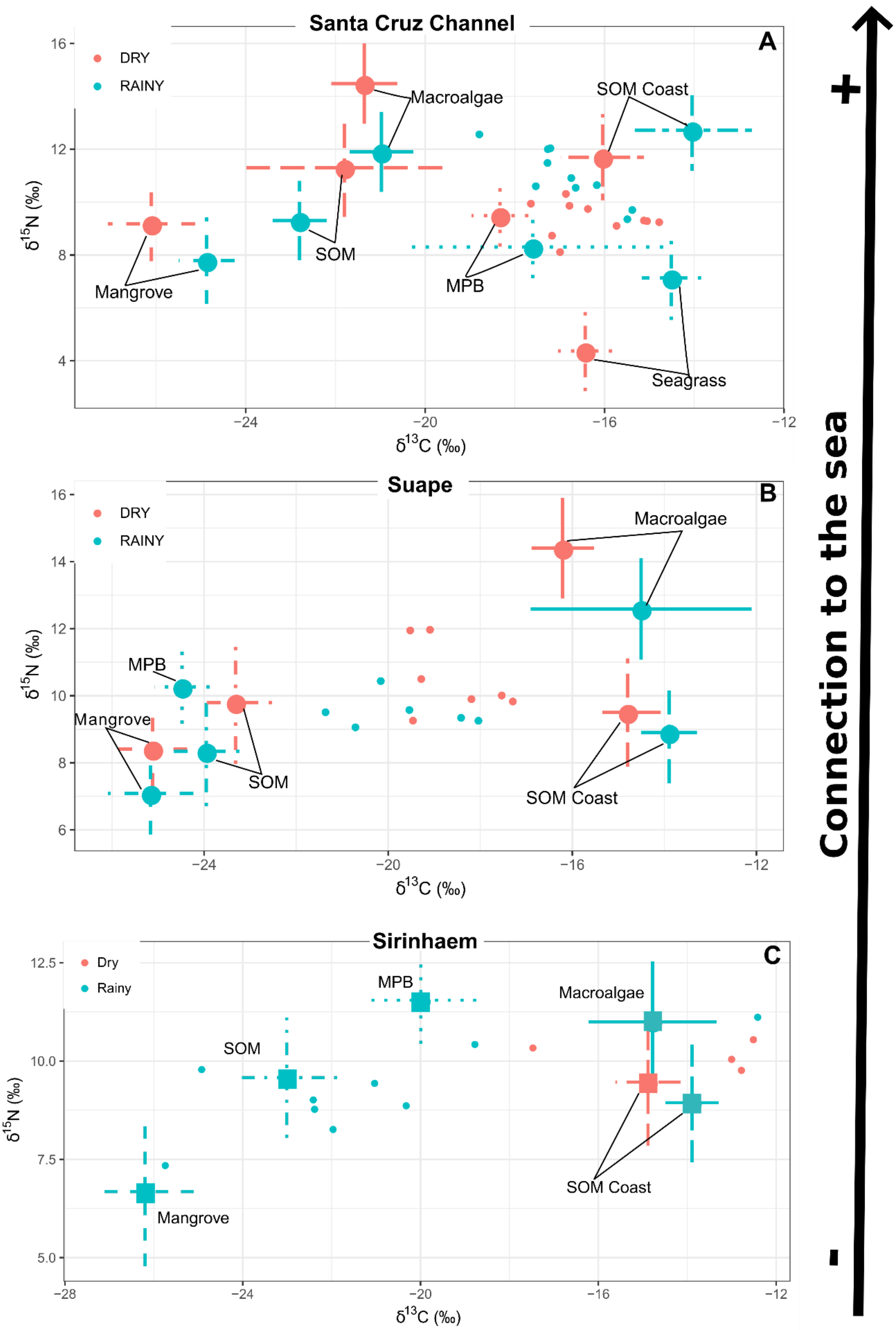

Figure 4: *Dual plots of the sources (represented by the squares and whiskers showing mean values ±SD of the* δ^{13}*C and* δ^{15}*N) and the consumers (represented by the points) after applying the trophic enrichment factor to the basal sources in the three estuaries (Santa Cruz Channel, Suape and Sirinhaem). SOM: Sedimentary organic matter; mangrove: mangrove leaves; MPB: microphytobenthos.*

In all three estuaries, several basal sources exhibited overlapping isotopic signatures (Figure 4). Indeed, the macroalgae ratios were close to those of the SOM in the Santa Cruz Channel while in Suape, depending on the season, both the microphytoplankton and mangrove leaves signatures were similar to the isotopic ratios of the estuarine SOM. Likewise, in Sirinhaem, the macroalgae overlapped the coastal SOM during the dry season. Most of the Sirinhaem fish ratios during the dry season were close to those of the macroalgae and the coastal SOM but were distant from the other sources. During the dry season, most of the Sirinhaem *E. argenteus* points displayed on the dual plot were outside the mixing polygon delimited by the basal sources. This pattern suggests that the selected sources were not appropriate to adequately describe the organic matter that supports the diet of *E. argenteus*. Thus, the MixSiar model was run only for the rainy season in this estuary.

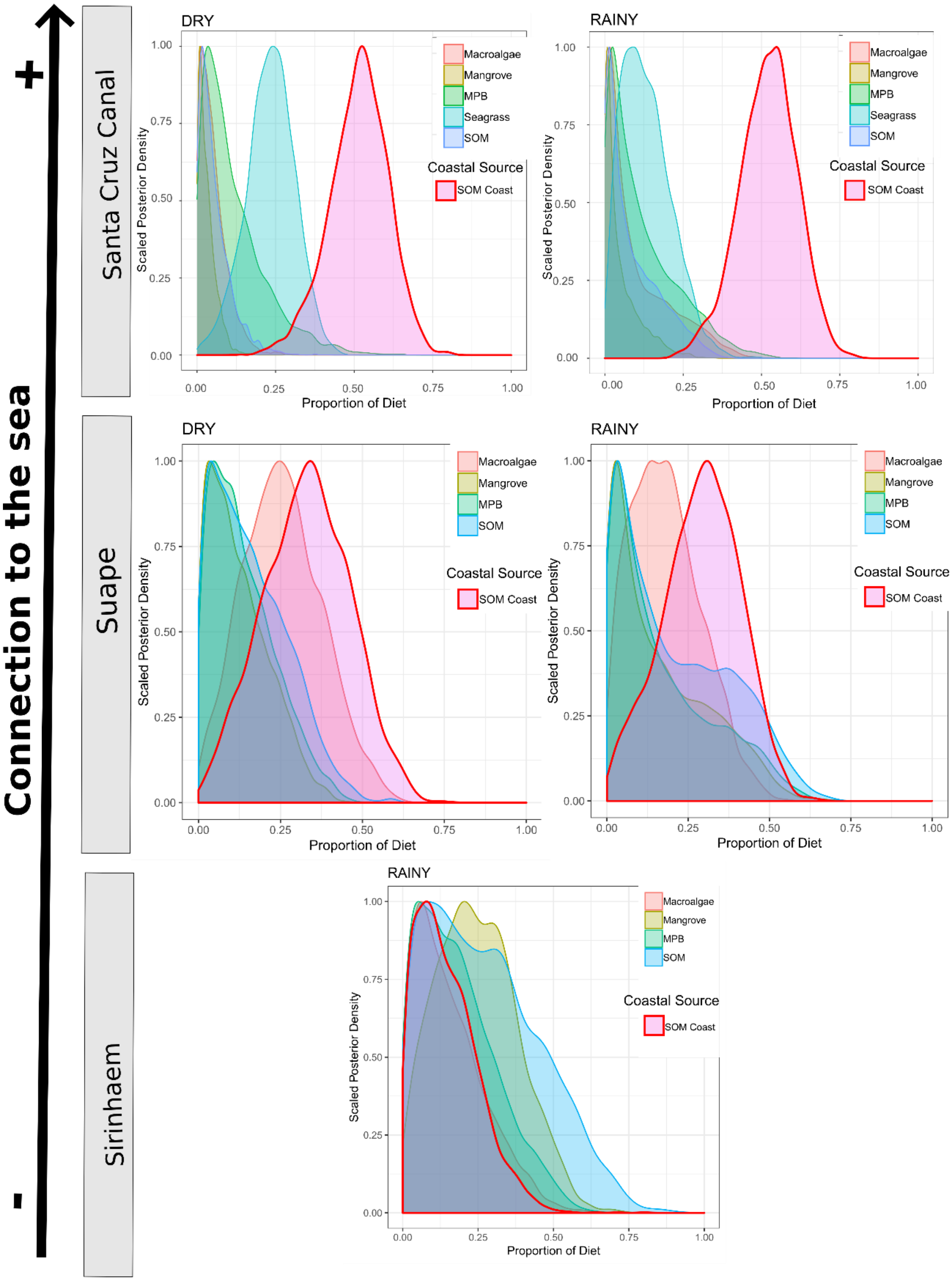

__Figure 5:__ Density plots representing seasonal proportion of the basal sources in the diet of E. argenteus at the Santa Cruz Channel, Suape and Sirinhaem. SOM: Sedimentary organic matter; mangrove: mangrove leaves; MPB: microphytobenthos.

Overall, the relative contributions of the basal sources were different between estuaries (Figure 5, Supplementary Table 6). The proportion of coastal source compared to estuarine sources increased with the sea connection (Figure 5, 6 and

Supplementary Table 6). The relative contribution of the coastal source is high in Santa Cruz Channel, lower in Suape and minimal in Sirinhaem (Figure 5, 6).

The contribution of the coastal SOM in the Santa Cruz Channel was the largest in both seasons, accounting for slightly more than 50% of the diet of *E. argenteus*. During the dry season, the seagrass had an important contribution during the dry season (23 %) whereas it contributed less during the rainy season (13%). The microphytobenthos contribution was also relevant in both dry and rainy season (around 12 %). The other sources contributions are low (less than 10%).

The coastal SOM had also the highest contribution during both seasons in Suape attaining the value of 32% during the dry season and 29% during the rainy season, but its relative participation was less substantial than in the Santa Cruz Channel. The other sources with relevant contribution in that area were the macroalgae during the dry season (25 %), and the SOM in the rainy season (20 %). The contributions of the other sources were similar, fluctuating between 10 and 20 %.

On the other hand, in Sirinhaem, the Coastal SOM exhibited the lowest contribution during the rainy season. No sources contribution was higher than 30 %. The two sources with relevant contribution were the estuarine SOM (27%) and the mangrove leaves (24%). The rest of the source contributions varied between 10 and 20%. Yet, the amplitude of the curves expressed a high variability among individuals.

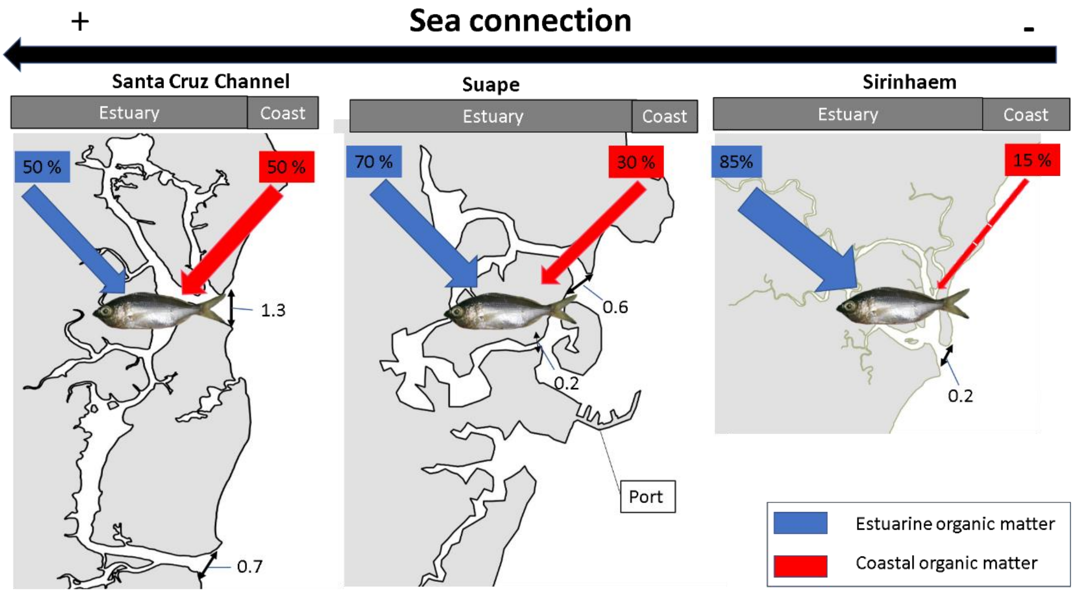

__Figure 6:__ Conceptual model summarizing the relative contribution of estuarine and coastal organic matter supporting the diet of E. argenteus in the three estuaries studied. In Sirinhaem, the relative contributions of organic matter correspond only to the rainy season

3.5. Discussion

3.5.1. Landscape properties and ecological connectivity

Our study showed that *E. argenteus* diet varied between the three estuaries studied and that the contribution of coastal resources increased with the connection to the sea. This connection with the sea, embodied by the morphological features (shape, mouth width) controlling key physico-chemical variables in this habitat (depth, salinity), is thus a suitable indicator of the connectivity between estuarine and coastal habitat. Thereby, ecological connectivity was promoted by the high sea connection in the Santa Cruz Channel. The fish of this estuary had high $\delta^{13}C$ values and the coastal SOM was therefore the source that contributed most to the diet.

The coastal SOM was also important in Suape but to a lesser extent since the geomorphological alterations, induced by the construction of the port, resulted in a restricted land-sea connection, isolating the two environments. The landfills caused by the dredging have led to a reduction in the river regime and hence to sedimentation, as well as to a reduction in water exchange and depth (Muniz et al., 2005). In Sirinhaem, estuarine sources were predominant, showing that ecological connectivity is reduced in this enclosed lagoon during the rainy season.

Tracing the origin of basal sources of organic matter was important to illustrate the complexity of trophic pathways between coasts and estuaries. In addition, *E. argenteus* was an appropriate indicator to provide information on the ecological connectivity between these two systems, since the adults (>13 cm) that inhabit preferentially coastal areas, enter the estuaries to feed (Ramos et al., 2016). Thus, no correlation was found between the $\delta^{13}C$ and the sizes of the fish in Sirinhaem or in the Santa Cruz Channel while the larger individuals in Suape presented depleted $\delta^{13}C$ values, characteristics of estuarine sources (Moens et al., 2002). This corroborates the fact that adults of this species use estuaries as a feeding area, as is the case for many species of the Gerreidae family (Ramos et al., 2012).

Though the movements performed by adult fish might exacerbated the connectivity patterns, local variations in marine intrusion are likely to explain the gradient of connectivity seen in the three estuaries. Organisms that have limited mobility can be influenced by this connectivity as estuarine food webs can be supported by coastal organic matter (Selleslagh et al., 2015). Methods using only stable isotopes cannot separate the effect of fish movement from the effect of marine intrusion. A tagging study with artificial loggings would have provided information on the frequency of foraging movements (Olds et al., 2017) and would have helped to distinguish these two components of ecological connectivity. Yet, the influence of ontogenetic migrations on the results seems to be reduced since fish sizes were not correlated to their respective $\delta^{13}C$ and $\delta^{15}N$ values when the three estuaries were grouped. Thus, differences in diet observed among estuaries cannot be attributed to differences in fish size. Furthermore, although fish sizes differed from one estuary to another, trophic positions were similar among estuaries. However, the low trophic position values found in this study (about 2.7) could be due to the tendency of trophic position underestimation with isotopic methods (Hussey et al., 2014) . Indeed, studies in the region of study did find a trophic position a little higher (i.e. 2.95) (Lira et al., 2018)) and diet studies in estuaries classified *E. argenteus* as a second degree consumer (Chi-Espínola et al., 2018; Vasconcellos et al., 2018).

Coastal and estuarine sources have been successfully distinguished using Bayesian mixing models. This analysis, despite the need for careful parameterisation, remain the best method for quantifying the relative contributions of sources in a diet (Parnell et al., 2010). They offered a useful tool for identifying differences in flows between habitats based on landscape properties. Nevertheless, the trophic enrichment factor, which strongly affects source contributions, should be specifically measured in the laboratory (Herzka, 2005). Yet, since the TEF chosen was consistent with those measured in the laboratory for zoobenthivorous fish (Kostecki et al., 2012), our results should appropriately reflect the relative contribution of coastal and estuarine sources.

Another weakness of mixing models is their lack of accuracy in an environment where several sources present overlapping isotope ratios, which usually occurs in estuaries (Phillips et al., 2014). Nonetheless, as our objective was to relate the degree of connection of the sea to the contribution from coastal sources, uncertainties in determining the participation of low contribution overlapping estuarine sources were not an issue. The fact that the coastal SOM was not available in Suape is not likely to bias the model, since Suape and Sirinhaem are part of the same coastal region with homogeneous habitats (Assis et al., 2015). Furthermore, the anthropogenic pressures locally occurring in Suape are not expected to significantly alter the isotope ratios of its coastal area, since Suape is a small estuary.

Contributions of basal sources were also influenced by seasonality. For example, marked seasonal variations in isotope ratios of *E. argenteus* occurred in Sirinhaem, the smallest estuary, due to the fact that in small tropical estuaries, where freshwater inputs are intermittent, seasonal nutrient inputs from terrestrial systems are favoured (Abrantes et al., 2015). Indeed, Sirinhaem location in the south, where rainfall is higher than in the north of the state (CPRH, 2003), allows for a large difference in freshwater supply between seasons. Differences in land use can also affect the seasonal intensity of inputs, but as Sirinhaem is the least anthropized estuary (Pelage et al., 2019) , environmental fluctuations seems to be the main driver of seasonal patterns in this region. Hence, during the dry season, we could not run the model for Sirinhaem because consumers were not within the trophic polygon of sources suggesting the existence of unknown sources probably derived from the sea due to their enriched $\delta^{13}C$ ratios. This implies that the low connection to the sea, inherent in the morphological characteristics of this enclosed lagoon, was not the factor that influenced the ecological connectivity during this season. Therein, high marine intrusion, fostered by high evaporation during the dry season and reduced river flow (Potter et al., 2010), may have caused a diet shift. During the rainy season, freshwater discharges reduce salinity, which is propitious for the development of polychaetes, while during the dry season, evaporation increases salinity, promoting the presence of marine benthic larvae (Chi-Espínola et al., 2018). This shift in marine invertebrates community could thus explain the higher $\delta^{13}C$ values observed in Sirinhaem during the dry season.

Despite the lack of significant differences in $\delta^{13}C$ values between seasons in Suape and in the Santa Cruz Channel, the seasonal shift observed in the models and with the ellipses may also be attributed to a difference in fish size within each estuary. Especially the differences in size between seasons could reflect the reproduction period of *E. argenteus* in the study area, which takes place at the beginning of the dry season and resulted the capture of smaller fish. Ontogenetic shifts were already observed in estuaries and hypersaline lagoons, where smaller specimens fed on zooplankton and larger individuals preyed on polychaetes (Silva et al., 2016; Chi-Espínola et al., 2018).

3.5.2. Implication for ecosystem functioning

Greater variability in fish isotope ratios was measured at Sirinhaem during the rainy season, translated by a large ellipse area and a high variability in sources contributions. The size range of Sirinhaem fish during that season was narrow so its variability cannot be attributed to ontogenetic shifts. This may be due to resource partitioning to avoid interspecific competition in this closed estuary thanks to the plasticity of *E. argenteus* diet. Diet plasticity is generally associated with the quantification of the extent of the trophic niche, i.e. the set of resources that can be exploited by a population to maintain acceptable growth parameters (Polis, 1984). The ability to occupy wider trophic niches, thus feeding on a wide range of different items, ensures greater adaptability (Pörtner et al., 2010). The opportunistic behaviour of the silver mojarra was already registered (Chi-Espínola et al., 2018) and could promote species adaptation in closed lagoons where coastal foraging opportunities are limited. In tropical estuaries, spatial differences in the diet of *E. argenteus* have already been reported (Guedes et al., 2015). This habit could mitigate the adverse effects caused by natural or anthropic isolation between estuarine and coastal environments. However, more detailed health studies need to be conducted to assess the actual status of the fish population since variation in food web carbon sources, may cause differences in growth and condition (Isnard et al., 2015). Hence, variability in ecological connectivity and thus in diet could lead to variation in growth parameters.

Estuaries are suffering from various anthropic geomorphological modifications (Wetzel et al., 2012Du et al., 2016; Lechêne et al., 2018) which could alter the flows with adjacent coastal areas. Variations in ecological connectivity are likely to affect the functioning of ecosystems as they influence trophic pathways and energy flows between adjacent habitats. Besides, this study shows that ecological connectivity extent evolves with sea connection degree which controls essential environmental variables such as salinity and depth. These variables are key structuring factors for the distribution and abundance of species. Indeed, *E. argenteus* is a saltwater species, and in general Gerreidae abundance is usually positively correlated with salinity (Corrêa and Vianna, 2016). In the Santa Cruz Channel, where sea connection is strong, *E. argenteus* was the second most abundant species, while in the other two estuaries, Suape and Sirinhaem, the silver mojarra was not among the ten most abundant species (Merigot et al., 2017).

Indeed, the availability of the prey is controlled by environmental factors (Arula et al., 2012; Martino and Houde, 2010) and a narrow sea connection degree does influence estuarine resources access for saltwater species and coastal resources for estuarine resident. Reduced access to food resources resulting from habitat fragmentation is likely to exacerbate competition patterns (Hasegawa, 2017). In the case of a change in the sea connection such as during the construction of the port, coastal preys were less available for *E. argenteus* juveniles but the diet plasticity of this species may have allowed it to switch to a more estuarine diet, which may have increased competition with estuarine resident species.

In addition, marine migratory species whose juveniles use estuaries as nurseries and whose adults extensively use estuaries as feeding or reproductive grounds (e.g. Mugilidae, Gerreidae) represent an important yield for small-scale estuarine and coastal fisheries in tropical areas (Salas, 2011). Modification in ecological connectivity through alterations in estuarine morphology is susceptible to have adverse effects on these species and could have socioeconomical repercussion since they constitute essential source of proteins for local populations. In addition, they represent key prey for piscivorous fish, therefore a change in their abundance due to habitat degradation would have drastic repercussions on the ecosystem.

3.6. Conclusion

Differences in the degree of connection to the sea in estuaries influenced the origin of the organic matter supporting the diet of E. argenteus, chosen as an indicator of ecological connectivity with adjacent coastal habitats. Consequently, the geomorphological disturbances caused by the increasing human modification of estuaries are likely to seriously affect the functioning of these habitats, in particular by altering the energy flows that support their food networks. The influence of seascape properties on biological processes is complex and assessing their effects on ecosystem services such as fisheries yields is a difficult but necessary task in a context of global change and dwindling resources. Preliminary multidisciplinary studies should be carried out before implementing any changes to the seascape that alter the connectivity between habitats, in particular in the case of habitats of significant importance for migratory species. In addition, understanding the extent to which the restoration of geometric properties through eco-engineering allows for ecosystem resilience in terms of connectivity would be of interest.

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SUPPLEMENTARY MATERIAL:

Environment	**Gear**	**net height(m)**	**net length (m)**	**Mesh (mm)**
Estuary	Block net	1.9	80	70
	Beach seine	5.0	180	25
	Seine net	8.0	75	10
Coast	Gill net	1.5	200	50
	Trawler	6.1	10	25
	Bottom trawl	10.0	28	25

Supplementary Table 1: Characteristics of the fishing gears used to capture the fish in the different environment.

Estuary	**DRY**				**RAINY**			
	n	TL (cm)	δ^{13}C (‰)	δ^{15}N (‰)	n	TL (cm)	δ^{13}C (‰)	δ^{15}N (‰)
Santa Cruz Channel	10	8.2-9.6	-16.3±1.0	9.3±0.6	10	11.9-14.6	-16.9±1.0	11.0±1.0
Suape	7	3.6-5.2	-18.6±0.9	10.6±1.1	6	5.2-14.1	-19.8±1.3	9.7±0.5
Sirinhaem	4	7.4-16.6	-14.0±2.4	10.3±0.3	9	7.7-9.5	-21.2±3.9	9.3±1.1
Coast	10	9.4-15.0	-16.9±2.27	11.2±0.8	-			

Supplementary Table 2: Number of samples (n), Total length (TL), Isotopic means (± S.D.) of carbon (δ^{13}C), nitrogen (δ^{15}N) of the fish collected in the estuarine and coastal areas during the dry and the rainy season.

Source	**Environment**	**Area**	**DRY**			**RAINY**		
			n	**δ^{13}C**	**δ^{15}N**	**n**	**δ^{13}C**	**δ^{15}N**
SOM	Estuary	SCC	6	-23.8±1.4	4.5±1.0	6	-24.0±1.0	4.2±0.7
		SUAPE	6	-25.3±0.5	4.2±0.9	6	-26.0±0.4	2.7±0.6
		SIR	6	-25.9±0.7	4.7±0.5	6	-25.0±1.0	4.0±0.3
	Coast	SCC	6	-21.2±1.2	5.5±1.2	6	-20.6±1.1	5.8±1.1
		Sir	6	-16.8±0.4	3.9±0.6	6	15.9±0.1	3.3±0.2
Algae (Ulva sp)	Estuary	SCC	3	-23.4±0.4	8.9±0.2	3	-23.0±0.4	6.3±0.2
Algae Sargassum sp.	Estuary	SUAPE	3	-18.2±0.3	8.9±0.1	3	-16.5±2.3	7.0±0.2
		SIR		-	-	3	-16.8±1.3	5.4±0.4
Microphyto benthos	Estuary	SCC	3	-20.3±0.2	3.9±0.4	2	-19.6±2.8	2.7±0.4
		SUAPE	1	-26.4	4.7	-		
		SIR	4	-24.8±0.7	4.2±0.5	3	-22.0±1.1	5.9±0.2
Seagrass	Estuary	SCC	3	-27.6±0.6	2.4±0.5	3	-28.0±0.3	1.4±0.4
Mangrove	Estuary	SCC	3	-27.8±0.7	3.0±0.9	3	-27.4±0.2	1.8±1.2

		SUAPE	3	-27.1±0.5	1.5±0.5	3	-27.1±0.7	1.5±0.5
		SIR	3	-28.5±0.1	1.8±1.9	3	-28.5±0.5	2.1±0.8
C. rhizophorae	Estuary	SCC	4	-23.2±0.4	7.7±0.4	3	-22.5±0.1	8.2±0.1
		SUAPE	3	-19.3±0.8	7.1±0.3	3	-19.6±1.7	8.0±0.2
		SIR	6	-16.1±0.4	6.8±0.4	6	-20.4±0.2	8.0±0.2

Supplementary Table 3: Number of samples (n) and isotopic means (± S.D.) of carbon (δ13C), nitrogen (δ15N) of basal sources and the oyster *C .rhizophorae* during dry and rainy season in the Santa Cruz Channel (SCC),Suape and Sirinhaem (SIR).

Estuary	Variable	P value	Rho	S
ALL	$\delta^{15}N$	NS	-	11975
	$\delta^{13}C$	NS	-	13919
Santa Cruz	$\delta^{15}N$	0.00236	0.6402471	478.47
Channel	$\delta^{13}C$	NS		1795.9
Suape	$\delta^{15}N$	NS	-	428.35
	$\delta^{13}C$	0.009764	-0.6850933	613.37
Sirinhaém	$\delta^{15}N$	NS	-	227.62
	$\delta^{13}C$	NS	-	232.64

Supplementary Table 4: Results of the Spearman ranking tests between the total length off the fish and the the $\delta^{13}C$ / $\delta^{15}N$ of fish in all the estuaries together as well as in each estuary (Santa Cruz Channel, Sirinhaem and Suape). NS: Not Significant (p value >0.005).

Estuary	Variable	p value	W
ALL	$\delta^{15}N$ (‰)	NS	250
Santa Cruz Channel	$\delta^{15}N$ (‰)	0.0007253	8
	$\delta^{13}C$ (‰)	NS	66
Suape	$\delta^{15}N$ (‰)	NS	35
	$\delta^{13}C$ (‰)	NS	33
Sirinhaem	$\delta^{15}N$ (‰)	NS	28
	$\delta^{13}C$ (‰)	0.03357	32

Supplementary Table 5: Results of the Mann-Whitney tests verifying seasonal differences in the $\delta^{13}C$ / $\delta^{15}N$ of fish in each estuary (Santa Cruz Channel, Sirinhaem and Suape) . NS: Not Significant (p value >0.005).

Source	**Estuary**	**Contribution**	
		DRY	**RAINY**
Algae	Santa Cruz Channel	0.051	0.096
	Suape	0.256	0.182
	Sirinhaem	-	0.156
Mangrove	Santa Cruz Channel	0.031	0.050
	Suape	0.123	0.161

	Sirinhaem	-	0.246
Microphytoplancton	Santa Cruz Channel	0.115	0.116
	Suape	0.136	0.165
	Sirinhaem	-	0.181
Seagrass	Santa Cruz Channel	0.233	0.133
SOM	Santa Cruz Channel	0.055	0.084
	Suape	0.157	0.203
	Sirinhaem	-	0.270
SOM Coast	Santa Cruz Channel	0.514	0.521
	Suape	0.329	0.290
	Sirinhaem	-	0.146

Supplementary Table 6: Mean of the relative contributions of the organic matter sources in each of the estuaries during the rainy and the dry season.

Parte 2: Impactos das frotas coexistentes que exploram esses ambientes conectados.

Composition and spatio-temporal dynamics of artisanal fisheries catches in mangroves using passive intertidal devices.

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4. Mangrove fisheries targeting migrant species.

4.1. Abstract

Mangrove serves as a nursery for many marine migrant species that depend on these habitats for their growth. Yet, data on fishing in mangrove systems remain scarce, even though they are the scene of a unique type of subsistence fishing reported in Asia, Africa, and South America, involving passive gears that use the tide to retain fish. In this study, we investigated the particularities of this intertidal fishing in mangroves and their consequences in terms of management. The assessment of block net fisheries impact is complex because it must consider the changing dynamics of intertidal environments. In this purpose, we described the spatial and temporal variability of the samples in terms of fish family composition. We also studied the impact of block net fishing on three species with different growth parameters. The spatiotemporal variability of catches in terms of number of individuals per sample, sizes and ontogeny was described and information was provided on the selectivity of this gear for each of the three species. A strong spatiotemporal variability between samples was observed both in the composition of fish families and in the size and abundance of the selected species. The block net harvested 96 species but 65% of their catches consisted of Mugilidae and Gerreidae. Overall, the block net catches were dominated by immature individuals even if this gear harvested an important range of sizes and ontogeny classes. The S-shape selectivity unusual for passive gears and the targeting of productive species and size compartments allowed the capture of a broad range of sizes and species, according to the spatiotemporal productivity of the fishing ground. This fishing pattern is characterized by a high yield and an adaptability to abundance changes in line with balanced harvest objectives for food security. These fisheries could represent a sustainable subsistence fishery in the case of moderate effort. However, the yields from block net and coastal fisheries are interdependent, as they exploit a common resource, making multi-ecosystem fisheries management necessary.

4.2. Introduction

Mangroves are widely recognised for their role in improving the productivity of coastal fisheries and thus ensuring essential livelihoods for fishing communities (Carrasquilla-Henao and Juanes, 2017; Manson et al., 2005; Reis-Filho et al., 2019. In particular, mangroves contribute to the maintenance of fisheries by supporting nearshore food webs since they are highly productive systems whose primary production is important for food webs within this biome, but also for those in adjacent or connected habitats (Abrantes et al., 2015; Kelleway et al., 2018). Moreover, this habitat enhances coastal fisheries yield by serving as a nursery for commercial species migrating from the mangroves to the open sea (Hutchison et al., 2014; Seary et al., 2020). As these migratory species need to use several components of the coastal continuum throughout their life cycle, they rely on the complementarity and proper functioning of the different coastal ecosystems (mangroves, coral reefs, seagrass beds) (Berkström et al., 2013; Olds et al., 2018, 2016). For example, the juveniles of many marine migratory species depend on mangrove areas which afford adequate conditions for their growth (Archambault et al., 2018). These species use mangroves as a shelter because their complex structure can protect them from predators (Nanjo et al., 2014). Consequently, any significant impact on mangrove forests will have significant consequences in fish biodiversity and abundance (Nip and Wong, 2010; Estoque et al., 2018).

In this regard, mangroves are essential for tropical small-scale fisheries (SSFs) which are characterized by a multidimensional fishing pattern (fleets, gears, habitats, species) resulting in the harvesting of migratory species in each of the connected environments they exploit (Pelage et al. a Submitted). Yet, despite numerous studies that corroborate the key role of mangroves for fish fauna, data on fishing in mangroves remain scarce (Reis-Filho et al., 2019; Seary et al., 2020). Indeed, most of mangrove fisheries are found in tropical countries with poor management (Mozumder et al., 2018; Reis-Filho et al., 2019; Seary et al., 2020), hampering the quantification of fishing impact on mangrove systems. However, mangroves have been exploited for centuries and are the scene of a traditional type of fishing involving passive gear that uses the tide to retain fish. Such intertidal devices , commonly known as block nets, are used in mangroves all over the world and have been documented particularly in

Asia, Africa and South America (Rasolofo, 1997; Shahraki et al., 2016; Merigot et al., 2017). Deploying block nets directly along the front edge of the mangroves is the most successful method to study fish entering the mangroves (Sheaves et al., 2016). Nevertheless, this type of fishery is poorly described and its selectivity (probability that a given length class of a given fish species will be caught, assuming that it is available to the gear) has never been reported to our knowledge.

Hence, selectivity curves differ according to the different types of gear and their configuration (Huse et al., 2000; Stepputtis et al., 2016). For example, passive gear, such as gillnets and trammel nets, generally have bell-shaped curves and are characterized by low retention probabilities in small and large length classes, resulting in the targeting of medium length classes (Irwin et al., 2008; Acarlı et al., 2013; Akongyuure et al., 2017; Saber et al., 2020). However, the passive intertidal devices used in mangroves are designed to retain all fish that were in the area at high tide for collection at low tide. The only fish that can escape are those that can pass through the meshes of the net. This feature is not documented although the selectivity shape of a gear is strongly linked to the kind of management that can be applied (Stepputtis et al., 2016).

In Brazil, SSFs are usually inadequately managed so that several stocks are overexploited and not systematically assessed (Gasalla et al., 2017). These SSFs target a wide range of species resulting in technical interactions (where several fish are caught together in the same net or fishing operation) which difficult the evaluation of fishing impact. Moreover, tropical Brazilian fisheries are composed of co-existing fleets, which have different fishing grounds (Hazin et al., 1998;Frédou et al., 2009) but harvest migrant species across connected environments implying that the fishing pressure from one fleet can impact on the yield of another. In north-eastern Brazil, mangrove fisheries are known to be complex socio-ecological systems and fishermen have reported exploiting several fishery resources (including fish species, crabs, molluscs, and shrimps) (Santos et al., 2017). The block net (locally named “Camboa”) is used throughout the north and northeast of Brazil as the main fishing gear harvesting fish in mangrove areas. Indeed, it targets the most commercially important fish species that are found in the estuaries which are represented by the families Centropomidae, Gerreidae, Mugilidae, Lutjanidae, Carangidae and Sciaenidae

(Giglio and Freitas, 2013). All these species are estuarine dependants and all rely on the nursery function of the estuaries.

Furthermore, block nets are socially important for the region and have been used for centuries by fishermen using ancestral techniques and knowledge handed down from generation to generation (Araújo and Pereira, 2015). Moreover even if this fishing method is becoming scarce in the region, some fishing communities still rely on this activity for their income and livelihoods (Mendonça et al., 2011).Though, the assessment of block net fisheries impact is complex because it must consider the changing dynamics of intertidal environments that make mangrove catches highly variable (Seary et al., 2020). In addition, the impact of mangrove fisheries on migrant species will depend on the life cycle parameters of the species. Consequently, the intricate spatial segregation of their different ontogenetic stages constitutes an additional difficulty in the management of these fisheries. Besides, since the impact on the life cycle of these species depend on the characteristics of the gear employed (Burns et al., 2019), there is a necessity to have information on which sizes are targeted. However, studies on selectivity are rare in north-eastern Brazil and, to our knowledge, none are available for teleost fish.

In this study, we investigated the particularities of intertidal fishing in mangroves and their consequences in terms of management. In this purpose, we described the spatial and temporal variability of the samples in terms of fish family composition. We also studied the impact of block net fishing on three model species with different growth parameters in order to study the spatiotemporal variability of catches in terms of number of individuals per sample, sizes and ontogeny and to provide information on the selectivity of this gear for each species.

4.3. Material and methods

4.3.1. Study area

The sampling was carried out in four estuaries located along the coast of Pernambuco State (Figure 1, Table 1): The Santa Cruz Channel, Suape, Sirinhaem and Rio Formoso. The Santa Cruz Canal constitutes the largest estuarine complex in

Pernambuco. This U-shaped canal has an extension of 22 km and two accesses to the Atlantic Ocean (Silva, 2012). It is strongly influenced by landscape modifications, industrial waste, shrimp farming and suffers from strong fishing pressure (Gondim et al., 2015; Lira et al., 2010).The Suape estuary, situated in the south coast, is a coastal lagoon with a flattened geomorphological shape (Santos and Costa, 1974). Geomorphological and hydrodynamic conditions were modified by the construction of a port in 1979 (Muniz et al., 2005) with consequences for biodiversity and fishing productivity (Paiva and Araújo, 2010). Sirinhaem and Rio Formoso are located in the southernmost area, and although situated within an environmental protected area (APA Guadalupe), they are also impacted by agricultural pollution (mainly sugar cane), shrimp farming and fishing (Dos Santos et al., 2020)

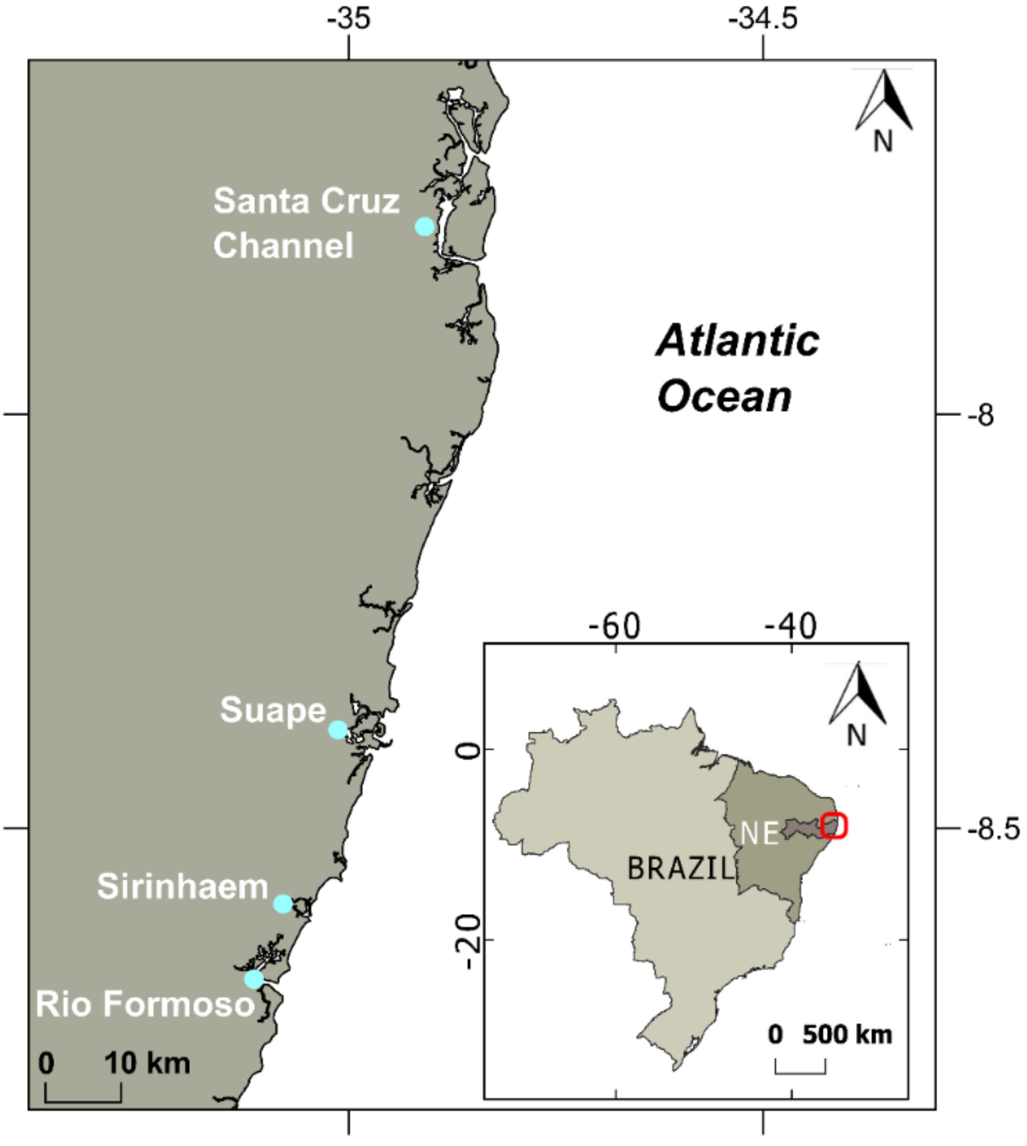


Figure 1: Map of the study area showing the four sampled estuaries in Pernambuco state, situated in the Northeast region (NE) of Brazil.

Table I: Characteristics of the four estuaries under study (adapted from Gonzalez et al. 2019).

Characteristics	Estuary			
	Santa Cruz	Suape	Sirinhaem	Rio Formoso
Type	Ria	Coastal Lagoon	Coastal Plain	Coastal Plain
Estuary (km²) †	73.5	29.7	18.7	27.7
Vegetated area (km²) †	48.0	23.1	17.0	19.9
Watershed area (km²) †	25.5	6.6	1.7	7.8
Depth (m)	3	3.1	2.6	4.7
Mouth width (km) †	(0.5 - 1.3)	0.6	0.2	1.07
Numb. of mouths	2	1	1	1
Reference	Flores Montes et al. 1998; Medeiros et al. 2001; Silva et al. 2011	Borges 2011; Silva et al. 2011	Silva 2009b; Silva et al. 2011	Honorato da Silva et al. 2004; Silva et al. 2011

† Data obtained with the software ArcGis 10.1 based on satellite images from LANDSAT 7. Type, geomorphological classification of estuaries according to (Pritchard 1952). Estuary, respective area of the estuary; the area was considered as the sum of watershed and vegetated areas. Depth mean depth during low tide. Mouth width, width of the connection to the sea; mean width was considered when the estuary exhibits more than one connection to the sea. Numb. of mouths, number of connections to the sea. Reference, bibliographic references used for study area description.

4.3.2. Data collection

A total of 24 samples were collected in the four estuaries: 7 in the Santa Cruz Channel, 5 in Suape, 8 in Sirinhaem and 4 in Rio Formoso from October 2012 to September 2017 (see dates of each sampling in the Supplementary Table 1). The sampling was performed during the rainy (from March to August) and the dry season (from September to February) identified by the amount of rain during the year (APAC, 2020). Fish were captured using block nets with the same dimensions (70–90 m long, 2.5 m high mesh size of 50 mm) in the four estuaries. The gears were set along the mangrove forest and anchored to the bottom at low tide by local fishermen in their usual fishing grounds. The net was attached to stakes and pulled taut at slack high-water so that it was above the water, enclosing the mangrove area. Blocking was initiated at the end of the high tide and continued throughout the entire ebb tide cycle (approximately 6 h). Thereupon, fish were frozen and transported to the laboratory, where they were identified, measured to the nearest millimetre and weighed to the nearest gram.

4.3.3. Selected species

We chose to understand the variability in abundance of the six migratory families that are mainly targeted by the block net (Gerreidae, Mugilidae, Centropomidae, Lutjanidae, Carangidae and Sciaenidae) while the other families were grouped into a category called "others”. Three species have been selected from the three most captured families among the six mentioned above. Besides their importance in block net catches, *Eucinostomus argenteus*, *Mugil curema* and *Centropomus undecimalis* were chosen because they have distinct growth parameters (Supplementary table 2). Moreover, although all three species use estuaries as nursery areas and their adults are more abundant in coastal areas (Gonzalez et al., 2019; Menezes et al., 2015; Ramos et al., 2014), they exhibit a difference in their life cycles. Indeed, adults of *C. undecimalis* are not present in estuaries whereas adults of *E. argenteus* and *M. curema* may use estuaries as reproductive or feeding grounds (Figure 2) (Menezes et al., 2015; Ramos et al., 2014).

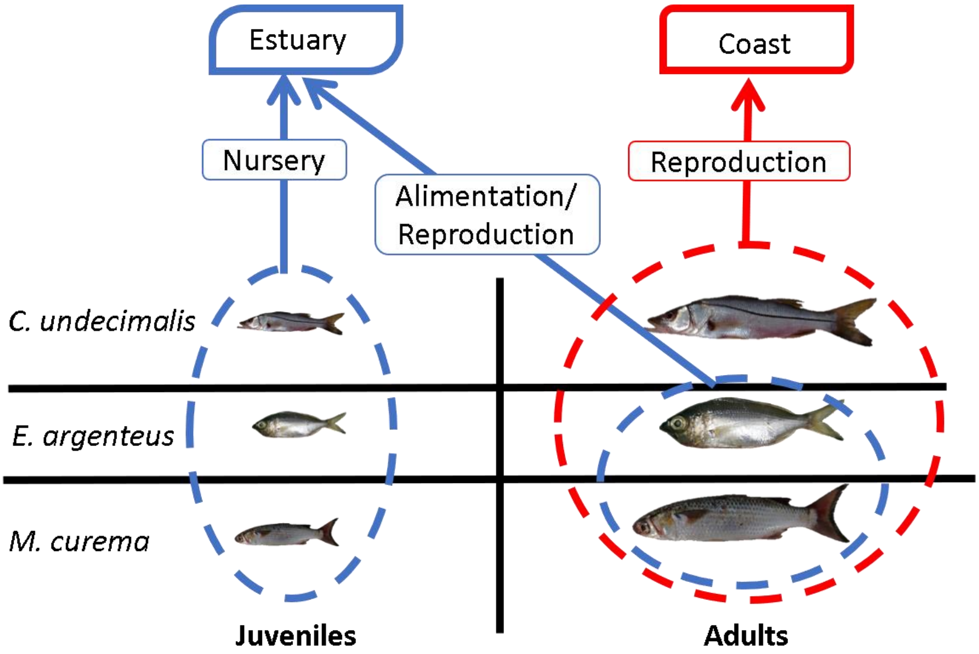


Figure 2: Habitat use in estuarine and coastal environment of the three selected migrant species according to their ontogeny.

4.3.4. Data analysis

For each species, spatial and temporal differences in size and number of individuals caught per sample were tested using Kruskal Wallis tests and Dunn's post hoc tests. Fish were separated into three ontogenetic stages (juveniles, subadults and adults) according to a method based on two criteria (Ramos et al. 2012): (1) the inflection point of the length-weight curve by species, used to distinguish juveniles from sub-adults, and (2) the size at first maturity (L_{50}) according to the literature found for regions with a climate similar to the study area (Supplementary table 2).The graphical criterion for separating juveniles from sub-adults assumes that individuals whose metabolism is dedicated to length gain can be distinguished from specimens that are beginning to gain weight (Ramos et al., 2012). This categorization provides a semi-quantitative understanding of catch structure in terms of ontogeny by species. This enabled us to draw up graphs describing the relative catch in terms of ontogeny for each species and to see how it varied spatially and temporally.

The selectivity of passive nets is generally described by bell-shaped curves. However, intertidal fishing in mangroves (Figure 3) is characterised by S-shaped selectivity curves which are generally associated with active nets such as trawls (Huse et al., 2000). Indeed, the action of water withdrawal combined with the encircling of the fish leads to the escape of small fish that can pass through the meshes and the retention of larger specimens.

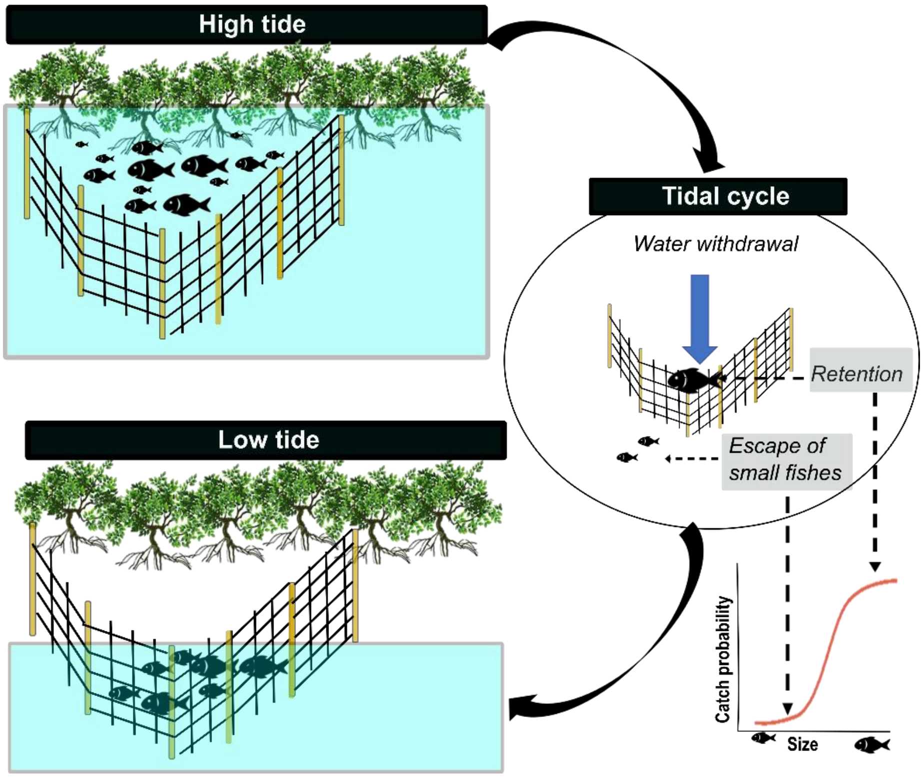


Figure 3: Conceptual model describing the functioning of the block net and its selectivity.

The selectivity of the block net was determined with linearized catch curve for length data that were converted to age. This length-based approach is recommended under data-poor situations (Cope and Punt, 2009). Age conversion was performed with the von Bertalanffy growth function, rearranged to express age as a function of length:

$Age=t_0-1/K*\log(1-length/ L\infty)$

where t_0 is the theoretical age at length 0, K is the growth coefficient and $L\infty$ is the asymptotic length.

The estimation of body growth parameters t_0, K and $L\infty$ were realized with bootstrapped and single-fit ELEFAN-based (Electronic LEngth Frequency ANalysis) curve fitting methods which is widely used to fit a growth curve to length-frequency distribution (LFD) data (Schwamborn et al., 2019). For modelling the selectivity at a given age (sel (a)) we used a simple logistic regression. This equation produces a smooth, S-shaped curve that is often used in gear-selection studies to mimic the phenomenon that young (small) fish are caught less frequently than old (large) fish:

$sel(a)=(1+\exp((beta0-a)/ beta1))-1$

where *beta0* is the parameter which controls the steepness of the curve and beta1 is the parameter which controls the age at 50% selectivity.

A linear model was used to predict the values of the catch for each age and these predicted values of catch were compared with the observed catch to adjust the selectivity curve. The calculation of a confidence interval was realized by exploring the uncertainties on the estimations of *beta0* and *beta1*. Because there are two parameters, this is a distribution of the joint probability of *beta0* and *beta1*. The posterior distribution was obtained using Bayesian inference by applying the Markov chain Monte Carlo (MCMC) with the runjags package (Denwood and Plummer, 2019). To determine whether a stationary distribution was achieved in the MCMC procedure, graphs were generated and diagnostic tests were performed using the Coda package (Plummer et al., 2019).The age where 50 % of the fish were retained by the gear (a50) was calculated for each species (a50=beta0/beta1). All these analysis were realized with the software R (R Core Team, 2019).

4.4. Results

4.4.1. Abundance and composition spatiotemporal variability

A total of 10,196 individuals were captured, distributed in 96 species 56 genera and 32 families (Supplementary table 3). The six families targeted by the block net (Gerreidae, Mugilidae, Centropomidae, Lutjanidae, Carangidae and Sciaenidae) represented more than 80 % of the global catch. Specifically, the catch was composed by 40% of Gerreidae, 25% of Mugilidae, 7% of Centropomidae, 4% of L*utjanidae*, 3% of Carangidae, 2% of Sciaenidae and 19% of other families. Gerreidae represented an important part of the catches in all estuaries but Mugilidae predominated in the Santa Cruz Channel catches (representing 45% of the catches in this estuary).

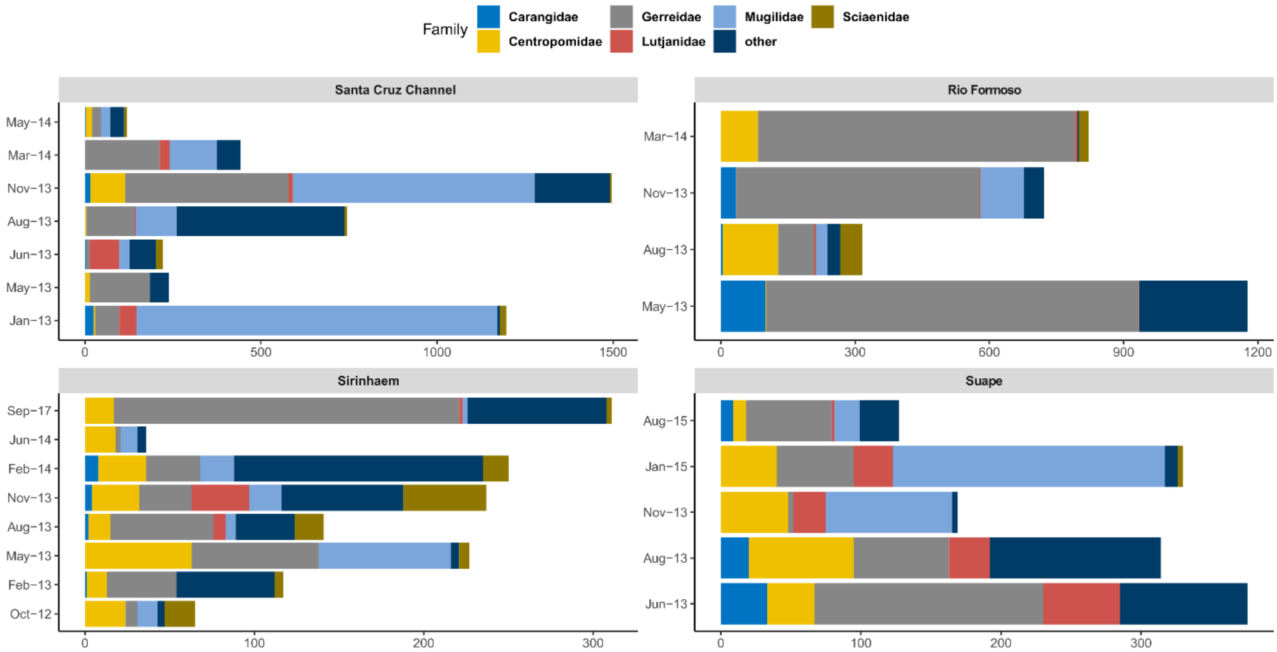


Figure 4: Fish family composition and samples abundance of block net catches by estuary (Suape, Sirinhaem, Santa Cruz Channel and Rio Formoso).

Fish family composition and samples abundance differed greatly from one estuary to another and even within each estuary (Figure 4). Indeed, the number of individuals per sample varied from 120 to 1497 in The Santa Cruz Channel, from 37 to 312 in Sirinhaem, from 317 to 1178 in Rio Formoso and from 128 to 377 in Suape. Regardless the estuary, Gerreidae were important in almost all the collected samples. In addition, abundance peaks of Mugilidae were observed in all estuaries, but they were less pronounced in Rio Formoso and were not observed at the same time in all estuaries. For example, Mugilidae represented an important part of the catch in November 2013 for all estuaries except Sirinhaem. In this estuary, a high number of Mugilidae were captured in May 2013 while Mugilidae were almost absent at that date in Rio Formoso and the Santa Cruz Channel catches. In addition, Sciaenidae were relatively important in Sirinhaem catches, mainly represented by the species *Bairdiella ronchus* (which made up 75% of the Sciaenidae caught in the region). Centropomidae also represented an important part of the catches for all samples collected in Sirinhaem.

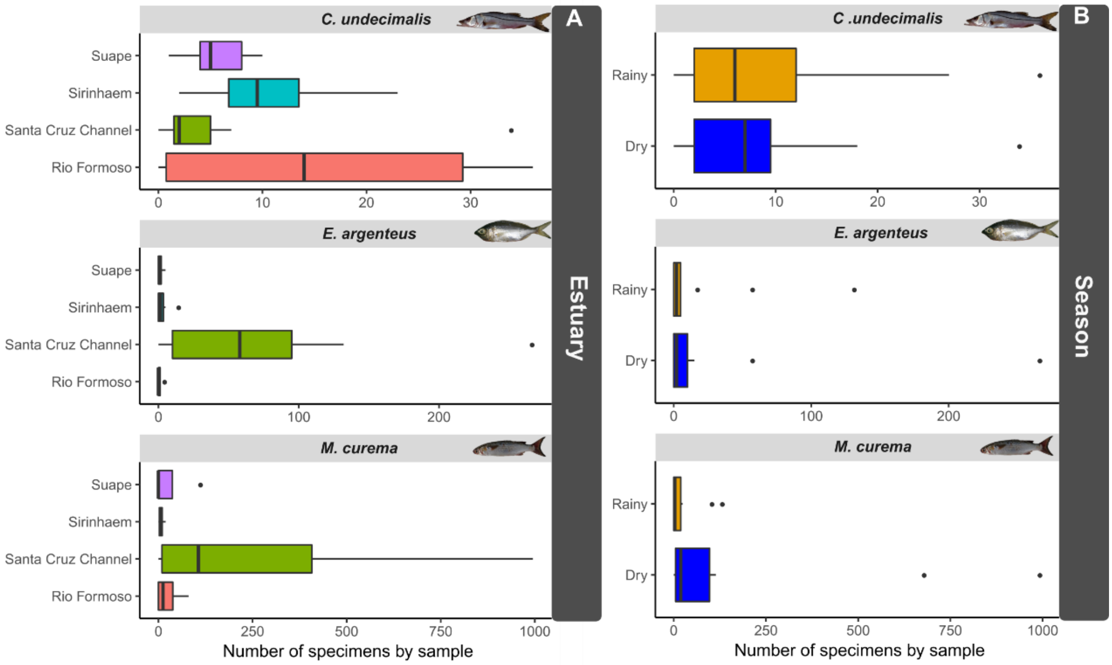


Figure 5: Boxplots of the number of specimens per sample for each of the three selected species showed A) by estuary (Suape, Sirinhaem, Santa Cruz Channel and Rio Formoso) and B) by season (dry and rainy).

A total of 3,048 individuals of the three selected species were captured being 228 of *C. undecimalis*, 574 of *E. argenteus* and 2246 of *M. curema*. The number of specimens per sample fluctuated greatly in each estuary and during each season particularly for *M. curema* (from 0 to 995) and *E. argenteus* (from 0 to 237). The medians of the number of specimens per sample were not significantly different among estuaries for any of the three species (p value > 0.05). Nevertheless, high abundance peaks of *M. curema* (2 samples recording 681 and 995 individuals) and E. argenteus (2 samples registering 132 and 267 individuals) were observed in the Santa Cruz canal, whereas both species were not abundant in any of the samples from the other three estuaries (Figure 5A). Conversely, individuals of *C. undecimalis* were more evenly distributed in the samples of the four estuaries and no abundance peaks superior to 36 individuals per sample were registered.

The medians of the number of specimens per sample did not significantly varied between seasons for any of the three species (p value > 0.05, Supplementary table 3). The abundance per sample was evenly distributed between both seasons for *C. undecimalis* (Figure 5B). For *E. argenteus*, the highest peak (267 individuals) observed in the Santa Cruz Channel can be visible in the dry season. The two samples with the highest number of *M. curema* individuals were registered during the dry season (681 and 995 individuals) while no abundance peak was noticed during the rainy season.

4.4.2. Size and ontogenetic variability

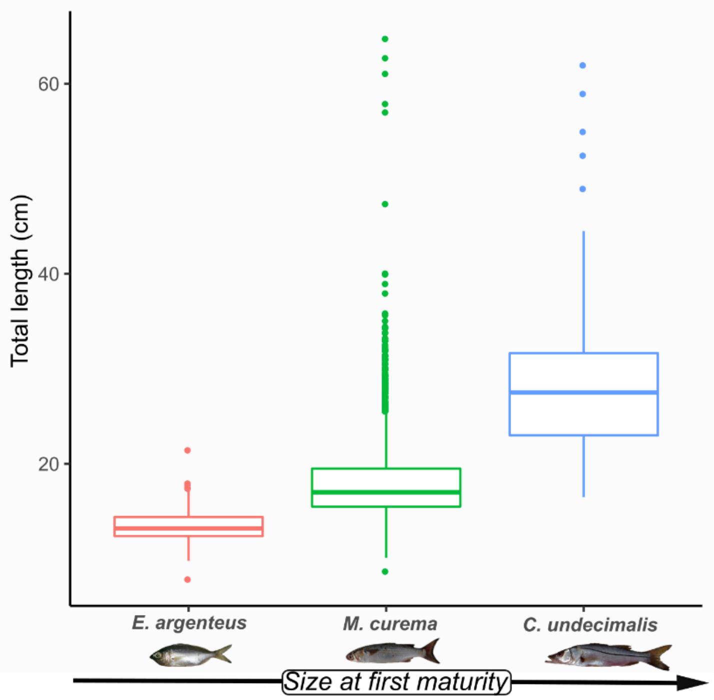


Figure 6: Boxplots of the Total length for each of the three selected species (E. argenteus, M. cure*ma and* C. undecimalis*).*

Overall, the three species were harvested with significantly different sizes (p value <0.05). The captured sizes increased with the size at first maturity of the species (Figure 6). Captured individuals ranged in size from 7.9 to 21.5 cm for *E. argenteus*, 8.7 to 64.8 cm for *M. curema and* 16.5 cm to 62 cm for *C. undecimalis*.

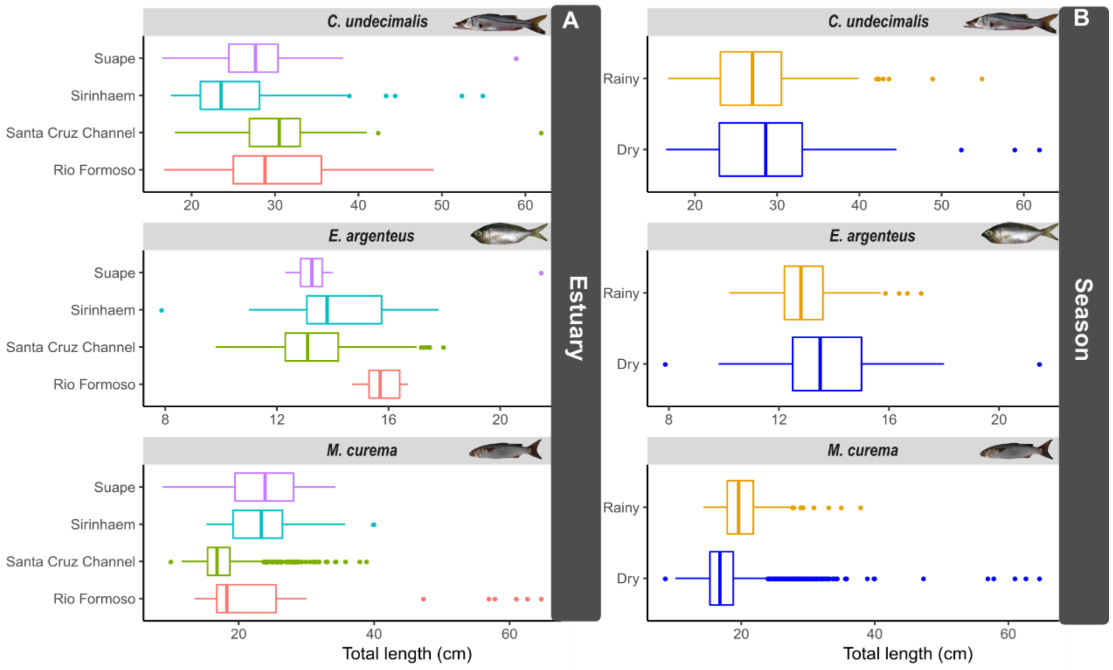


Figure 7: Boxplots of the total length for each of the three selected species showed A) by estuary (Suape, Sirinhaem, Santa Cruz Channel and Rio Formoso) and B) by season (dry and rainy).

There were significant size differences between estuaries for all three species (Supplementary table 4). *C. undecimalis* specimens fished in Sirinhaem were significantly smaller than those of the Santa Cruz Channel and Rio formoso (Supplementary table 5, Figure 7A). In addition, larger individuals of *E. argenteus* were captured in Rio Formoso than in Sirinhaem. Smaller sizes of *M. curema* were observed in the Santa Cruz Chanel compared to the other estuaries (Supplementary table 5).

Significant size differences between seasons were observed only for *M. curema* and *E. argenteus* (Supplementary table 4). A higher size range and larger individuals of *E. argenteus* were sampled in the dry season (Figure 7B, Supplementary table 4). On the other hand, larger individuals of *M. curema* were captured during the rainy season although all the individuals above 40 cm were fished during the dry season.

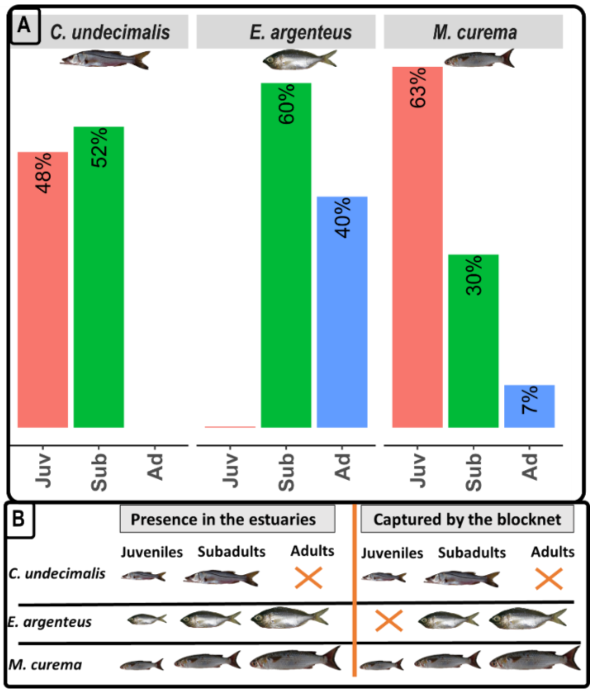


Figure 8: A) Relative catches of the ontogenetic category (Juv = juveniles, sub = Subadults, Ad = adults) for each selected species (E. argenteus, M. cure*ma and* C. undecimalis*) and B) comparison of the ontogenetic present in the estuaries (from Pelage et al. a Submitted) and captured by the block net.*

Ontogenetic catch composition varied between species and *M. curema* was the only species for which all ontogenetic stages were harvested (Figure 8). Indeed, adults of *C. undecimalis* were absent and juveniles of *E. argenteus* were negligible (only one individual fished in Sirinhaem). Thereby, all ontogenetic stages available in the estuaries were captured (Figure 8) except from the juveniles of *E. argenteus*. Individuals of *M. curema* were mostly captured as juveniles (representing 63 % of the total catch) whereas *C. undecimalis* and *E. argenteus* catches were dominated by subadults. However, juveniles constituted an important part of *C. undecimalis* specimens and adults represented a significant proportion of *E. argenteus* catches while they were not numerous for *M. curema*.

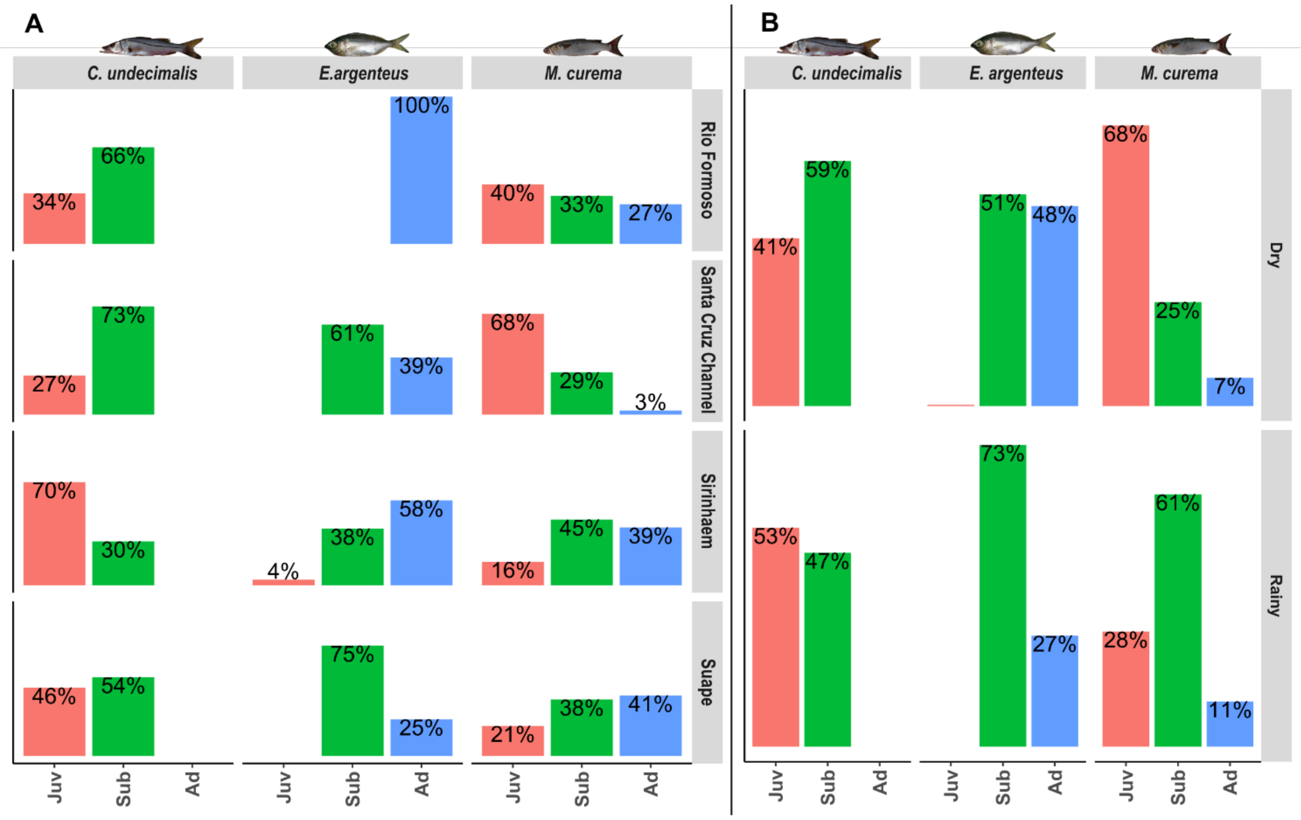
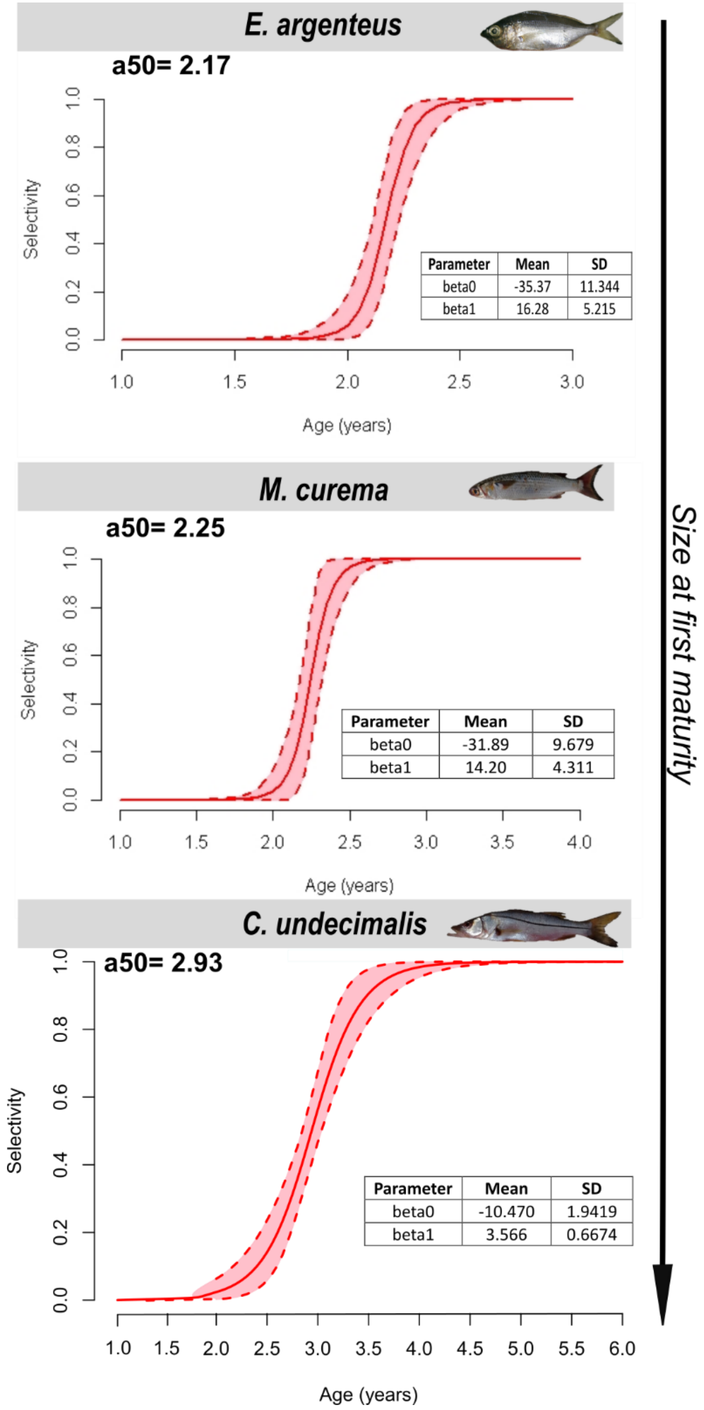


Figure 9: Relative catches of the ontogenetic category (Juv = juveniles, sub = Subadults, Ad = adults) for each selected species (E. argenteus, M. cure*ma and* C. undecimalis*).*

Harvesting of the different ontogenetic classes also varied between seasons and among the different estuaries (Figure 9). Adult individuals were predominant only in Sirinhaem and Rio Formoso for *E. argenteus* and in Suape for *M. curema*. Juveniles of *C. undecimalis* were very abundant in Sirinhaem (70% of the catch) while this species catches in the other estuaries were mainly represented by subadults. Immature individuals (juveniles and subadults) accounted for most of the catches of all three species in both seasons but a greater number of *M. curema* juveniles were harvested during the dry season (68%) compared to the rainy one (28%).

4.4.3. Block net selectivity for the 3 selected species



Parameter	Mean	SD
beta0	-35.37	11.344
beta1	16.28	5.215

Parameter	Mean	SD
beta0	-31.89	9.679
beta1	14.20	4.311

Parameter	Mean	SD
beta0	-10.470	1.9419
beta1	3.566	0.6674

Figure 10: Selectivity curves for each selected species (E. argenteus, M. curema *and* C. undecimalis*).*

The age at 50% selection(a50) increased with the size at first maturity (Figure 10). *E. argenteus* was the species for which the smaller range of years was captured (between 1 and 3 years) followed by *M. curema (*between 1 and 4 years) while the larger range was for *C. undecimalis* (between 2 and 6 years). Moreover, the parameters beta0 governing the slope of the selectivity curves were similar for *M. curema* and *E. argenteus* (-31.9 and -35. 4 respectively) while the selectivity curve of *C. undecimalis* exhibited a lower beta 0 (-10.470).

4.5. Discussion

Our study highlighted the great spatiotemporal variability between block net samples concerning fish family composition. Catches varied in abundance and composition among estuaries and between the samples of each estuary showing the plasticity of this fishing method. This gear harvested many families, but the fishing pressure on each of them fluctuated in time and space. Moreover, 65% of their catches consisted of Mugilidae and Gerreidae, which are known to include highly productive species on which the yield from this fishery depends. The tendency to harvest species that are productive in terms of abundance has already been reported by several studies (Islam and Ikejima, 2010; Castellanos-Galindo and Krumme, 2013). At species level, the number of individuals varied greatly from one sample to another, that is the species could be either absent from the catches or represented by several hundred individuals within an estuary or season. Regional (biogeography) and local (geomorphology, precipitation) factors have already been stressed as strongly affecting block net catches (Castellanos-Galindo and Krumme, 2013).

Hence, high abundance peaks were observed in the Santa Cruz Channel for *E. argenteus* and *M. curema* and overall Gerreidae and Mugilidae were very abundant in the catches of this estuary. Indeed, these two families are associated with high salinity (Chi-Espínola et al., 2018; Mai et al., 2018) and this estuary could represent a propitious environment thanks to its greater connection to the sea (Pelage et al. b Submitted). The presence of smaller individuals of *M. curema* in the Santa Cruz Channel suggests that this estuary is an important nursery for this species. Besides, samples with high abundance of *M. curema* were registered during the dry season (between September and February) which corresponds to their reproduction period (between November and March) (Oliveira et al., 2011). Smaller individuals of *M. curema* were captured during this season which suggests that the observed peaks were due to the arrival of juveniles seeking shelter in mangroves. In addition, peaks of abundance have been observed during the dry season for *E. argenteus*, which also coincides with its reproduction, occurring between October and November (Leão, 2016). Despite the known importance of estuaries as nursery grounds for Gerreidae, juveniles of *E. argenteus* were too small to be caught by the block net and, on the other hand, many adults were fished during this season. High evaporation and reduced river flow during the dry season may have resulted in higher salinity suitable for adults of this species (Ramos et al., 2016). In contrast, the lower salinity found in Sirinhaem favoured the relative importance of the Sciaenidae *B. ronchus* which are known to be euryhaline (Marceniuk et al., 2019). Moreover, smaller individuals of *C. undecimalis* were found in Sirinhaem which corroborates the studies highlighting the preference of this species for low salinity habitats during its juvenile period (Peterson and Gilmore, 1991; Gracia-Lopez et al., 2006).

Furthermore, the deployment of the block net in the mangroves gave us information on the use of this habitat by three migratory species. A lot of immature individuals were captured, corroborating the well-known fact that this habitat is a nursery for many migratory species (Laegdsgaard and Johnson, 2001; Sheaves et al., 2015; Whitfield, 2017). Only juveniles and sub-adults of *C. undecimalis* were captured because mangroves are important feeding areas for juveniles while adults are generally not present in estuaries (Gonzalez et al., 2019). Adults of *E. argenteus* accounted for a significant proportion of the catches of this species. This suggests that mangroves also provide important habitat for adults of Gerreidae which use estuaries as feeding and reproduction grounds (Ramos et al., 2014). Likewise, some individuals of *M. curema* have been captured because adults of this species can be found on muddy bottoms where they feed on organic remains and small particles (Menezes et al., 2015).

Yet, most of the large individuals of the targeted migrant species are more abundant in adjacent coastal areas or over the shelf (Pelage et al. a, Submitted). This reduces the probability of catching large adults even though the theoretical selectivity of the block net would allow them to be caught, thus preventing the age truncation observed in the selective fisheries of developed countries (Kuparinen et al., 2016). In fact, the block net caught an important range of sizes and ontogeny. Overall, all ontogenetic classes present in the estuaries were captured by the block net even though juveniles of *E. argenteus* were negligible but the catches of the three species were dominated by immature individuals (juveniles and subadults). Intertidal gears usually have a selectivity favouring the predominant modal class of catch to be below the size at first sexual maturity (Newman et al., 2007; Giarrizzo and Krumme, 2009) which is viewed as an indicator of overfishing (Stergiou 2002; Froese 2004).

In this study, we described for the first time the selectivity pattern of intertidal fixed gears used in the mangroves. The selectivity information (age at 50% selection, beta 0 and beta 1) could be used for a further assessment of block net impact on the three selected species using different fishing efforts. The observed S-shaped curves are a unique feature for fisheries using passive nets. This type of selectivity is achieved because the device is designed to catch all fish that cannot escape from the mesh, resulting in a selectivity that resembles that of active nets such as trawls (Stepputtis et al., 2016). Yet, intertidal devices differ in active gears in terms of the escapement possibility since generally, larger individuals may possess greater physical ability to escape the path of the active gear (Irwin et al., 2008). This lack of escapement possibility makes block nets very efficient in terms of yield capacity. However, current worldwide fishing policies aims at selective fishing to protect juveniles (Burgess et al., 2016; Froese et al., 2016) and preventing from growth overfishing (the depletion of the young part of the stock before it has reached its full biological and economic potential) is at the heart of ongoing fisheries management policies (Diekert and Rouyer, 2011; Fauconnet and Rochet, 2016). As a result, the block net would not be in line with the currently adopted fisheries guidelines since block net fisheries do not target large size classes.

On these regards, the main concern about these intertidal gears is their ability to catch juveniles of long-lived species such as Carangidae, Lutjanidae and Centropomidae, which have lower resilience and reproductive success and are therefore more vulnerable to fishing (Quetglas et al., 2016). Fishing in the mangrove, their nursery habitat, can thus constitute a real danger for these high trophic species. This could endanger their stocks and threaten the yield of coastal fisheries that target the adults of these species. Ecological and socioeconomics stakes are thus involved in the management of block net fisheries. Hence, conventional single-species methods like minimum size limits are generally suggested for the management of this type of artisanal fishing gears (Kolding and Van Zwieten, 2011).

For block net fisheries, the implementation of these limits would result in a shift of the selectivity curve to the right. Minimum size limits would be beneficial for adjacent coastal fisheries but would be inapplicable for block net fisheries. In fact, the reduction of juveniles in catches would have serious consequences for food security as they constitute an important source of protein for local fishing communities. In addition, the mesh size required to avoid the capture of long-lived species would prevent the catch of adults of small species such as *E. argenteus*. Thus, in the current context of resource depletion, since all size compartments are important for nutritional safety, focusing solely on the preservation of juveniles, the most productive stage, may not be the best option and this presumption should at least be questioned.

Catches from intertidal devices depend entirely on where and when they are deployed and therefore on fishermen's knowledge of ecosystem productivity. These gears are therefore suitable for practicing a balanced harvesting strategy, fishing the widest range of species according to the productivity of the ecosystem (Garcia et al., 2015). The selectivity and fishing area of the block net allowed the capture of a broad range of sizes, ensuring a high yield in line with balanced harvest objectives for food security. However, the sustainability of this fishery, particularly for long-lived species, will depend on the effort of the gear. Reduced effort from block net fisheries is essential to maintain adjacent coastal fisheries since their yield depends on the productivity and nursery function of the mangroves. Besides, it has been observed that mangrove areas suffering from reduced fishing pressure have similar characteristics as the “no fishing” areas and a positive relationship was observed between the impact of mangrove fisheries and the scale of the different uses of the catch (subsistence, marketed locally or marketed regionally) (Reis-Filho et al., 2019).

The dependence of block net on the presence of migratory species in the habitat implies that the yield of this fishery is strongly linked to the reproduction of these species and therefore to the fishing pressure exerted by coastal fleets targeting adults. Because the yields of the estuarine and coastal fleets are co-dependent, there is a need to distribute the fishing effort between the different fleets, as this would be beneficial for each of them. Moreover, the abundance of fish in mangroves, on which block net yield relies, depends on environmental conditions that vary naturally but can also be adversely impacted by anthropogenic pressures. Worldwide block net catches are likely to be directly affected by the increase in activities that have degraded mangroves in recent decades (Carugati et al., 2018) .

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Supplementary material:

Supplementary Table 4: Information on dates and location of sampling.

Date	Estuary	Season
31/10/2012	Sirinhaem	Dry
23/01/2013	Santa Cruz Channel	Dry
28/02/2013	Sirinhaem	Dry
24/05/2013	Rio Formoso	Rainy
24/05/2013	Santa Cruz Channel	Rainy
29/05/2013	Sirinhaem	Rainy
27/06/2013	Suape	Rainy
20/08/2013	Sirinhaem	Rainy
20/08/2013	Rio Formoso	Rainy
20/08/2013	Suape	Rainy
29/08/2013	Santa Cruz Channel	Rainy
01/11/2013	Santa Cruz Channel	Dry
15/11/2013	Sirinhaem	Dry
15/11/2013	Rio Formoso	Dry
15/11/2013	Santa Cruz Channel	Dry
15/11/2013	Suape	Dry
16/02/2014	Sirinhaem	Dry
15/03/2014	Rio Formoso	Rainy
15/03/2014	Santa Cruz Channel	Rainy
16/05/2014	Santa Cruz Channel	Rainy
05/06/2014	Sirinhaem	Rainy
18/01/2015	Suape	Dry
06/08/2015	Suape	Rainy
06/09/2017	Sirinhaem	Dry

Supplementary Table 5: Growth parameters for each of the three selected species where Lsub is the size separating the juveniles from the subadults, L_{50} the size at first asturity, t_0 the theoretical age at length 0, K the growth coefficient and $L\infty$ the asymptotic length.

Species	Family	Lsub (cm)	L_{50} (cm)	References L_{50}	$L\infty$ (cm)	K	t0
Eucinostomus argenteus (Baird & Girard 1854)	Gerreidae	8.5	13.5 (TL)	(Costa et al., 2018)	23.4	0.41	0.43
Mugil Curema Valenciennes,1836	Mugilidae	18	25.5 (TL)	(Oliveira, dos Santos Costa, and Chellappa 2011)	58.7	0.18	0.64
Centropomus undecimalis (Bloch, 1792)	Centropomidae	25	68.0 (FL)	(Perera-García et al., 2008)	105	0.07	-0.0976

Supplementary Table 6: Composition of the block net catches in the four estuaries with the number of individuals (N) and their proportion in the catches (%N).

	ITAPISSUMA		RIO FORMOSO		SIRINHAÉM		SUAPE		TOTAL	
	N	%N	N	%N	N	%N	N	%N	N	%N
MYLIOBATIFORMES										
Dasyatidae										
Hypanus guttatus (Bloch & Schneider, 1801)					1	0.07%			1	0.01%
ELOPIFORMES										
Elopidae										
Elops saurus Linnaeus, 1766	2	0.04%	13	0.43%					15	0.15%
Megalopidae										
Megalops atlanticus Valenciennes, 1847					1	0.07%			1	0.01%
ANGUILLIFORMES										
Muraenidae										
Gymnothorax funebris Ranzani, 1839			2	0.07%	1	0.07%			3	0.03%
CLUPEIFORMES										
Engraulidae										
Anchoa spinifer (Valenciennes, 1848)							1	0.08%	1	0.01%
Anchovia clupeoides (Swainson, 1839)	172	3.86%							172	1.69%
Cetengraulis edentulus (Cuvier, 1829)	604	13.55%							604	5.92%
Clupeidae										
Harengula clupeola (Cuvier, 1829)	45	1.01%							45	0.44%
Lile piquitinga (Schreiner, Miranda & Ribeiro, 1903)					6	0.43%			6	0.06%
Opisthonema oglinum (Lesueur, 1818)	1	0.02%							1	0.01%
Platanichthys platana (Regan, 1917)							1	0.08%	1	0.01%

SILURIFORMES										
Ariidae										
Ariidae sp.					38	2.75%			38	0.37%
Aspistor luniscutis (Valenciennes, 1840)					3	0.22%			3	0.03%
Aspistor quadriscutis (Valenciennes, 1840)					1	0.07%			1	0.01%
Cathorops agassizii (Eigenmann & Eigenmann, 1888)	2	0.04%			4	0.29%			6	0.06%
Cathorops spixii (Agassiz, 1829)					3	0.22%			3	0.03%
Sciades herzbergii (Bloch, 1794)	13	0.29%	10	0.33%	43	3.11%	8	0.61%	74	0.73%
GOBIIFORMES										
Eleotridae										
Guavina guavina (Valenciennes, 1837)					5	0.36%			5	0.05%
Gobiidae										
Gobionellus oceanicus (Pallas, 1770)					1	0.07%			1	0.01%
MUGILIFORMES										
Mugilidae										
Mugil.sp					3	0.22%			3	0.03%
Mugil brevirostris (Ribeiro, 1915)							8	0.61%	8	0.08%
Mugil curema Valenciennes, 1836	1991	44.65%	104	3.42%	143	10.33%	166	12.61%	2404	23.58%
Mugil curvidens Valenciennes, 1836			17	0.56%			85	6.46%	102	1.00%
Mugil incilis Hancock, 1830							11	0.84%	11	0.11%
Mugil liza Valenciennes, 1836	29	0.65%	1	0.03%	2	0.14%			32	0.31%
Mugil rubrioculus Harrison, Nirchio, Oliveira, Ron & Gaviria, 2007							32	2.43%	32	0.31%
ATHERINIFORMES										
Atherinopsidae										

Membras dissimilis (Carvalho, 1956)	1	0.02%							1	0.01%
BELONIFORMES										
Hemiramphidae										
Hemiramphus brasiliensis (Linnaeus, 1758)	1	0.02%							1	0.01%
Belonidae										
Strongylura timucu (Walbaum, 1792)			1	0.03%	2	0.14%			3	0.03%
CARANGIFORMES										
Echeneidae										
Echeneis naucratis Linnaeus, 1758			1	0.03%					1	0.01%
Carangidae										
Caranx crysos (Mitchill, 1815)			29	0.95%					29	0.28%
Caranx hippos (Linnaeus, 1766)	12	0.27%	11	0.36%			10	0.76%	33	0.32%
Caranx latus Agassiz, 1831	29	0.65%	66	2.17%	12	0.87%	44	3.34%	151	1.48%
Chloroscombrus chrysurus (Linnaeus, 1766)					1	0.07%			1	0.01%
Oligoplites palometa (Cuvier, 1832)			10	0.33%	2	0.14%			12	0.12%
Oligoplites saurus(Bloch & Schneider, 1801)	2	0.04%	18	0.59%			1	0.08%	21	0.21%
Selene setapinnis (Mitchill, 1815)							3	0.23%	3	0.03%
Selene vomer (Linnaeus, 1758)	3	0.07%	4	0.13%			4	0.30%	11	0.11%
ISTIOPHORIFORMES										
Sphyraenidae										
Sphyraena barracuda (Edwards, 1771)	5	0.11%			2	0.14%	31	2.36%	38	0.37%
Sphyraena guachancho Cuvier, 1829	3	0.07%					6	0.46%	9	0.09%
PLEURONECTIFORMES										
Paralichthyidae										

Citharichthys spilopterus Günther, 1862	6	0.13%			2	0.14%	1	0.08%	9	0.09%
Paralichthys brasiliensis (Ranzani, 1842)	1	0.02%					3	0.23%	4	0.04%
Achiridae										
Achirus lineatus (Linnaeus, 1758)					4	0.29%	1	0.08%	5	0.05%
Trinectes paulistanus (Miranda Ribeiro, 1915)					5	0.36%			5	0.05%
SYNGNATHIFORMES										
Syngnathidae										
Hippocampus erectus Perry, 1810							1	0.08%	1	0.01%
SCOMBRIFORMES										
Trichiuridae										
Trichiurus lepturus Linnaeus, 1758					9	0.65%			9	0.09%
PERCIFORMES										
Centropomidae										
Centropomus ensiferus Poey, 1860					25	1.81%			25	0.25%
Centropomus mexicanus Bocourt, 1868			44	1.45%	1	0.07%	50	3.80%	95	0.93%
Centropomus parallelus Poey, 1860	85	1.91%	60	1.98%	66	4.77%	124	9.42%	335	3.29%
Centropomus pectinatus Poey, 1860	5	0.11%	41	1.35%	24	1.73%	5	0.38%	75	0.74%
Centropomus undecimalis (Bloch, 1792)	49	1.10%	64	2.11%	87	6.29%	27	2.05%	227	2.23%
Gerreidae										
Diapterus auratus Ranzani, 1842	282	6.32%	1835	60.42%	146	10.55%	204	15.50%	2467	24.20%
Diapterus rhombeus (Cuvier, 1829)	11	0.25%	146	4.81%	119	8.60%	20	1.52%	296	2.90%
Eucinostomus argenteus Baird & Girard, 1855	534	11.98%	5	0.16%	24	1.73%	8	0.61%	571	5.60%
Eucinostomus gula (Quoy & Gaimard, 1824)	233	5.23%			28	2.02%	6	0.46%	267	2.62%
Eucinostomus havana (Nichols, 1912)	17	0.38%					13	0.99%	30	0.29%

Eucinostomus melanopterus (Bleeker, 1863)	14	0.31%	33	1.09%	12	0.87%	7	0.53%	66	0.65%
Eugerres brasilianus (Cuvier, 1830)	3	0.07%	147	4.84%	126	9.10%	93	7.07%	369	3.62%
Gerres cinereus (Walbaum, 1792)					1	0.07%			1	0.01%
Serranidae										
Epinephelus adscensionis (Osbeck, 1765)							2	0.15%	2	0.02%
Epinephelus itajara (Lichtenstein, 1822)							3	0.23%	3	0.03%
Epinephelus marginatus (Lowe, 1834)	1	0.02%							1	0.01%
Mycteroperca bonaci (Poey, 1860)							1	0.08%	1	0.01%
Haemulidae										
Haemulon parra (Desmarest, 1823)	3	0.07%					2	0.15%	5	0.05%
Haemulon plumierii (Lacepède, 1801)							2	0.15%	2	0.02%
Haemulopsis corvinaeformis (Steindachner, 1868)			1	0.03%	1	0.07%			2	0.02%
Pomadasys crocro (Cuvier, 1830)	2	0.04%	3	0.10%	6	0.43%			11	0.11%
Lutjanidae										
Lutjanus alexandrei Moura & Lindeman, 2007	150	3.36%	10	0.33%	3	0.22%	107	8.13%	270	2.65%
Lutjanus analis (Cuvier, 1828)	3	0.07%					4	0.30%	7	0.07%
Lutjanus jocu (Bloch & Schneider, 1801)	20	0.45%	2	0.07%	40	2.89%	17	1.29%	79	0.77%
Lutjanus synagris (Linnaeus, 1758)							9	0.68%	9	0.09%
Polynemidae										
Polydactylus virginicus (Linnaeus, 1758)	3	0.07%			8	0.58%			11	0.11%
MORONIFORMES										
Ephippidae										
Chaetodipterus faber (Broussonet, 1782)	19	0.43%	8	0.26%	1	0.07%	6	0.46%	34	0.33%
ACANTHURIFORMES										

Sciaenidae										
Bairdiella ronchus (Cuvier, 1830)	54	1.21%	36	1.19%	84	6.07%	4	0.30%	178	1.75%
Cynoscion acoupa (Lacepède, 1801)					1	0.07%			1	0.01%
Isopisthus parvipinnis (Cuvier, 1830)					11	0.79%			11	0.11%
*Micropogonias furnie*ri (Desmarest, 1823)					15	1.08%			15	0.15%
Ophioscion punctatissimus Meek & Hildebrand, 1925					1	0.07%			1	0.01%
Ophioscion sp.	1	0.02%							1	0.01%
Stellifer brasiliensis (Schultz, 1945)			35	1.15%					35	0.34%
Stellifer microps (Steindachner, 1864)	1	0.02%			1	0.07%			2	0.02%
Stellifer stellifer (Bloch, 1790)	2	0.04%							2	0.02%
SPARIFORMES										
Lobotidae										
Lobotes surinamensis (Bloch, 1790)	1	0.02%							1	0.01%
Sparidae										
Archosargus rhomboidalis (Linnaeus, 1758)	14	0.31%	3	0.10%	1	0.07%	1	0.08%	19	0.19%
TETRAODONTIFORMES										
Tetraodontidae										
Colomesus psittacus (Bloch & Schneider, 1801)	5	0.11%							5	0.05%
Lagocephalus laevigatus (Linnaeus, 1766)							5	0.38%	5	0.05%
Sphoeroides greeleyi Gilbert, 1900	2	0.04%							2	0.02%
Sphoeroides testudineus (Linnaeus, 1758)	23	0.52%	277	9.12%	258	18.64%	179	13.60%	737	7.23%
Total	**4459**		**3037**		**1384**		**1316**		**10196**	

Supplementary Table 7: Results of Kruskal tests between the number of individuals per sample and total length according to the estuary and the season for each species.

Total length (cm)	Estuary	*C. undecimalis*	25.256	1.365e-05
		E. argenteus	16.214	0.001025
		M. curema	263.2	< 2.2e-16
	Season	*C. undecimalis*	1.5288	NS
		E. argenteus	37.71	8.207e-10
		M. curema	192.93	< 2.2e-16

Supplementary Table 8: Results of Dunn's tests between the total length according to the estuary for each species.

Species	Rio F./ Sirinhaem	SCC/ Sirinhaem	Suape/ Sirinhaem	Suape/ SCC	Suape/ Rio F.	SCC/ Rio F.
C. undecimalis	0.0001	0.0001	NS	NS	NS	NS
E. argenteus	NS	0.0314	NS	NS	0.0050	NS
M. curema	0.0110	0.0000	NS	0.0000	0.0123	0.0000

Co-existing fleets harvesting migrant species across several heavily exploited tropical habitats: a complex pattern compatible with Balanced harvest.

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5. Co-existing fleets harvesting migrant species across several heavily exploited tropical habitats: a complex pattern compatible with Balanced harvest.

5.1. Abstract

Tropical small-scale fisheries (SSFs) are characterized by multidimensionality (fleets, gears, habitats, species). In this study, we investigated how a data-poor SSF in northeast Brazil affect migratory species along an estuarine-shelf gradient. As many of these targeted species move along this gradient throughout their life cycle, managing coexisting fleets that exploit the same resource is a challenge. The SSF is expected to offer a fishing pattern compatible with balanced harvest (BH) - the idea of catching all species and sizes according to their productivity -, enabling to discuss BH applicability for tropical SSFs management. We examined the variables influencing the probability of capturing a given ontogenetic stage, described the relative captures by gear type and draw a conceptual model to understand the SSF fishing patterns on migrant species. All ontogenetic classes of the selected species were harvested along the estuary-shelf gradient and several gears harvested each species. Moreover, young individuals and low trophic level species predominated in most gears achieving a pattern in line with BH. Our results suggested that size limitations would not be applicable to these coexisting fleets as the yield benefits will not be attained by the fleet which would suffer from size limitations. An ecosystem-based approach like BH, consistent with existing fishing patterns, could be an interesting alternative. Nevertheless, there should be empirical tests to give concrete guidelines about how to achieve the required effort distribution among the coexisting fleets for a sustainable exploitation of these connected habitats in poor documented SSFs.

5.2. Introduction

The context of global change raises questions about how humans will cope with the dwindling resources on which society depends for its proper functioning (Moore, 2016). Particular attention has been paid to the fisheries crisis, which consists in the gradual depletion of the sea resources, either due to pollution, overfishing or ecosystem degradation (Pauly 2019). Hence, the development of a set of relevant measures to ensure fisheries sustainability emerged as a priority considering its social, ecological, institutional and financial dimensions (Stephenson et al., 2018). Consequently, Ecosystem Approaches to Fisheries (EAF), which take into account social-ecological systems, are increasingly being discussed (De Young et al., 2018). Several ecosystem approaches have been suggested, submitting different management strategies (Ramirez-Monsalve et al., 2016; Koen-Alonso et al., 2019).

One of the proposed EAF is the balanced harvest (BH), i.e., “fishing across the widest possible range of species, stocks, and sizes in an ecosystem, in proportion to their natural productivity, so that the relative size and species composition is maintained” (Garcia et al., 2012). The rationale of BH is to maintain the structure of the ecosystem in accordance with the Principle n°5 of the Convention for Biological Diversity, by distributing and thus reducing the current excessive fishing pressure on large fish (Garcia 2016). It is presented as an EAF that maximizes yield by exploiting highly productive compartments that are not currently targeted, while minimizing the structural impact on the fish size spectrum (Kolding, Jacobsen, et al., 2016). Thereby, the objective of BH is to address simultaneously both biological concerns in fisheries management, namely how (fishing pattern) and how much (fishing effort) (Garcia et al., 2015). Hence, BH aims to ensure a sustainable resource with the high yield needed to satisfy the demands of a growing population (Kolding et al., 2016b). However, BH has hardly been put into practice due to the many controversies that have arisen over how BH could be applied and whether it could actually be beneficial (Burgess et al. 2016; Froese et al. 2016).

The possibility of a global implementation of BH has been severely criticized, especially in terms of economic cost (as catches are dominated by low-value individuals), feasibility and institutional questions (Burgess et al., 2016; Pauly et al., 2016). Nevertheless, BH has been recommended at the local level, particularly for small-scale fisheries (SSFs) in developing countries. Indeed, SSF fishing patterns are more "balanced" than those of large-scale fisheries making them more compatible with this EAF (Garcia et al., 2015). In fact, SSFs in developing countries harvest over a wider range of sizes because their catches are mainly destined to a market where size preferences are not essential (Kolding and van Zwieten, 2014). As a result, artisanal fishers are expected to make decisions maximising their yield generating spontaneously an aggregate fishing mortality rate approximately corresponding to the productivity of the fish stock in relation to body size (Plank et al., 2017). Unregulated SSFs in African lakes whose fishing pattern is seen as a proxy for the BH, achieved better results in terms of yield and thus food security compared to their regulated counterparts (Kolding, Jacobsen, et al., 2016). Still, the scarcity of empirical evidence of its benefits moderates the support of the scientific community (Froese et al., 2016).

Moreover, the establishment of BH in marine rather than continental systems would face additional difficulties. Indeed, the absence of physical boundaries in the seascape favours the emergence of large-scale ecological processes and makes it necessary to carry out inter-habitat studies (Marzinelli et al., 2015). Such an approach is essential to provide a better understanding of tropical small-scale fisheries which are best characterized by their multidimensional complexity (multi-species, multi-gear, multi-fleets, multi-ecosystems) (FAO, 2017; Herrón et al., 2019). Hence, the technical interactions (where several fish are caught together in the same net or fishing operation) and multi-species interactions (where population abundances influence each other through predation and competition) hamper the proper administration of these mixed fisheries (Thorpe et al., 2017; Dolder et al., 2018). Since productivity is species dependant, the resilience to harvesting will depend on reproduction strategies, growth, and other specific traits. For example, fish with large size, long lifespan, late maturity and low reproductive rate are more vulnerable to fishing (Cheung et al., 2005; Lucena-Frédou et al., 2017; Elsler et al., 2019).

Furthermore, productivity also varies according to the ontogenetic stages, with higher abundance in the early stages (Law et al., 2016). In this regard, the multi-ecosystem fishing pattern of SSFs involves the capture of migratory species in each of the connected environments they harvest. These species represent a linkage between coastal habitats since they need to use several parts of the coastal marine continuum along their life cycle (Berkström et al., 2013; Olds et al., 2016). Complementarity between different coastal ecosystems (i.e. mangrove, corals, seagrass) is therefore of great importance in meeting the changing needs of migratory species (Olds et al., 2018). In particular, estuaries are known as preferential nursery areas providing shelter and food for the juveniles of many commercial marine species, ensuring the growth of individuals that will contribute to coastal stocks (Archambault et al., 2018; Whitfield, 2017). Consequently, the complex spatial distribution of the different ontogenetic stages constitutes an additional difficulty in the management of these fisheries.

Although SSFs provide more than half of the catches in developing countries and 90-95% of their landings are destined for human consumption (Kelleher et al., 2012), their multidimensional functioning is understudied (Chuenpagdee et al., 2019). Their marginalization as low-yield, low-technology fisheries and their highly non-selective fishing methods, at odds with long standing management policies, contribute to the lack of their valorisation in the industrial world (Kolding and van Zwieten 2011; Kolding, Jacobsen, et al., 2016). In Brazil, SSF account for more than 90% of employment in the fishing sector and nearly 600,000 fishermen were directly engaged in full-time fishing activities in 2011 (Barange et al., 2018). Brazilian SSF are inadequately managed so that several stocks are overexploited and not systematically assessed (Gasalla et al., 2017). Yet, these SSF are a crucial source of protein for traditional communities more particularly in the northeast since 47% of the country's artisanal fishers are registered in this region (Gomes De Alencar et al., 2019).

Furthermore, northeast Brazil is the second region with the highest percentage of population (comprising almost 30% of the national population) (IBGE 2020). This implies an important demand for food supplies, particularly from the fishing sector, as 60% of Brazilian regular consumers of fish live in the North-Northeast region (Sonoda et al., 2012). It can be assumed that the population density of north-eastern Brazil, combined with its high dependence on natural resources for food, leads to intense fishing pressure on these oligotrophic environments and therefore high fishing mortality. The SSFs in northeast Brazil have fishing patterns whose complexity must be tackled for management purposes, since as in most part of the world, they are composed of multiple fleets, which have different fishing methods, fishing power, and fishing grounds (Hazin et al., 1998; Frédou et al., 2009). However, because they are multi-species fisheries that exploit connected environments, fishing pressure from one fleet can impact on the yield of another. Indeed, these co-existing fleets are likely to exploit the same resource because of the large number of marine species that migrate between different coastal and marine habitats.

In this study, we investigate how a multi-gear fishery in northeast Brazil may affect the complex life cycle of migratory species along an estuarine-shelf gradient. Here, the objective is not to exhaustively document all species captured by this SSF but to focus on model species that migrate ontogenetically across the estuarine-shelf gradient and are relevant to fisheries at all phases. These species therefore inhabit all environments exploited by this SSF so that their management requires a multi-gear, cross-shelf vision. The gear diversification in each environment exploited by this SSF is expected to allow a wide range of sizes and species to be harvested, providing a fishing pattern compatible with a BH approach. Thereby, we propose to (i) describe the relative catches by gear type in terms of ontogeny and species, (ii) characterize this SSF fishing pattern on migrant species and (ii) discuss the applicability of BH for the management of migrant species in tropical SSFs.

5.3. Material and Method

5.3.1. Study area

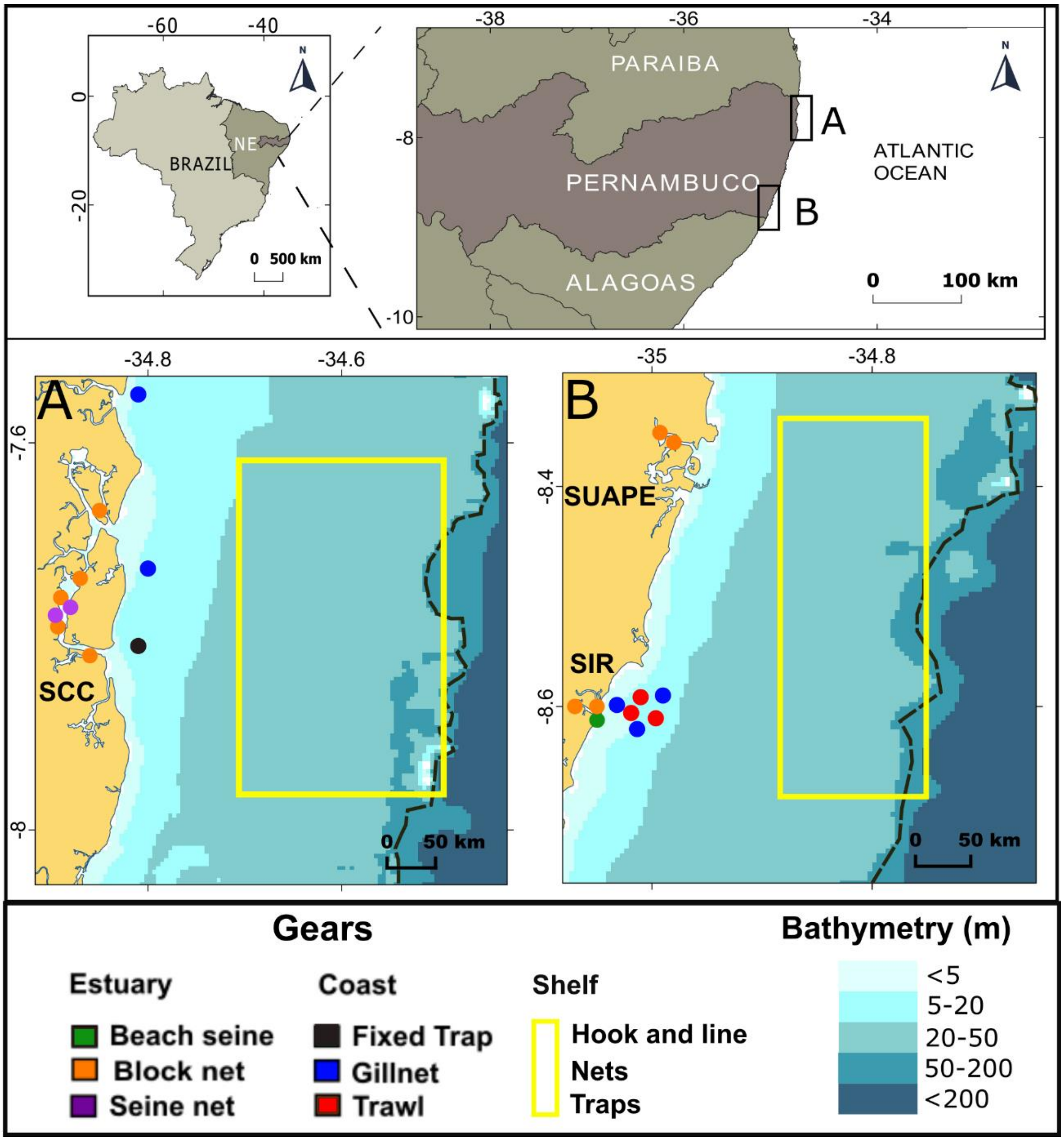

Figure 1: Map of the study area showing the sampling points in Pernambuco state, northeast Brazil, in A) the Santa Cruz Channel (SCC) in the north, and B) the Suape and Sirinhaem (SIR) estuaries in the south, and the fishing gears in use. The black dotted line represents the shelf break.

The study area extends along the continental shelf of Pernambuco state, encompassing a variety of connected sub-systems from estuaries to the shelf break (Figure 1). This narrow continental shelf (average 35 km wide) is characterized by a gentle slope bounded by a sharp shelf break at 50 - 60 m deep, warm (typically 26-29°C) and high salinity waters (typically 36-37) (Assunção et al., 2020), and a sedimentary cover composed of carbonate and terrigenous sediments. The sampling was carried out within two zones comprehending three estuaries: in the north, the Santa Cruz Canal, and in the south the Suape and Sirinhaem estuaries (Fig. 1). These areas were chosen because of their importance in the fishing production of Pernambuco and their different fishing patterns (Oliveira Lima and Andrade, 2018).

The Santa Cruz Canal is located on the north coast and forms the largest estuarine complex in Pernambuco. It constitutes a U-shaped surface area of 22 km and has two accesses to the Atlantic Ocean, resulting in high salinity (Silva, 2012). In addition to being the main fishing hub of Pernambuco, this estuarine complex is also important for tourism, aquaculture, artisanal fishing, industry and agriculture (Moura & Candeias, 2009, Quinamo 2006). The Suape estuary, is a coastal lagoon whose hydrodynamic conditions were modified by the construction of an industrial port in 1979 (Muniz et al., 2005). Sirinhaem is the smallest estuary among the three and exhibits low salinity and low depth (CPRH 2001). Artisanal fishing, agroindustry and trade are important socio-economic activities in the zone (Dos Santos et al., 2020).

5.3.2. Fish sampling

This study focussed on nine artisanal fishing gears operating across the estuarine-shelf gradient (whose characteristics are described in Table 1). These selected gears represent almost 90% of the landings in weight from the fisheries of northeast Brazil (Lessa et al., 2004) .

In estuaries and at the coast, sampling for research purposes was realised using six artisanal fishing gears (Table 1). The differences in mesh size (ranging from 25 to 70 mm) and gear types (passive and active) between and within estuarine and coastal environments (Table 1) are assumed to adequately provide the complementary selectivity required for BH. Those gears were deployed in their usual area of operation in collaboration with the local fishermen of the different fleets (estuarine and coastal). The fixed trap, trawl and gillnet were used at the coast, and the beach seine, block net and seine net in estuaries. In the northern and southern zones, different fishing gears are used given the specificities of tradition at each local fishery system. Indeed, local fishers from Sirinhaem and Suape do not operate the fixed trap and the seine net, while trawl and beach seine are not employed in the Santa Cruz canal (Oliveira Lima and Andrade, 2018). As a result, the complementarity of mesh sizes is not achieved at the scale of each zone. This is not necessarily a limitation in this work since BH considers that fisheries should be balanced as a whole and not necessarily individually (Garcia et al., 2012). A total of 120 samples were conducted between 2011 and 2017, of which 56 between September and February and 64 between March and August, corresponding to the local dry and rainy seasons, respectively (Medeiros et al., 2001). Fish were collected at depths ranging between 1 and 2 metres in estuarine environments, between 2 and 14 metres on the coast. In the field, individuals were frozen. Thereupon, fish were transported to the laboratory, where they were identified, sexed, measured to the nearest centimetre and weighed to the nearest gram.

Over the shelf, line, nets and traps catches were obtained using landings data from the program REVIZEE (Evaluation of the Potential of the Live Resources from Brazilian Exclusive Economic Zone), which was established in 1996 in order to provide information on the ecology and fishery of main living resources along the entire Brazilian coast (Frédou and Ferreira, 2005). Although each of the three gear categories includes a range of gear with different dimensions, the traps, the longlines and nets were selected to allow for complementary harvesting because of their differences in fishing methods (passive/active) and sampled habitats (Table 1). Sampling took place from August 1996 to March 2000, but these data are the only ones documenting multi-gear catches of northeast Brazil SSF on the shelf. Only fish caught in the study area between 20 and 200 m depth were selected from the database. Fish from the REVIZEE program were identified and measured in situ.

Twelve species, belonging to 7 genera and 6 families (Supplementary material 1) presenting ontognetic migrations and constituting an essential source of protein for the local population were selected: *Caranx hippos*, *Caranx latus*, *Centropomus parallelus*, *Centropomus undecimalis*, *Diapterus rhombeus*, *Eucinostomus argenteus*, *Eucinostomus gula*, *Lutjanus analis*, *Lutjanus jocu*, *Lutjanus synagris*, *Micropogonias furnieri* and *Mugil curema*. Their trophic level varied between 2.0 (*M. curema)* and 4.4 (*L. jocu*).

Table 2: Information about gears and sampling.

	Gear	**Number of samples**		**Depth (m)**	**Mesh (mm)**	**Dimensions (m)**	**Habitat**	**Fishing method**
		Dry	**Rainy**					
Shelf (depth>20 m)	Hook and line (REVIZEE Project)	REVIZEE Database		20-200	-	-	Hard substrate / Shelf-break	Passive
	Traps (REVIZEE Project)						Hard substrate / Shelf-break	Passive
	Nets (REVIZEE Project)						Sandy / Shelf-break	Active
Coast	Fixed trap	10	14	6	70	Diameter: 27	Sandy	Passive
	Gillnet	10	12	2-14	50	horizontal*vertical:200*2.5	Sandy	Passive
	Trawl	11	14	2-13	25	horizontal*vertical:10*6.1	Sandy	Active
Estuary	Beach Seine	7	8	1 -2	25	horizontal*vertical:180*5	Channel	Active
	Block net	9	8	1-2	60	horizontal*vertical:100*4.3	Mangrove	Passive
	Seine net	9	8	1 -2	10	horizontal*vertical:75*3.3	Channel	Active

5.3.3. Data Analysis

Fish were separated into three ontogenetic stages (juveniles, subadults and adults) according to a method based on two criteria (Ramos et al., 2012): (1) the inflection point of the length-weight curve by species, used as a reference point to distinguish juveniles from sub-adults, and (2) the size at first maturity (L_{50}) according to the literature found for regions with a climate similar to the study area (Supplementary material 1 and 2). The graphical criterion for separating juveniles from sub-adults assumes that individuals whose metabolism is dedicated to length gain can be distinguished from specimens that are beginning to gain weight. This categorization provides a semi-quantitative understanding of catch structure in terms of ontogeny.

The species were allocated into two categories according to their trophic levels (TL) values when adults that were estimated for the region with an Ecopath model (Lira et al., in press) or found on Fishbase (Froese and Pauly 2018) (Supplementary Table). Hence, species in the High Trophic Level (HTL) category were those with a TL superior to 3.5 (piscivorous diet when adult) while Low Trophic Level (LTL) category encompassed species with a TL inferior to 3.5 (not dominantly piscivorous when adult). This division was established because forage and predatory species are affected differently by exploitation and, therefore, management trade-offs between these coexisting fisheries are a major challenge for mixed fisheries (Houle et al., 2013). In addition, a conservation status has been assigned to each species based on the International Union for Conservation of Nature (IUCN) Red List criteria, evaluated by a regional classification, (ICMBio, 2018) which includes 10 categories: 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 evaluated (NE).

Kruskal Wallis tests were used to check whether the three categorical factors "zone", "gear", "habitat" and the numerical variable "depth" influenced catches in terms of ontogenetic stages (juveniles, sub-adults, adults). The habitat variable included five categories "sandy", "channel"," mangrove", "hard substrate" and "shelf-break", the latter category is for shelf gears when used beyond a depth of 60 metres whereas the factor "gear" had nine levels (see Table 1) while the variable "zone" was divided into two categories (north and south). Considering that each gear is associated with a habitat of a given depth and that the gears installed in both zones are different, these categorical factors are related to both habitat and gear characteristics. Kruskal-Wallis tests were also applied to examine the relationship between the ontogenetic classes caught and two species-related numerical variables (namely the size at first maturity and the trophic level of the species).

A random forest model was run to determine which variable contributed the most to the probability of capture of a given ontogenetic stage. Random forests are combinations of decision trees that are built by a bootstrap procedure on the original data set (Breiman, 2001). Randomization in the selection of predictors for each tree in the forest increases the differences between them, and therefore decreases their autocorrelation as well as the forest classification error rate (Breiman, 1996). A by-product of this method is the ranking of model predictors according to their importance by calculating the change in error when one variable is excluded and the others are retained, thereby identifying the explanatory variables that have the most impact on the response variable (Liaw and Wiener, 2002).

To construct the random forests, the ontogenetic stage (juveniles, subadults or adults) was the categorical variable to be predicted, while the following six explanatory variables were included: 1) Species size at first maturity (L_{50}), 2) Zone, 3) Habitat, 4) species TL, 5) Depth, and 6) Gear. We used the Random forest R package (Liaw and Wiener, 2002) together with the Caret R package (Kuhn et al., 2020) to tune our model by fitting the optimal number of variables to each decision tree and selecting the number of trees needed for the model.

Bar charts showing the percentage of each size class in the catch-per-gear composition, in number, were elaborated. The proportional catches of each gear in terms of species were also plotted as well as the catch-per-gear composition by species. These graphs were used as the basis for the development of a conceptual model to synthesize the fishing pattern for the full range of gears used. All the analysis were performed in R, version 3.5.2 (R Core Team, 2019).

5.4. Results

A total of 10,923 individuals of the selected species were caught where 4,452 were captured with seine nets, 3,708 with block nets, 1,275 with hooks and lines, 519 with beach seines, 385 with trawl nets, 234 with the nets deployed on the shelf, 139 with traps, 127 with gillnets, and 84 with fixed traps. Ontogenetic categories in catches varied significantly according to the zone, gear, habitat, depth, L_{50} and trophic level of the species ($p < 0.05$ for all variables) (Supplementary material 3). The out-of-bag estimate error of the random forest model was 16.9 % in the predictions of catch for a given size class. For the accuracy of the model, the species related variable L_{50} and trophic Level were the most important to identify the ontogenetic size of each fish followed by the habitat/gear related variables (gear, zone and depth, and then habitat; Figure 2).

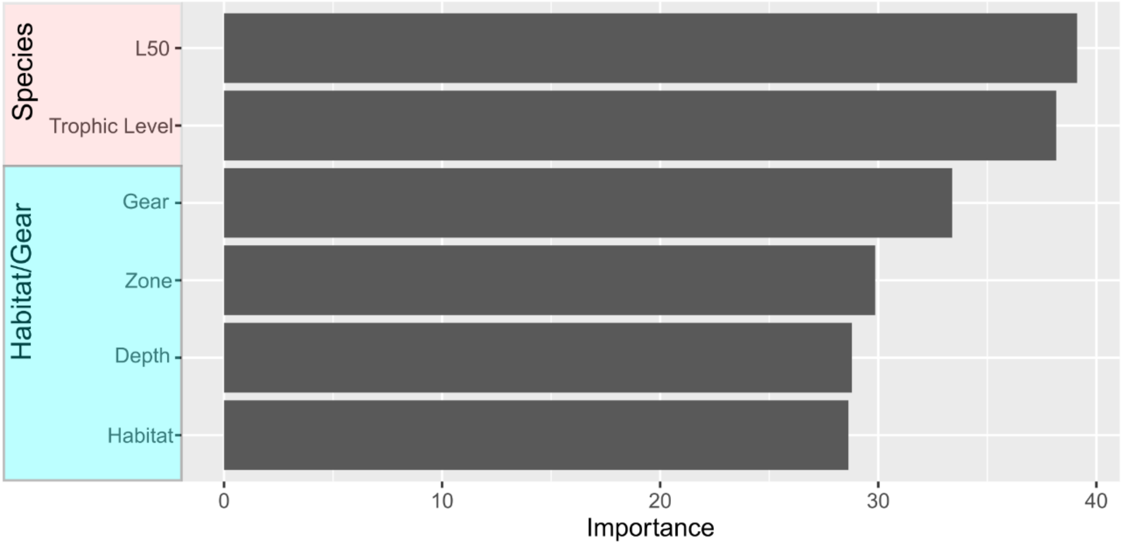

Figure 2: Variable importance barplot in terms of square error minimization for the ontogenetic categories based on a random forest with 500 trees.

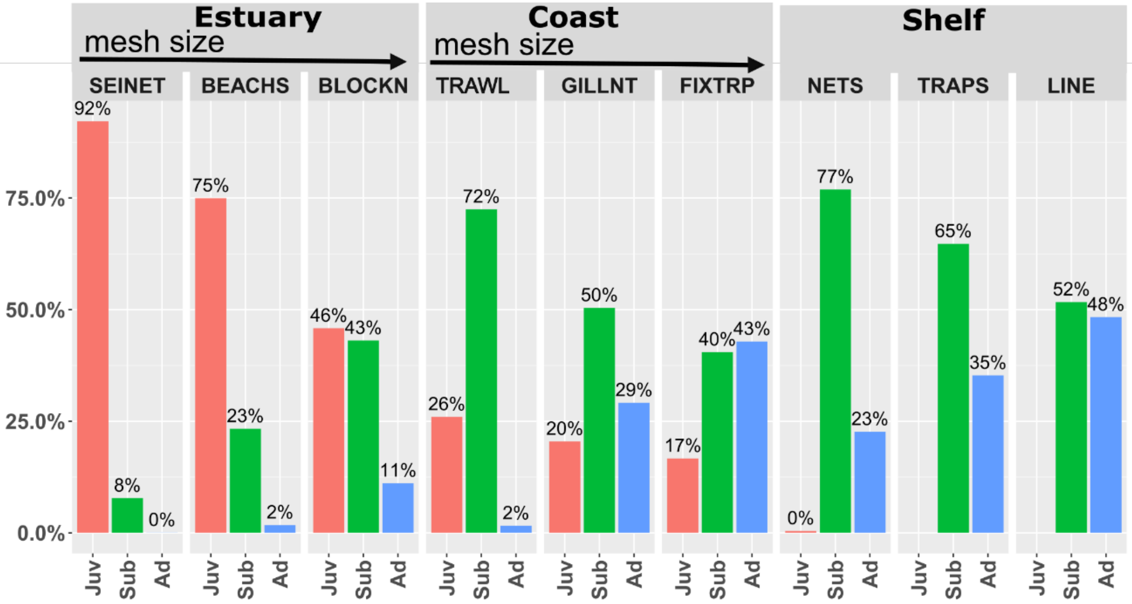

Figure 3: Relative catches (in number) of the ontogenetic categories (Juv = juveniles, sub = Subadults, Ad = adults) by gear type (SEINET=seine net, BEACHS=beach seine, BLOCKN=block net, TRAWL=trawl, GILLNT=gillnet, FIXTRP= fixed trap, NETS=nets, TRAPS= Traps and LINE=line) within the studied environments (estuary, coast and shelf).

The ontogenetic composition of catches by gear in number (Figure 3) showed that in estuaries, all gears caught mostly juveniles, which made up almost all the seine net catches (92%), 75% of beach seine catches and 46% of block net catches (Figure 3). Conversely, adults were always the least harvested class for all the gears deployed in estuaries. Overall, the percentage of juveniles decreased (from 92% to 46%) while the proportion of subadults and adults increased (from 8% to 43% and 0% to 11% respectively) with mesh size.

On the coast, subadults dominated both trawl (72%) and gillnet (50%) catches. Although subadults were also important in the fixed trap captures (40%), this gear retained more adults (43%). The increase in mesh size was accompanied by a slight decrease in the proportion of juveniles (from 26% to 17%) and a drop in subadults (from 72% to 40%), but this was also associated with an increase in the contribution of adults in the catches (from 2% to 43%).

On the shelf, catches of all gears were dominated by subadults. In addition, these gears caught only subadults and adults (although juveniles made up a negligible proportion of the catch with nets). Subadults accounted for 77% of the net catch, 65% of the traps catch and 52% of the line catch. The line was the gear where the proportion of adults was the highest.

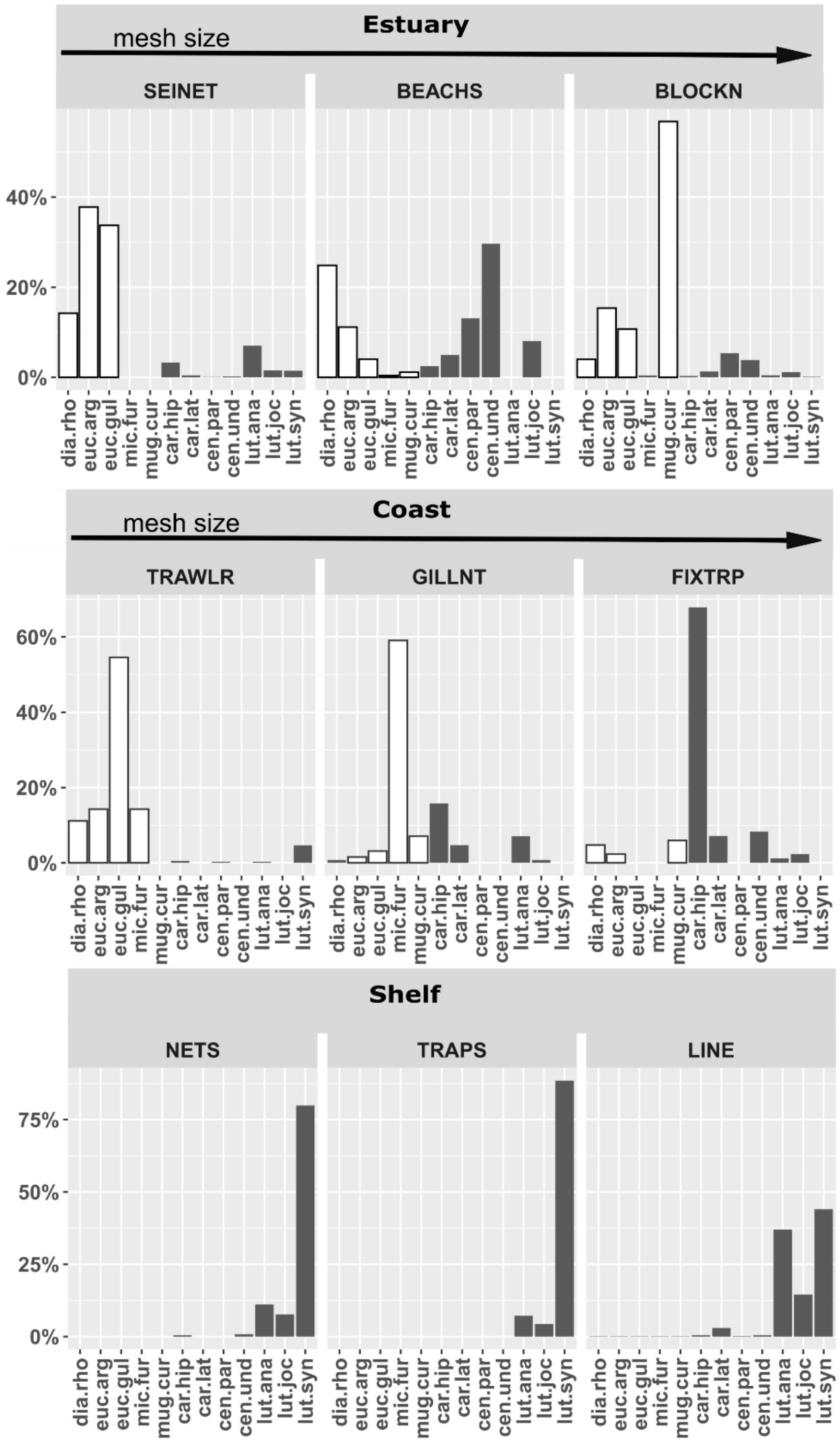

Figure 4: Relative catches of the selected species by gear type (seine net, beach seine, block net, trawl, gillnet, fixed trap and line) within the studied environments (estuary, coast and shelf). The species are divided in two categories low and high trophic level (in black and white respectively).

Species composition varied by gear (Figure 4). Overall, species from the low trophic level category accounted for most of the seine net, block net, trawl and gillnet catches. On the other hand, beach seine, nets, traps and line caught mainly species from the high trophic level category. On the coast, the proportion of species from the high trophic level category increased with mesh size. Gerreidae dominated the seine net catches (38% of *E. argenteus,* 34% of *E. gula,* 14% of *D. rhombeus*) and trawl catches (55% of *E. gula*, 14% of *E. argenteus* and 11% of *D. rhombeus*). They were also important for beach seine (25% *D. rhombeus,* 11% *E. argenteus*) and gillnet (15% of E. argenteus and 11% of *E. gula*), although beach seine catches were dominated by Centropomidae (30% of *C. undecimalis* and 25% of *C. parrallelus*) and the gillnet captured mostly *M. curema* (57%). The fixed trap mainly caught *C. hippos* (68%).

Over the shelf, high trophic level (HTL) species were all fished by at least one of the gears and the genus Lutjanus dominated the catches, representing almost the totality of the catches for the three gears (100% of the traps catches, 99% of the nets catches and 97% of the line catches). In particular, *L. synagris* predominated in the catches of the three gears (88% of the traps catches, 80% of the nets catches and 44% of the line catches).

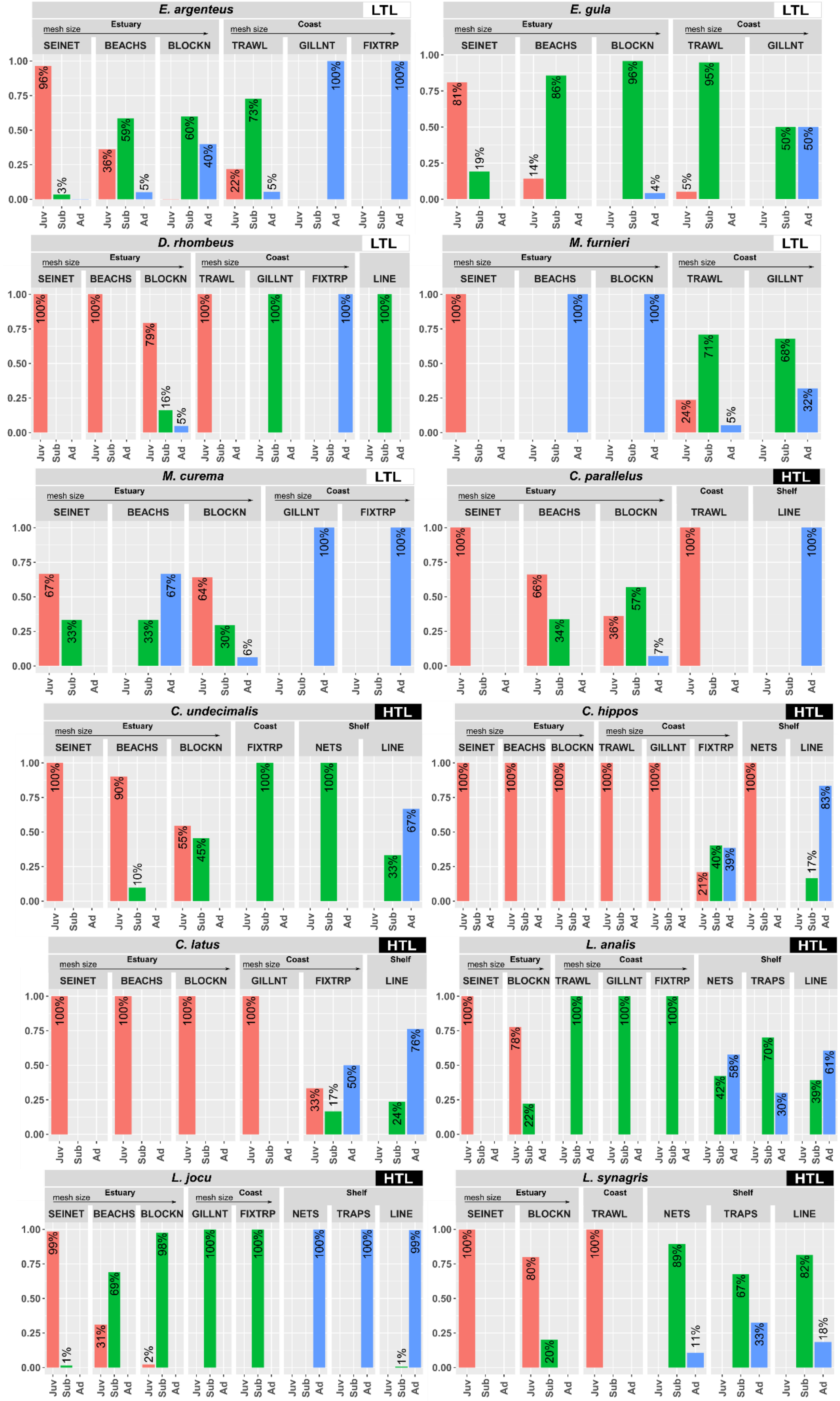

Figure 5: Relative catches of the ontogenetic category (Juv = juveniles, sub = Subadults, Ad = adults) by gear type (seine net, beach seine, block net, trawl, gillnet, fixed trap and line) for each selected species within the studied environments (estuary, coast and shelf). The species are divided into two categories LTL and HTL (low and high trophic level).

All ontogenetic categories were harvested for each species along the estuarine to shelf gradient with a growing importance of the larger ontogenetic stages with mesh size (Figure 5). All species were caught by at least six artisanal gears, but none was retained by all nine gears. *C. hippos*, *L. jocu* and *L. analis* were captured by eight gears, while fish of the genus *Eucinostomus* (*E. argenteus* and *E. gula*) were caught by all six gears operated in estuarine and coastal environments. Fish of the genus *Lutjanus* were the only species which were caught by the three fishing gears of the shelf.

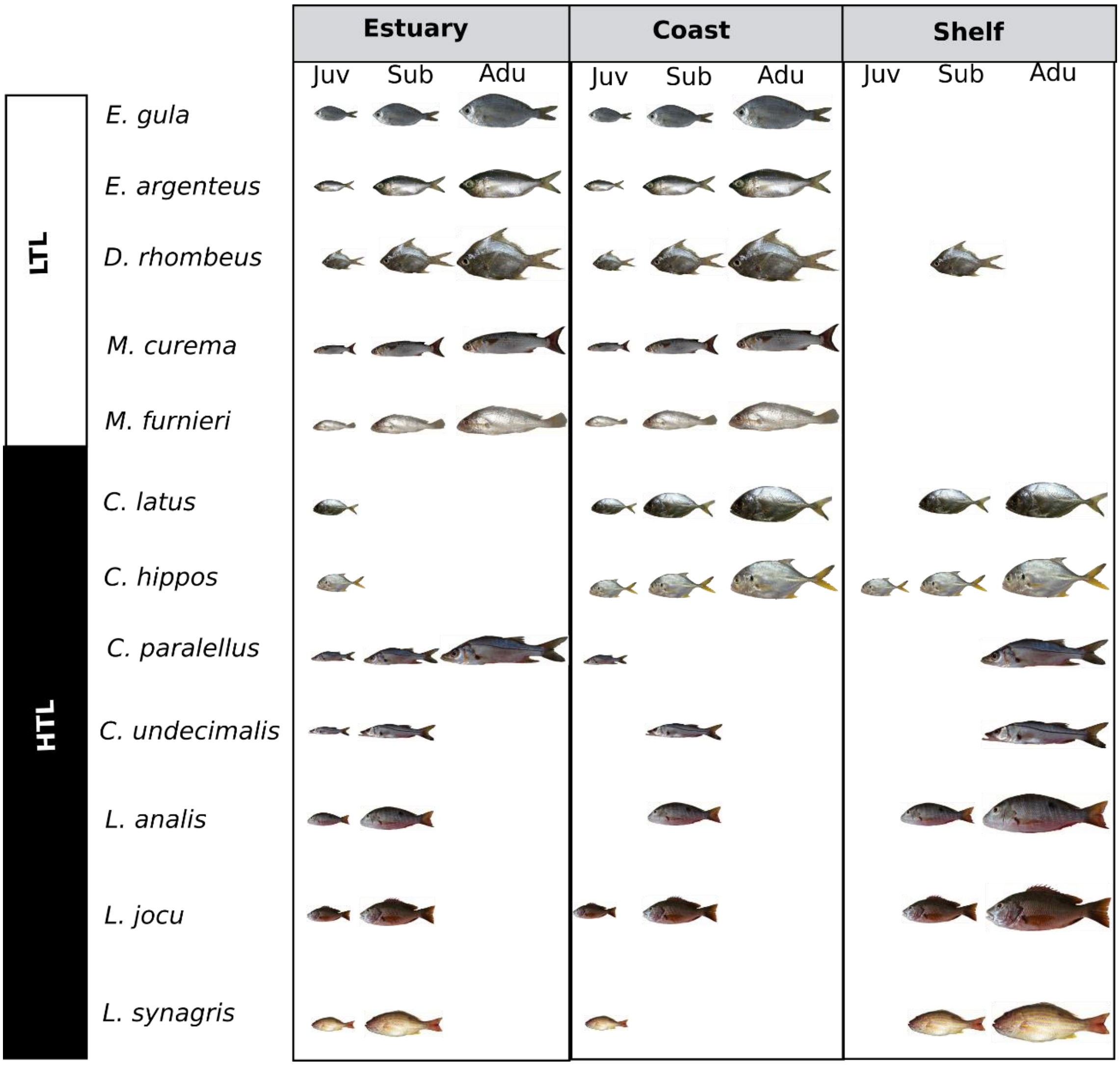

Figure 6: Schematic representation of the occurrence of ontogenetic categories (juveniles, subadults and adults) in the catches of each selected species within the studied environments (estuary, coast and shelf).

The spatial distribution in the catches of the different ontogenetic categories differed between the species (Figure 6). The five species from the low trophic level category were captured at all three ontogenetic classes in estuarine and coastal environments. On the shelf, species from the low trophic level category were not harvested, except for one individual of *D. rhombeus*. Furthermore, the proportion of a given ontogenetic class in the catches varied between species (Figure 5). Species from the high trophic level category were captured by the gears present in all three environments, but the presence of a given ontogenetic stage differed among species.

In estuaries, juveniles of all species have been captured, particularly by the seine net (Fig. 5). Subadults of all species were also fished except for the species of the genus *Caranx* (*C. latus* and *C. hippos*) which were caught only as juveniles (Figures 5 and 6). Only adults of the species from the low trophic level category (*M. furnieri, M. curema, D. rhombeus, E. argenteus,* and *E. gula*) were captured in estuaries as well as few adults of *C. parallelus* that were harvested by the block net, the estuarine gear with the largest mesh size (Figure 5). Nevertheless, the adults *of D. rhombeus* and *E. gula* were fished only by the block net and did not account for a high percentage of these species captures (5% and 4% respectively).

In the coastal zone, subadults dominated catches of most species for most gears, but juveniles made up a significant proportion of catches for some high trophic level species (*C. hippos, C. latus, C. parallelus, L. synagris*). The two high trophic species *C. paralllelus* and *L. synagris* were only fished as juveniles. Adults dominated coastal gillnet and fixed trap catches for low trophic level species (*M. curema, D. rhombeus, E. argenteus,* and *E. gula*) while adults of high trophic level species were present only in fixed trap catches with the largest mesh size (*C. hippos, C. latus, C. undecimalis*) (Figure 5). In addition, except for the fish of the *Caranx* genus, no adults of high trophic level species were captured on the coast (Figures 5 and 6).

Over the shelf no juveniles were harvested except for 1 juvenile of *C. hippos*. Fish of the genus *Centropomus* were captured only as adults, while other high TL species were captured as sub-adults and adults. *L jocu* was exclusively fished as adults by the three gears (except 1% of subadults in the line catches). *L. synagris* was predominantly fished as subadults and *L. analis* was captured mostly as subadults by the traps but adults dominated the line and nets catches.

Overall, fish of the same genus were not caught with the same gears (Figure 5). In addition, differences can be observed between species of the same genus regarding their ontogenetic composition of catches in terms of proportion: *E. argenteus* was harvested more in its adult phase while subadults dominated catches of *E. gula*.

Likewise, *C. parallelus* was fished only at the juvenile stage at the coast by the trawl while *C. undecimalis* was caught in the subadult phase with the fixed trap. *Caranx* was the only genus with a similar ontogenetic distribution and similar catches in terms of ontogenetic proportion, consisting mainly of juveniles both in the estuary and on the coast (except for fixed trap catches where all three stages were fished) and essentially adults on the shelf (Figure 5).

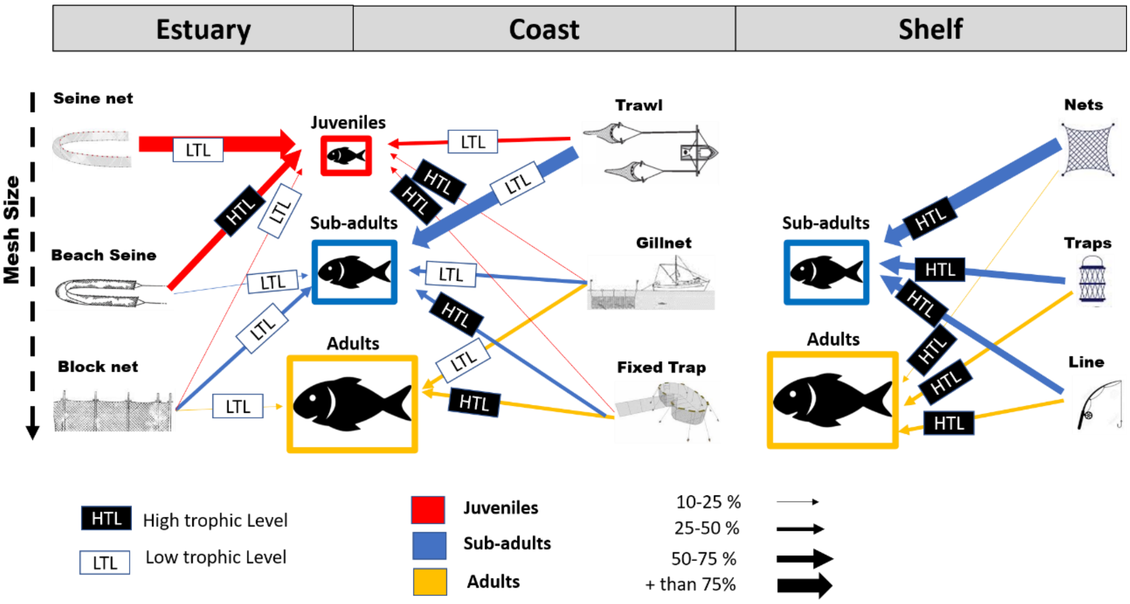

Figure 7: Conceptual model of the overall fishing pattern of artisanal gears according to the environment. The size of the arrows is proportional to the relative importance of each ontogenetic category in the catch of each gear. This figure also represents the category of species (low trophic level or high trophic level) which dominated the catches of each gear according to the ontogenetic category.

The overall pattern of the fishery was summarized by a conceptual model (Figure 7) showing that all size classes at all trophic levels are fished along the estuary-shelf gradient. At the environment level, this was achieved also at the coast. On the other hand, in the estuaries only the species from low trophic level category are harvested at all size classes while on the shelf, only the species from the high trophic level category are captured. Moreover, in each environment the ontogenetic classes dominantly targeted changed. Indeed, in the estuaries, the targets are essentially the juveniles, while on the coast it is the subadults, and on the shelf, only adults and subadults are caught. However, in the three environments the immature classes (juveniles and subadults) constituted the biggest part of the catches.

5.5. Discussion

5.5.1. A complex pattern which cannot be managed with only size limitations

The fishing gears deployed in this study represented a simplified SSF pattern, comprising the main gear types used by co-existing fleets across an estuary-continental shelf gradient. This multi-gear/multiple habitats fishing pattern is common among tropical SSFs which are characterised by their multidimensionality in terms of fleets, species, gears and habitats (FAO, 2017; Herrón et al., 2019). In this study, the complementary mesh-sizing in estuarine and coastal environments as well as the diversification of gear types on the shelf led to this multidimensionality. In a context of insufficient data, this type of sampling allows a characterization of the fishing pattern of a tropical SSF, although knowledge on fishing effort is not available. This characterization is essential for designing the appropriate management measures that these systems urgently need, as they are threatened by intense fishing pressure due to the high dependency of populations on their natural resources for food (Robinson et al., 2020).

Our study shows that the impact of this mixed SSF on migratory species resulted in a complex pattern where each of the twelve studied species was harvested by six to eight gears (Figure 5), which is common in mixed fisheries (Newman et al., 2018). Hence, it is necessary to understand the global pattern of these co-existing fleets because their yields are co-dependent. Overall, the multiplicity of fishing gears with complementary selectivity harvested all the ontogenetic classes of the twelve selected species along the estuary-shelf gradient. The probability of capturing a given ontogenetic stage was highly dependent on species-related parameters and was a function of the characteristics of the gear and the habitat where it was operated. Indeed, in this type of tropical SSFs, the spatial differences observed in the catches reflect not only the inherent selectivity of the gears, but also the habitat of the fishing ground for each gear (Herrón et al., 2019). Consequently, single gear approaches would not accurately quantify the impact of tropical SSFs.

In shallow waters (estuaries and coast) most gears harvested mainly the selected low trophic level (LTL) species (Figure 3), which is a typical feature of all tropical SSFs

(Sydeman et al., 2017). The three ontogenetic stages of these LTL species were caught in estuaries and on the shallow coastal environments. The presence of adults of the LTL category, although not abundant in the estuary catches, indicates that spatial segregation in terms of ontogeny is not very pronounced for these species. Adults of these species use estuaries as feeding grounds (Ramos et al., 2014) even if they are generally more abundant in adjacent coastal areas (Franco et al., 2012; Ramos et al., 2016). On the other hand, the selected LTL species were not fished over the shelf even if they are known to occur there, especially on hard substrates (Eduardo et al., 2018). The absence of these LTL species in shelf catches suggests that hard substrates were not harvested by gears with the appropriate selectivity to catch them. This could be beneficial for their stocks since these hard substrates in the shelf are generally important for larger fecund specimens of low trophic levels species (Reis-Filho et al., 2019). This study reveals that these large individuals of the selected LTL species are not suffering from fishing pressure over the shelf which could enhance the sustainability of their stocks.

Conversely, for the selected HTL species, the harvesting of all ontogenetic classes only happened at the estuary -shelf scale. The increase in average size with depth has already been reported for these species in the area (Frédou and Ferreira, 2005), resulting in significant ontogenetic spatial differences in the catches (Figure 6). All selected HTL species were caught as juveniles in estuaries, and this ontogenetic phase was also frequently captured in shallow coastal waters, thus corroborating the importance of these environments for the growth of these species (Gillanders et al., 2015). In contrast, larger sizes of migrant species from high trophic level (HTL) were scarcely captured in shallow waters (coastal and estuarine) where the diversity and complementarity of gear deployed should have been adequate to catch them if they were present. Moreover, even if the three gears used over the shelf fished larger sizes, their catches were mostly made of subadults. As the fleets deployed over the shelf target large individuals, the predominance of young individuals in shelf catches could be a sign of depletion of large fish in these environments due notably to overfishing.

Overall, juveniles and subadults (individuals below L_{50}) dominated the capture in eight of the nine gears deployed (Figure 2). This pattern, where the predominant modal class of catch is below the size at first sexual maturity, is common for tropical SSFs (Cavole et al., 2015) but is viewed as an indicator of overfishing (Froese, 2004;

Stergiou, 2002). Yet, since adults are not the major target of all the fleets, this fishing pattern does not aim at the selective fishing pattern encountered in industrial fisheries (Fauconnet and Rochet, 2016). Consequently, the most commonly applied regulations, which aim at selective fishing to protect juveniles (Burgess et al., 2016; Froese et al., 2016) and prevent against growth overfishing (the depletion of the young part of the stock before it has reached its full biological and economic potential) (Diekert and Rouyer, 2011; Fauconnet and Rochet, 2016) are not applicable for these systems as they stand.

Still, concerns against growth overfishing are particularly relevant for HTL species. Indeed, these long-lived species, some of which are on the list of near threatened species, have lower resilience and fecundity and are more vulnerable to exploitation (Quetglas et al., 2016). Minimum size limits are the traditional single-species method used to mitigate growth overfishing in SSFs (Kolding and van Zwieten, 2014). Nonetheless, the implementation of these limits in northeast Brazil SSFs would result in a transition to a fishing model aiming at catching large sizes involving the use of adapted gears. This guideline to prevent stock depletion is mainly based on yield-per-recruit (YPR) models, whereby yield maximisation is achieved by delaying the age of capture (Kolding et al., 2016a). However, for this strategy to work, the benefits in terms of yield must be obtained for those who suffer from size limitations. Since different fishing systems are operated by different fleets along the estuarine - shelf gradient, the yield benefits may not be perceived by all fleets within an acceptable timeframe. In particular, this transition could favour some coastal fleets, but would be unachievable for estuarine fleets whose yield depends on small organisms.

Besides, minimum harvest size regulations are a key factor in harvest induced selection since they promote small size, early reproduction and recruitment overfishing (Kuparinen et al., 2016). This could exacerbate the problem of large size HTL depletion observed in shelf catches. Indeed, long-lived fish such as *Lutjanidae*, need big old fat fecund female fish (BOFFFFs) to promote stock productivity and stability (Lowerre-Barbieri et al., 2015). Compared to smaller mature females, BOFFFFs produce many more eggs, often larger, which can develop into larvae that grow faster and are more resistant to starvation (Hixon et al., 2014). Large size depletion could be alleviated by upper size limits. In our study, these upper size limits could be relevant over the shelf, where gears may harvest large fishes, to increase the survival of BOFFFFs. Nevertheless, upper size limits would only be effective if the fishing effort applied at intermediate sizes is moderate enough to allow individuals to survive until they reach the upper size limit (Birkeland and Dayton, 2005).

Furthermore, this study highlights that the global fishing pattern of the co-existing fleets is strongly linked to spatial scale and gear complementarity. Thus, looking only at estuaries, juveniles of these migrant species are the obvious target and the gears deployed have the appropriate characteristics to capture them. Yet, on a larger scale, from the estuary to the continental shelf, the overall fishing pattern may be comparable to the harvest of a natural predator over the available size classes in an ecosystem (Figure 7). This design, which mimics nature, can be considered optimal, as fishing focuses on the most productive size classes (individuals below L_{50}) and species (LTL species) provided by each ecosystem adapting to abundance changes (Van Zwieten et al., 2011). Burgess and Plank (2020) suggest that the adaptive pattern of SSFs could have a stabilizing effect on targeted food webs for the same reasons that generalist predators increase stability in food webs (McCann and Rooney, 2009). However, this situation is sustainable only if the predator population is regulated by the size of the prey population. Thereby, this fishing pattern alone cannot protect against overfishing (Plank et al., 2017) and . The sustainability of this fishery will depend on the effort of each gear, particularly those catching HTL species.

5.5.2. Towards a BH approach

Our study described a complex pattern of a tropical SSF with coexisting fleets that conventional size limitations may not be appropriate to manage, as the yield benefits will not be obtained by the fleet which would suffer from size limitations. Therefore, an approach that deals simultaneously with the fishing pattern and the fishing effort should be adopted. On these regards, this work shows that the use of complementary gears, at the appropriate scale achieves the harvest of all sizes and all trophic levels required by BH. The BH approach is by definition designed for mixed fisheries since it recommends harvesting from as many ecosystems, species, stocks and sizes as possible (Garcia et al., 2015). This approach could therefore be compatible with the multidimensionality of tropical SSFs in terms of taxa, life cycle strategies and habitat diversity. As BH would be reconcilable with existing fishing patterns, there would be no need for time-consuming and costly gear reform or training of fishers in new fishing methods. Nevertheless, the studied fishing pattern is only sustainable if the targets are harvested according to the productivity of the ecosystem. Consequently, the BH approach, in a configuration similar to the fishing patterns reported in this study, would have to focus on a sustainable fishing effort by considering a multi-gear, multi-fleet system. This challenging task in a data-poor context could be achieved using new approaches to estimate fish biomass production at the species level that are being developed with limited data requirements (Zottoli et al., 2020).

In our study, different fishing methods were applied in the north and south coasts, but it resulted in a pattern in accordance with BH approach. The plasticity offered by BH could allow the conservation of these traditional peculiarities to be integrated, since it is the catch in the system as a whole that needs to be balanced. Besides being a profession, artisanal fishing is part of a cultural heritage with regional specificities (McGoodwin, 2001; Ritzman et al., 2018) . The preservation of traditional ways of life could promote adherence to the local community and facilitate the implementation of BH. Furthermore, the social need to harvest productive compartments such as juveniles and low trophic level species, highlighted in this study, could be addressed in the BH approach. Since catches are dominated by low-value individuals, BH aims are more related to food security than to market profits (Kolding et al., 2016a). This objective could be a more relevant approach for SSFs management since they provide the main source of protein for local communities, it would be appropriate to focus on quantity rather than quality (high-value fish or adults).

Nonetheless, it is challenging to manage a, multi-species fishing effort where it is hardly possible to match catches exactly to quotas since species are caught simultaneously by many gear types with different fishing power (Branch and Hilborn, 2008). Moreover, this effort reduction should be applied by all fleets and should have a positive impact on each of them. Still, many doubts persist as to BH practical application, in a context of insufficient data. There should be empirical tests to give concrete guidelines about how to achieve the required effort distribution among the coexisting fleets for a sustainable exploitation of these connected habitats in a poor documented SSF.

Furthermore, the complexity of the studied fishing pattern supports the need for a spatially integrated management of SSFs which would be essential for the monitoring of migratory species caught at different points along the marine continuum. Hence, their survival requires the proper functioning of these connected ecosystems (Olds et al., 2018; Sheaves, 2009). Nonetheless, BH integrates the notion of the plurality of ecosystems without clearly addressing issues of habitat quality, although its success depends on it (Garcia et al. 2015). As fishing itself is not the only problem, management restricted to fishing methods cannot be the only solution (Garcia et al. 2015). Thereby, artisanal fishing occurs in degraded habitats and this could pose a greater and more immediate threat to the sustainability of stocks than overfishing. In the case of Pernambuco, mangrove removal, plastics, industrial pollutants and, recently, oil spill have modified coastal ecosystems (Ferreira et al., 2016; Pelage et al., 2019; Araújo et al., 2020). A fisheries policy that fails in mitigating these alterations would not be effective.

5.6. Conclusion

Interactions between the spatially intricate fishing pattern of coastal-marine coexisting fleets and migratory species result in a high level of complexity. The management of these fisheries must integrate species parameters, gear complementarity and fishing ground location. Moreover, juveniles and low trophic species represent an important part of the catches for these SSF which are primordial for food safety. Single species-based measures, such as size limitations, would not adequately address technical interactions and social needs, making holistic approaches preferable. Balanced harvesting is designed for marine SSFs, as it advocates a multidimensionality in line with their current fishing patterns. It could offer economic, social, and ecological sustainability by helping to preserve the traditional fishing patterns of local communities and meet their protein needs while maintaining ecosystem structure. Nevertheless, issues related to effort regulation in a multi-gear system remain a bottleneck for mixed fisheries management, regardless of the EAF to be implemented. These concerns need to be resolved by considering gear-species interactions and habitat connectivity, either through models or, if possible, empirical data.

5.7. References

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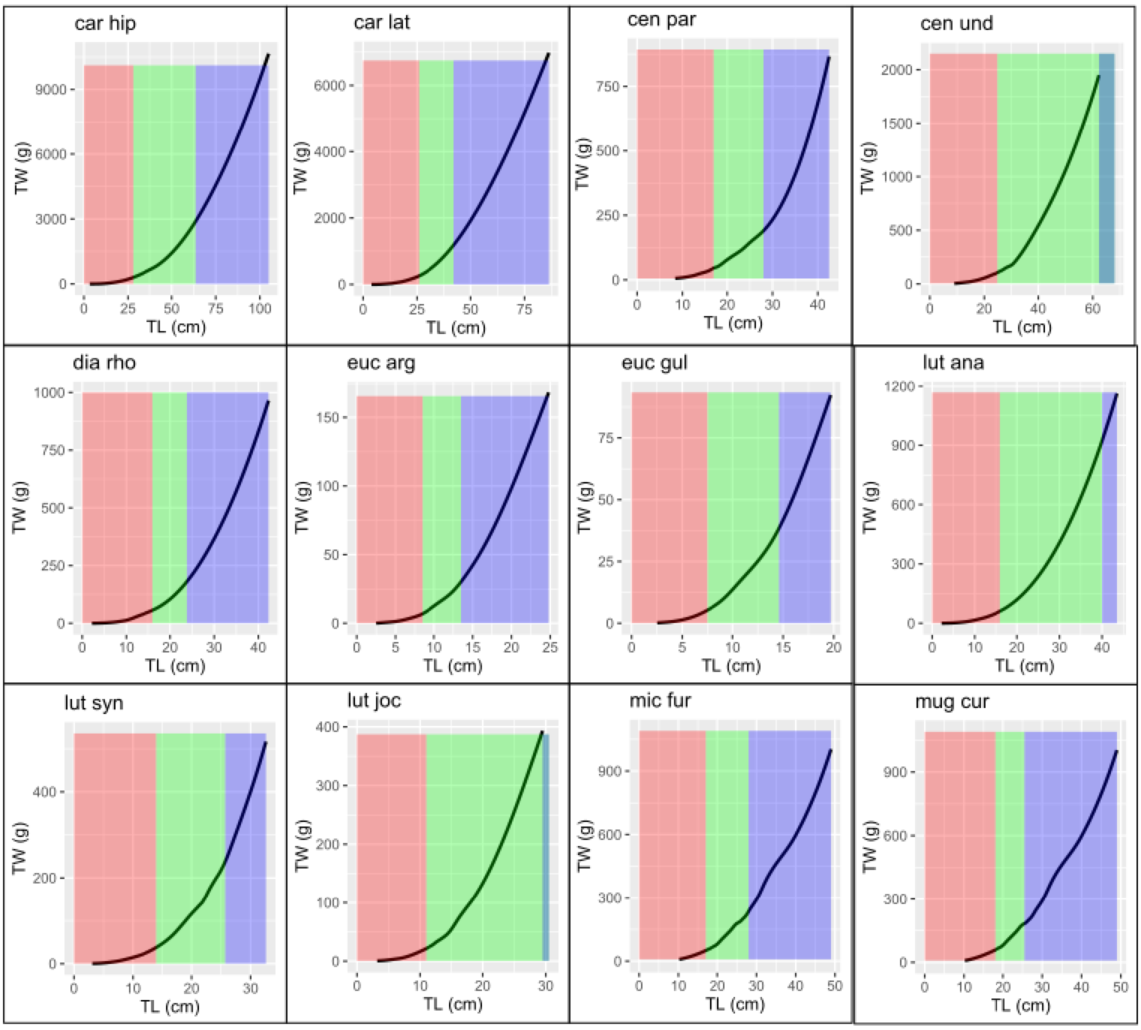
Supplementary material:



Supplementary Figure 1: Length-weight curves representing the three size classes for each species: juveniles (in pink), subadults (in green) and adults (in blue).

*Supplementary Table 1: Table with the characteristics of the species. TLmin and TLmax are the minimum and maximum sizes at which individuals of the species were found. * Trophic level found in Lira and al. L50 is the size at first maturity where TL=Total length, FL =Furcal Length and SL= Standard Length. For the references of the Guilds EUFMG (estuarine use functional group) see (Ferreira et al. 2019).*

Species	**Family**	**TLmin (cm)**	**TLmax (cm)**	**Nº**	**Lsub (cm)**	**L_{50}**	**References L_{50}**	**IUCN**	**Trophic Level**	**Category**	**Guild EUFG**
Diapterus rhombeus (Curvieur, 1829)	Gerreidae	2.2	42.3	978	16	23.8 (TL)	(Bouchon-Navaro et al., 2006)	LC	2.9*	Low trophic level	MM
Eucinostomus argenteus (Baird & Girard 1854)	Gerreidae	2.5	45.8	2494	8.5	13.5 (TL)	(Costa et al., 2018)	LC	3.1*		MM
Eucinostomus gula (Curvieur, 1830)	Gerreidae	2.6	20.7	2242	7.5	14.6 (TL)	(Bouchon-Navaro et al., 2006)	LC	3.1*		MM
Micropogonias furnieri (Desmarest, 1823)	Scianidae	10.2	49.0	149	17	28.0 (TL)	(Vazzoler, 1962)	LC	3.1		MM
Mugil Curema Valenciennes,1836	Mugilidae	10.1	54.0	2162	18	25.5 (TL)	(Oliveira, dos Santos Costa, and Chellappa 2011)	DD	2.0		MM
Caranx Hippos (Linnaeus, 1766)	Carangidae	4.3	105.0	256	30	63.6 (FL)	(Caiafa et al., 2011)	LC	4.0.*	High Trophic level	MS
Caranx Latus Agassiz,1831	Carangidae	3.9	88.0	146	26	42.1 (TL)	(Bouchon-Navaro et al., 2006)	LC	4.2		MS
Lutjanus analis (Cuvier,1828)	Lutjanidae	2.4	90.0	815	16	40.0 (SL)	(Freitas et al., 2011)	NT	3.9		MS
Lutjanus jocu (Bloch &Schneider, 1801)	Lutjanidae	3.2	100.0	349	11	30.5 (SL)	(Freitas et al., 2011)	NT	4.4		MS
Lutjanus synagris (Linnaeus, 1758)	Lutjanidae	3.1	48.0	265	14	25.7 (TL)	(de Moura Cavalcante et al., 2012)	NT	3.8		MS
Centropomus parallelus (Poey,1860)	Centropomidae	8.4	104.0	276	17	28.0 (TL)	(Rodrigues 2005)	LC	4.2		MM
Centropomus undecimalis (Bloch, 1792)	Centropomidae	9.0	85.0	318	25	68.0 (FL)	(Perera-García et al., 2008)	LC	4.2		MM

Supplementary Table 2: Table summarizing the output of the Kruskal Wallis tests.

Response variable	Explaining variable	pvalue	Chi squared	df
Ontogenetic category	Trophic Level	< 2.2e-16	1007.2	9
	L_{50}	< 2.2e-16	1386.4	11
	Gear	< 2.2e-16	878.93	8
	Habitat	< 2.2e-16	1140.6	4
	Zone	0.03115	0.01015	1

Supplementary Table 3: Table summarizing the output of the Kruskal Wallis tests

Response variable	**Explaining variable**	**pvalue**	**Chi squared**	**df**
Size category	Trophic Level	< 2.2e-16	1352.4	9
	L_{50}	< 2.2e-16	1529.6	11
	Gear	< 2.2e-16	786.81	6
	Habitat	< 2.2e-16	854.58	3
	Zone	< 2.2e-16	0.01015	1

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6. Considerações finais

Este estudo caracterizou alguns impactos antrópicos sobre os habitats costeiros e marinhos de Pernambuco. Uma abordagem holística foi utilizada considerando tanto a modificação da paisagem costeira resultante da implementação de atividades antropogênicas na costa (aquicultura, urbanização, atividades portuárias) quanto os impactos relacionados à coexistência de frotas que exploram esta sucessão de habitats marinhos conectados.

Assim, as atividades estabelecidas na costa de Pernambuco afetam os ambientes costeiros de maneira muito complexa. A aquicultura, a agricultura, a urbanização e a construção de portos mudaram a paisagem marinha através da remoção de habitat como o manguezal. Adicionalmente elas tiveram impactos complexos no funcionamento dos ecossistemas ao alterar variáveis ambientais que podem influenciar os fluxos entre os habitats conectados e induzir salinização. Embora a salinização, induzida pela construção de portos, barragens e elevação do nível do mar (mudança climática global), tenha tido um efeito positivo sobre os manguezais que substituíram a vegetação adaptada ao meio ambiente por água doce, ela poderia ter um efeito adverso sobre a função de berçário para espécies sensíveis à salinidades elevadas. Além disso, as alterações geomorfológicas causadas por essas atividades são susceptíveis de afetar drasticamente o funcionamento dos ecossistemas, pois influenciam os caminhos tróficos e os fluxos de energia entre os habitats adjacentes. Como as mudanças geomorfológicas sofridas pelos estuários alteram as variáveis ambientais, elas também podem ter um impacto direto na produtividade pesqueira, já que estão intimamente inter-relacionadas. Desta, maneira, estas alterações poderiam afetar os estoques pesqueiros e acarretar consequências socioeconômicas. Esse estudo, assim como aqueles que têm observado a degradação do habitat através da liberação de contaminantes e resíduos em Pernambuco ressalta a necessidade de conter a expansão destas atividades.

Além disso, o funcionamento adequado desses habitats é essencial para manter o rendimento das pescarias operando no gradiente estuário-plataforma continental. De fato, as pescarias da região nordeste são caracterizadas por diversas frotas que exploram vários habitats ao longo de um gradiente estuário-plataforma. Como muitas das espécies alvo se movem ao longo deste gradiente durante seu ciclo de vida, os rendimentos destas diferentes pescarias são codependentes. Consequentemente, há uma necessidade de um manejo integrado destas frotas coexistentes explorando o mesmo recurso. Nossos resultados sugerem que métodos baseados em espécies como limitações de tamanho não seriam aplicáveis a estas frotas coexistentes, pois os benefícios de rendimento não serão obtidos pela frota que sofreria de limitações de tamanho. Uma abordagem baseada em ecossistemas como o “Balanced harvest”, consistente com os padrões de pesca existentes, poderia ser uma alternativa interessante para o manejo dessas frotas. Entretanto, deveria haver testes empíricos para desenvolver diretrizes concretas sobre como alcançar a distribuição de esforço necessária entre as frotas coexistentes para uma exploração sustentável destes habitats conectados.

Assim, a complexidade do padrão de pesca estudado destaca a necessidade de uma gestão espacialmente integrada dessas pescarias, que seria essencial para o monitoramento das espécies migratórias capturadas em diferentes pontos ao longo do continuum marinho. Além disso, o rendimento dessas pescarias requer o bom funcionamento destes ecossistemas conectados. No entanto, o *Balanced harvest* integra a noção de pluralidade de ecossistemas sem abordar claramente questões de qualidade de habitat, embora seu sucesso dependa dela. Como a pesca em si não é o único problema, o manejo restrito aos impactos da pesca não pode ser a única solução. Deste modo, o fato da pesca artesanal ocorrer em habitats degradados poderia representar uma ameaça maior e mais imediata à sustentabilidade dos estoques do que a pesca excessiva. No caso de Pernambuco, a remoção de manguezais, a poluição por plásticos, e contaminação por resíduos industriais e, recentemente, o derramamento de petróleo modificaram os ecossistemas costeiros. Uma política de pesca que falhasse na mitigação dessas alterações não seria eficaz. Contudo, a tentativa de enfraquecer políticas de proteção de áreas de preservação, entre elas áreas de manguezais, como a implementação da Resolução CONAMA nº 500, de 19 de outubro de 2020 (Suspensa pelo Supremo Tribunal Federal), que revogava a resolução CONAMA nº 303, de 20 de março de 2002 coloca em risco um habitat essencial para as pescarias estuarinas e costeiras. Efetivamente, o rendimento destas pescarias depende da função de berçário, alimentação e reprodução dos manguezais.

Um planejamento espacial marinho seria então fundamental para a gestão dessas pescarias. Este planejamento deve integrar todas as atividades que possam causar danos aos habitats costeiros e marinhos. Como estes habitats estão interligados, o planejamento deve cobrir uma variedade de ecossistemas, tanto terrestres como marinhos. O conjunto de medidas a serem implementadas deve ser o resultado da colaboração entre os diversos usuários e resultar de diálogos que permitam concessões que não prejudiquem os diversos atores. Deve, portanto, adotar uma visão global de sistemas socioecológicos complexos, levando em conta seus componentes ambientais e econômicos de forma equilibrada.

Para finalizar, são apresentadas algumas recomendações visando a desenvolver um futuro planejamento espacial marinho para a região de estudo:

- Integração de todos os tipos de uso do solo e atividades antrópicas desenvolvidas em habitats interligados para compreender possíveis conflitos espaciais e criar espaços de diálogo entre usuários com vistas à construção coletiva de estratégias de gestão.

- Fortalecimento de políticas visando a evitar a expansão espacial das atividades que causam o desmatamento de mangues.

- Desenvolvimento de estudos sobre os efeitos diretos e indiretos de vários impactos sobre os compartimentos bióticos e abióticos dos habitats marinhos e costeiros (salinização induzido pelas atividades antrópicas, poluição...) e quantificação do custo econômico desses impactos.

- Realização de estudos preliminares multidisciplinares antes de implementar quaisquer mudanças na paisagem marítima que alterem a conectividade entre habitats, em particular no caso de habitats de importância significativa para as espécies migratórias.

- Ações de engenharia costeira na reestruturação de conexões em ambientes selecionados.

- Identificação de habitats importantes dos quais as espécies ameaçadas dependem para sua estratégia de vida, a fim de implementar medidas de conservação. A extensão e o tipo de habitats a serem protegidos dependeriam da região e também deveriam fazer parte dos trade-offs entre usuários, como em qualquer abordagem convencional de planejamento espacial marinho.

- Estudos integrados sobre o ciclo de vida das espécies e sua relação espaço-temporal

- Desenvolvimento de estudos envolvendo modelos multi-frotas multi-espécies que podem integrar simulações econômicas para entender como implementar concretamente o *Balanced harvest* e pescar de acordo com as produtividades das espécies alvo.

- Aplicação prática do “Balanced harvest” para investigar a possibilidade de uma redução de esforço aplicada por todas as frotas e que deve ter um impacto positivo em cada uma delas.

- Utilização de ferramentas analíticas para integrar aspectos socioeconômicos na pesca quantitativa afim de considerar todas as dimensões da sustentabilidade no desenvolvimento das medidas de gestão da pesca com a colaboração dos pescadores das diferentes frotas.

− Implementação de um sistema de monitoramento para avaliação regular do desempenho das medidas de gestão aplicadas e do rendimento das pescarias.

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8. Annexo

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Coastal Land Use in Northeast Brazil: Mangrove Coverage Evolution Over Three Decades

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Abstract

It is essential to monitor both the mangrove coverage and the encroachment of anthropogenic activities to assess the evolution of these highly valuable and threatened ecosystems. Using the grid technique, Landsat images of three estuaries in Pernambuco, northeast Brazil, were analyzed to track changes in land use over the past three decades. This study is the first time the grid technique has been used as a precise method to quantify and localize the mangrove coverage changes at local scale. We found that the technique was a synthetical and cost-effective way of observing land-use changes over the study period, and its precision was evident for local-scale study. An increasing trend of mangrove coverage was observed and can be related to the salinization. This increase in salinization is a result of anthropogenic activities and climate change. We also found that differences in geometrical properties influenced the spatiotemporal patterns of mangroves. To mitigate adequately the negative impacts induced by anthropization and climate change, the characteristics of mangrove forest configuration and human activities should therefore both be considered.

Keywords

mangrove, Landsat, grid technique, anthropic threats, land use, remote sensing

Introduction

Mangroves are of great socioeconomic and environmental importance globally. They provide raw materials and food and contribute greatly to coastal protection, erosion control, water purification, carbon sequestration, tourism, recreation, and education (Barbier et al., 2011). They also contribute to the maintenance of fisheries by providing nursery habitats and increasing marine productivity. This makes them of central importance to local culture and traditional knowledge (Walters et al., 2008). These forests of salt-tolerant species allow the development of complex food webs (Valiela, Bowen, & York, 2001), and because mangrove ecosystems carry out essential ecological functions, any significant loss of mangrove forests will have significant consequences on these food webs as well as on biodiversity and abundance.

Yet, mangrove wetlands are being threatened by the growth of anthropogenic activities. Moreover, climate change, the increase of CO_2 emissions, rising sea levels, and alterations in precipitation regimes are having a great impact on these ecosystems (McKee, Rogers, & Saintilan, 2012; Ward & Larcombe, 2010). Indeed, more than one third of the world mangrove disappeared in the past 60 years, and since the beginning of the 21st century, mangroves continue to be lost at a rate of about 0.2% of their total area per annum (Hamilton & Casey, 2016). Despite these threats, only 6.9% of mangroves worldwide come under any form of protected area legislation (Giri et al., 2011). Mangrove forest management is thus a great challenge for biodiversity preservation, particularly in the estuaries where human activities are encroaching (Alongi, 2002;

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De Souza-Machado, Kloas, Toffolon, Zarfl, & Spencer, 2016; Kennish, 2002).

Mangrove vegetation grows in a relatively narrow fringe between the land and the sea, between latitudes 25°N and 30°S (Valiela et al., 2001).The fact that mangroves are mostly found in developing regions of Asia, Africa, and South America adds additional challenges to their monitoring (Li, Mao, Shen, Liu, & Wei, 2013; R. O. Santos, Lirman, & Serafy, 2011). There is also a lack of precision over the definition of mangrove forests, and this contributes to a lack of clarity regarding the state of mangrove forests globally. For example, mangroves are often defined as intertidal communities of trees, but this can sometimes also refer to muddy sediments and other herbaceous vegetation. Therefore, it is impossible to establish whether the worldwide extent of mangrove ecosystems is 100,000 km^2 or 200,000 km^2 (Wilkie & Fortune, 2003).

The Brazilian coast has the second largest mangrove area in the world, distributed between the Oiapoque River on the French Guyana border (04°30′N) and Sonho Beach in the state of Santa Catarina (28°53′S; Kjerfve & Lacerda, 1993). There was approximatively 10,124 km^2 of mangrove in Brazil in 1983, but by 2000, the estimated area had shrunk to 9,630 km^2 (Giri et al., 2011). In the past three decades, the Brazilian mangrove has been severely impacted by harbor expansions, the development of tourism infrastructure, and the growth of agriculture and industry, especially along the northeastern and southeastern coasts (Schaeffer-Novelli, Cintrón-Molero, Soares, & De-Rosa, 2000). The destruction of mangrove areas for the installation of shrimp farms has been identified as a particularly important factor for mangrove loss in the northeast region of Brazil (Guimarães, Travassos, Souza Filho, Gonçalves, & Costa, 2010).

The Brazilian state of Pernambuco has significant mangrove coverage because its geographical position and low altitude is conducive to mangrove growth (Silva, Galvíncio, Brandão Neto, & Morais, 2015). Along the 187 km of coast, there are 15 estuaries with mangroves. This habitat is important for fauna biodiversity (e.g., fish, birds, mollusks, and crustaceans) in the area. Indeed, more advanced larval stages were observed in the mangroves in Pernambuco compared with adjacent habitats, indicating strategic ontogenic migration to this zone (Silva-Falcão, Severi, & De Araújo, 2013).

Biodiversity and fisheries are closely related, and many anthropogenic activities depend on a healthy mangrove ecosystem. There are 34 riverine communities in Pernambuco subsisting on fishing and gathering mollusks and crustaceans (Lessa, Monteiro, Duarte-Neto, & Vieira, 2009). Some 13,000 fishermen are involved in small-scale artisanal fishing (Silva et al., 2015). Collecting data describing the condition of mangrove coverage in the area should, therefore, be considered a top research priority, as an entire socioeconomic system depends on it.

To understand mangrove transformation trends, historical data on land use are required. Integrating geographic information systems (GIS) and remote sensing enables the generation of data that can accurately record landcover changes (Rawat & Kumar, 2015). Maps generated through this approach illustrate the ecological properties of the studied areas and can thus be used as tools to guide environmental managers and policy makers (Schmidt et al., 2004). Maps of anthropic activities must be used to implement concrete conservation strategies (Tulloch et al., 2015). Vulnerable areas need to be identified to enable efficient planning and prioritization of conservation measures (Groves et al., 2002). Several vegetation indices have shown the benefits of separating the detected vegetation into small grids that allow meaningful spatial and temporal comparisons of vegetation dynamics and enable research to be conducted with the precision necessary to characterize local variations (Huete et al., 2002).

The aim of this study was to provide information on the evolution of mangrove cover in three of the main estuaries of Pernambuco, over a period of three decades. These three systems have suffered from different forms of land exploitation, and we expected them to show different trends in their vegetation patterns. Our main hypothesis was that each type of landscape alteration (harbor expansion, aquaculture, urbanization, and agriculture) threatens the mangroves and has a quantifiable structural impact. The grid technique has never been applied to investigate mangrove coverage changes at a local scale, and this study is the first to test if this application is effective. It is assumed that dividing mangroves into small cells will aid with the precise identification and quantification of drastic spatiotemporal modifications.

Methods

Study Area

Three estuaries—the Santa Cruz channel, Sirinhaém, and Suape—were chosen because of the differences in their geomorphology and in the human activities being conducted within their areas (Figure 1).

The Santa Cruz Channel is a 20-km long coastal estuarine complex located in the north of Pernambuco and comprising a U-shaped area of 22 km^2. The Santa Cruz Channel circles Itamaracá Island and has two accesses to the Atlantic Ocean (Schwamborn et al., 2001). The mangrove composed by *Rhizophora mangle* (red mangle), *Avicennia shaueriana* (black mangle), and *Laguncularia racemosa* (white mangle; Medeiros, Kjerfve, Araujo, & Neumann-Leitão, 2001) is distributed all along the

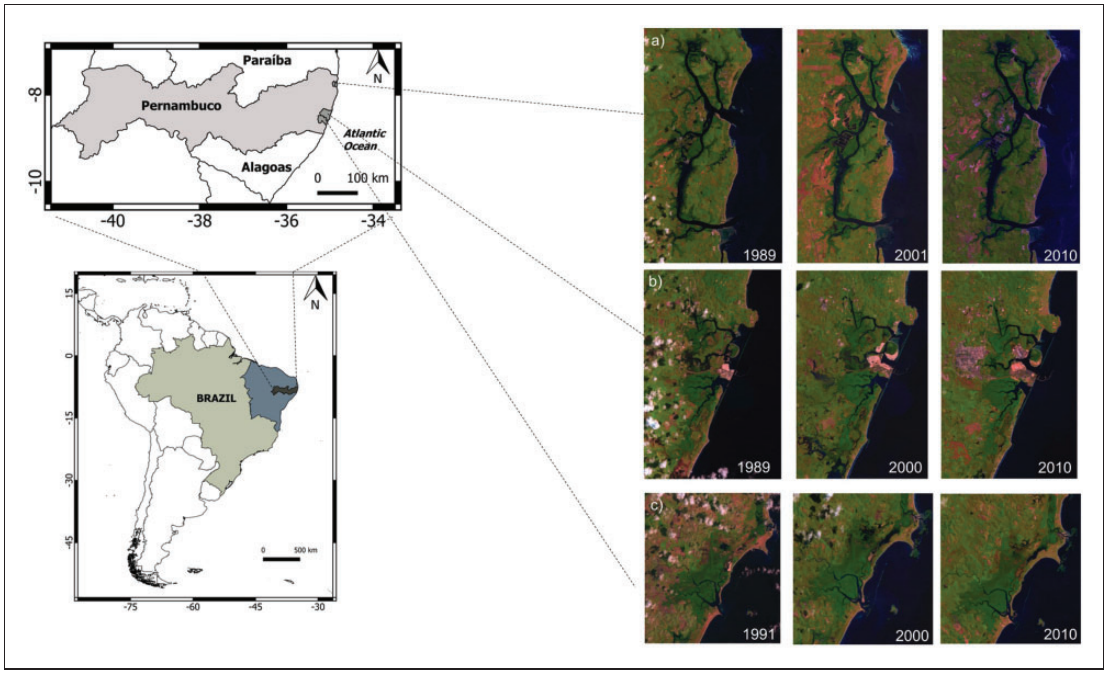

Figure 1. Location of the three estuaries of interest along the coast of Pernambuco with Landsat satellite images.

channel, particularly on its eastern borders (Gomes, Santos, Alves, Rosa-Filho, & Souza-Santos, 2002). Situated at the center of the channel, the city of Itapissuma is Pernambuco's main fishing hub and one of the region's best known tourism destinations (Quinamo, 2006). Aquaculture, artisanal fishing, industry, and agriculture are all important anthropogenic activities in the area (De Moura & Candeias, 2009).

The second estuary, in the Suape estuarine complex, has a watershed of 3,800 km^2 and is located approximately 40 km south of the city of Recife, between 8°15′–8°30′S and 34°55′–35°05′W (Koening, Leça, Neumann-Leitão, & Macêdo, 2003; Souza & Sampaio, 2001). The region has a flattened geomorphological shape, medially divided by the outcrops of the fractured volcanic masses of the cape of Santo Agostinho (M. A. Santos & Costa, 1974; Silva et al., 2015). Before the construction of a port in 1979, four rivers—the Massangana, Tatuoca, Ipojuca, and Merepe—drained into the Suape estuary, itself partly isolated from the ocean by an extensive sandstone reef line (A. P. Silva, Neumann-Leitão, Schwamborn, & Gusmão, 2004). The construction of the port altered the geomorphological and hydrodynamic conditions of these rivers (Muniz, Neto, Macêdo, & Filho, 2005). Indeed, landfills blocked the flow of both the Ipojuca and Merepe rivers, causing the retention and accumulation of water, which led to the inundation of the mangroves. The mangroves of the Suape complex are characterized by four genera belonging to three families: *R. mangle*, *L. racemosa (L.) c.f. Gaertn.*, *A. germinans*, *A. schaueriana Stapf & Leechman ex Moldenke*, and *Conocarpus erectus L*, as well as a variety of peripheral and generalist species (Silva et al., 2015).

The mangrove forest of the third study area, situated in Barra de Sirinhaém district in the southern coast of Pernambuco, is characterized by *R. mangle*, *L. racemosa*, and by species from the genera *Avicennia* (Maciel & Alves, 2009). In addition to artisanal fishing, agroindustry, services, and trade are important socioeconomic activities in the zone. A sugarcane plantation was established there in the 19th century, and Sirinhaém plays an important role in national sugar production.

GIS Analysis

To obtain the grid cells for our analysis (Figure 2), the following methods were used.

Satellite images. Satellite images of the three estuaries over the past three decades were analyzed to describe the evolution of the mangrove area. Optical images between 1989 and 2016 from the following satellites and sensors were used: Landsat 5/Thematic Mapper, Landsat 7/Enhanced Thematic Mapper Plus, Landsat 8/Operational Land Imager/Thermal Infrared Sensor (Table 1).

These images were acquired from the Landsat collection of the United States Geological Survey site (https://www.usgs.gov/). Different compositions of Red (R),

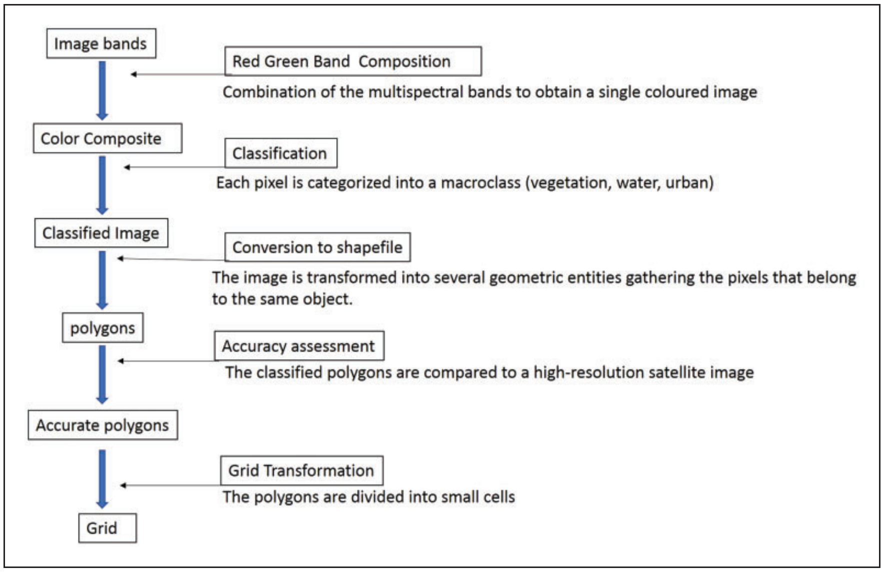

Figure 2. Flowchart of the GIS analysis.

Table 1. Characteristics of the Collected Images.

Estuary	Date	Satellite	Instrument	Orbite	Point
Santa Cruz Channel	12/09/1989	Landsat5	TM	214	65
	13/11/1994	Landsat5	TM	214	65
	04/08/2001	Landsat7	ETM+	214	65
	29/10/2006	Landsat5	TM	214	65
	08/10/2010	Landsat5	TM	214	65
	08/02/2015	Landsat8	OLI/TIRS	214	65
Suape	04/03/1989	Landsat5	TM	214	66
	14/08/1996	Landsat5	TM	214	66
	26/09/2000	Landsat5	TM	214	66
	26/08/2006	Landsat5	TM	214	66
	06/09/2010	Landsat5	TM	214	66
	20/07/2016	Landsat8	OLI/TIRS	214	66
Sirinhaém	20/10/1991	Landsat5	TM	214	66
	14/08/1996	Landsat5	TM	214	66
	26/09/2000	Landsat5	TM	214	66
	26/08/2006	Landsat5	TM	214	66
	06/09/2010	Landsat5	TM	214	66
	20/07/2016	Landsat8	OLI/TIRS	214	66

Note. TM = Thematic Mapper; ETM+ = Enhanced Thematic Mapper Plus; OLI = Operational Land Imager; TIRS = Thermal Infrared Sensor.

Green (G), and Blue (B) bands were realized to create false color images: Landsat 5 (bands 547), Landsat 7 (bands 543), and Landsat 8 (bands 654; Figure 1).

Image processing. To map different anthropogenic activities and identify features in the landscape, a semiautomatic (or supervised) image processing classification technique was applied to the satellite images which enabled the identification of landcover through its spectral signature (Congedo, 2016). Macroclasses of objects with similar spectral signatures were determined to create manual training spectral signatures to overlay pixels belonging to the same landcover class. Two algorithms were then applied to assign each pixel to a class: the minimum distance and the maximum likelihood. The minimum distance algorithm calculates the Euclidean distance between spectral signatures of image pixels and training spectral signatures and assigns, for each pixel, the closest class of spectral signature (Mather & Tso, 2016). The maximum likelihood algorithm was used to categorize a pixel using the probability distributions for the landcover classes, following Bayes' theorem (Munyati, 2004).

The supervised classification was carried out in Quantum GIS (QGIS 2016) using the Semi-Automatic Classification Plugin (Congedo, 2016). Ten macroclasses were established (water, urban area, mud bank, mangrove, vegetation, cloud, exposed soil, cloud shadow, port, and aquaculture). For each estuary, maximum likelihood and minimum distance algorithms were applied. Then, the accuracy of the classification was checked by looking at the satellite image to see if the class attributed to most of the pixels was valid. Moreover, the image classes were compared with a high-resolution satellite image from a similar date (http://www.google.com/earth/download/ge/). The algorithm that gave the most pertinent result was chosen: maximum likelihood for the Santa Cruz Channel and minimum distance for Suape and Sirinhaém. As the images were taken during different tide conditions, the mud flats that were exposed at low tide were classified as water to obtain the same pattern for high tide images.

Polygon processing. The classified raster images were converted into polygons and exported to a shapefile format. Manual cleaning was then applied using QGIS editing tools. The polygons that did not belong to the preestablished class were removed using the satellite image of the corresponding year as a visual reference. This hand editing method is often necessary after a supervised classification (Guimarães et al., 2010; Li et al., 2013).

Spatiotemporal patterns of changes in mangrove coverage. Using ThemaMap software (https://themamap.greyc.fr/), the mangrove polygons were converted into a grid to precisely and accurately identify the areas that have undergone change. The use of a grid allows changes to be monitored at a local scale and enables observations of how, when, and where these changes occurred, at a spatial resolution finer than polygons. For each year, the grid and the mangrove polygons were intersected, and a unique grid for all the years was obtained with a binary code representing the presence or absence of mangrove in each cell. The grid cells were 0.1 km by 0.1 km, which was considered to be an appropriate unit given our study scale and the resolution of the satellite image used for the polygons. The spatiotemporal patterns of coverage mangrove modifications were analyzed mapping the mangrove that was lost, gained, and remained stable over the three decades.

Anthropogenic encroachment on mangroves. The encroachment of anthropogenic activities on mangroves was investigated, and the polygons corresponding them were mapped. Anthropogenic activities considered to constitute the most direct threat to the mangrove forest were quantified: aquaculture, urbanization, the commercial port, and agriculture. The vegetation and the exposed soil polygons in a radius of 1 km around mangrove polygons were used as a proxy to represent agricultural expansion. Industries were classified as urban because their spectral signatures were similar. To determine which activity had the greatest impact on mangrove, the intersection between the mangrove area at the beginning of the study (1990) and the area of each anthropic activity at the end of the study (2015) was calculated. The polygons generated by this intersection enabled us to estimate the overlap between the anthropic activities and the initial mangrove, being able to quantify mangrove substitution.

Results

Spatiotemporal Patterns of Mangrove Coverage

Measuring the mangrove areas for each estuary in each image (Figure 3) helped assessing the spatiotemporal variations along the past three decades.

The Santa Cruz Channel mangrove area increased between 1989 and 1994 and then decreased between 1994 and 2001. A stable period is noticeable between 2006 and 2010, followed by a decrease between 2010 and 2015. An increase of the Sirinhaém mangrove coverage can be observed between 1991 and 2000. After 2000, the coverage area remained relatively stable. A sharp increase in Suape mangrove was observed between 1989 and 1996. After this, fluctuations were minor, but an increase between 2000 and 2006 can be seen, followed by a decrease between 2006 and 2010.

The mangrove areas that were lost, gained, and those which remained stable were mapped to see if the variations were localized in preferential areas (Figure 4).

The three estuaries exhibit different spatial patterns in terms of mangrove coverage changes over the three decades. Mangrove appearances and disappearances in Sirinhaém occurred in the outermost mangrove ecosystem, and mostly in the north. The appearances consist in a large patch of mangrove in the northeast and few dispersed cells, whereas the disappearances are insignificant over the three decades (see Appendix Figure A3).

By contrast, the appearances in the Santa Cruz Channel are scarce and border the stable mangrove. A relevant area of mangrove disappeared in the center of the estuary and a substantial mangrove withdrawal is observable on the north and south edges (see Appendix Figure A2).

Concerning Suape, the appearances of mangrove occurred mainly in the north and to a lesser extent in the south. A vast central area of new mangrove is noticeable, whereas significant areas of mangrove were lost on the coastal side of the estuary (see Appendix Figure A4).

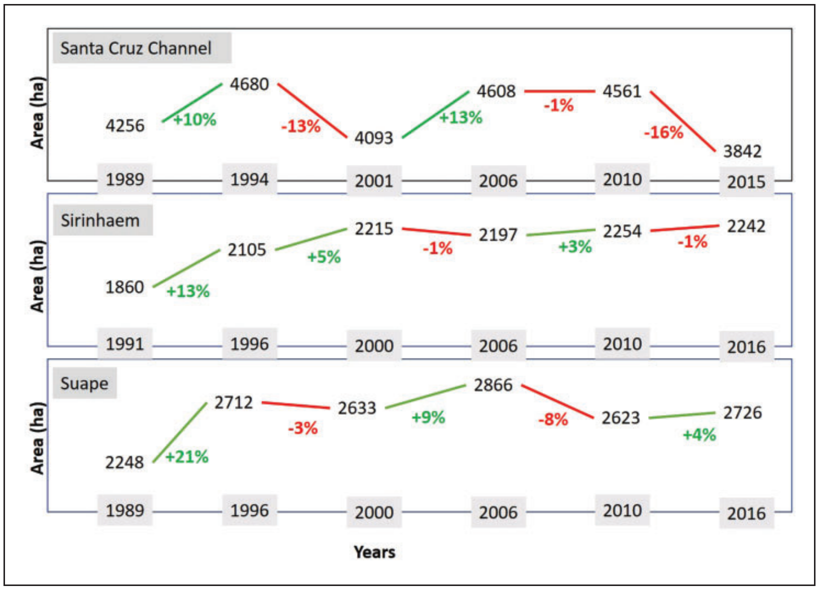

Figure 3. Slopechart of the mangrove areas of the three estuaries for each collected year and the evolution in percentage between two successive years.

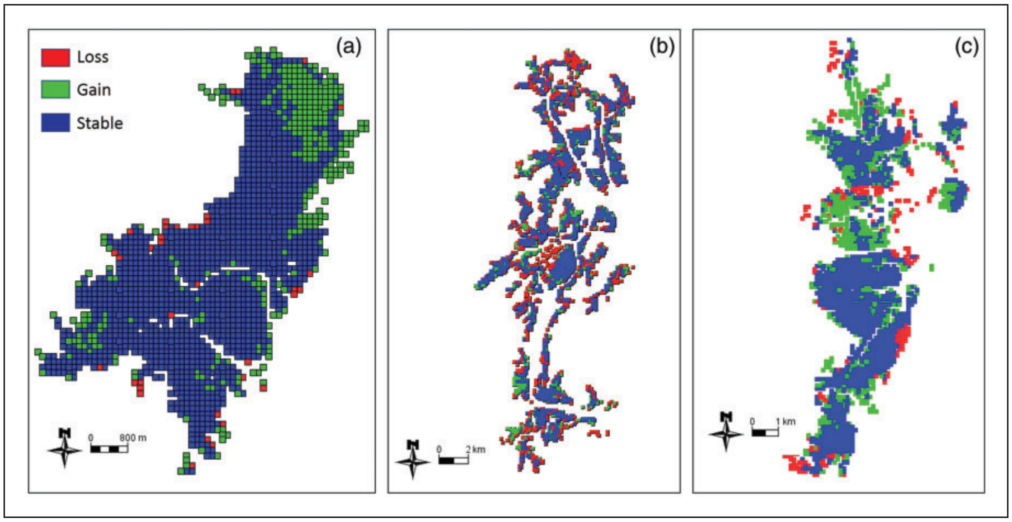

Figure 4. Maps of the mangrove areas that appeared, disappeared, and remained stable in (a) Sirinhaém, (b) the Santa Cruz Channel, and (c) Suape over the studied period.

Human Encroachment on Mangrove Forests

Maps of human activity and mangrove spatial occupation over the past three decades were produced (Figure 5; see Appendix Figure A1), enabling us to analyze the human impact on mangrove forests. Three years were chosen in each estuary to show landscape modification over time.

These images enabled a visualization of the impact of three main human activities that encroached on mangrove: aquaculture in the Santa Cruz Channel, the port in Suape, and agriculture in Sirinhaém. For the Santa Cruz Channel (Figure 5(a)), the conversion of mangrove into aquaculture has occurred in several zones distributed over the whole estuary but is more apparent in two major areas in the center. In Sirinhaém (Figure 5(b)), the entire edge of the mangrove forest is bordered by agriculture activity, leaving only a compact core of mangrove without direct exposure to this activity. In Suape, the encroachment of human activity is confined to a specific area in the central northern part of the estuary, where there has been gradual deforestation due to the construction of the industrial port (Figure 5(c)).

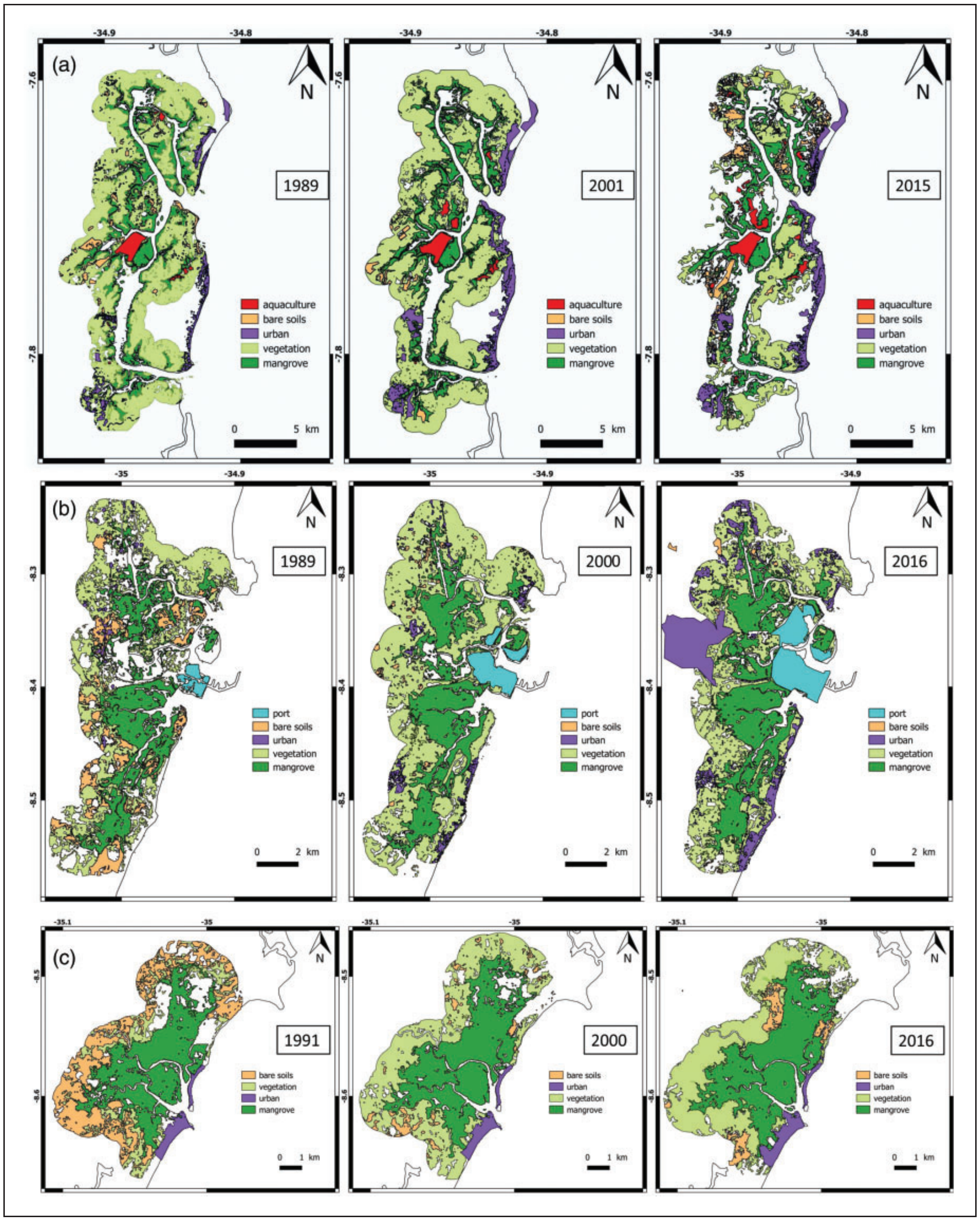

Figure 5. Evolution of mangrove areas and anthropic activities spatial occupation over the past three decades in (a) the Santa Cruz Channel, (b) Suape, and (c) Sirinhaém.

However, even if the other human activities did not replace the mangrove, a general trend of anthropization is observable in the three estuaries and is more pronounced in Suape and the Santa Cruz Channel. Indeed, the urban area has increased significantly in both estuaries during these years, and the Suape industrial complex, situated in the west of the mangrove forest, represents the anthropic activity which extended the most over the studied period.

The areas of the different land uses were measured over the three decades in the three estuaries (Table 2) to quantify the evolution of both the human activities and the mangrove coverage. It appears that the only estuary that suffered a mangrove reduction is the Santa Cruz channel with a 10% decrease of its mangrove area, whereas in both Suape and Sirinhaém, a 20% mangrove coverage increase occurred. An expansion of the human activities settled in the three estuaries is noticeable.

The aquaculture area, in the Santa Cruz Channel, increased by 92% over the three decades, with the expansion beginning most noticeably after 1994.

Table 2. Areas in Hectares of Each Land Use for the Three Estuaries, Their Total Evolution Over the Three Decades, and the Proportion of Mangrove That Was Removed for the Benefit of Each Land Use.

Estuary	Land use	1990	2000	2015	Total change ha	Total change %	Mangrove removal (%)
Santa Cruz Channel	Aquaculture	345	474	662	+317	+92	3
	Bare soils	812	279	2,486	+1,674	+77	0
	Mangrove	4,256	4,093	3,842	−414	−10	—
	Urban	4,690	7,413	7,432	+2,742	+58	0
	Other vegetation	12,276	13,796	5,399	−6,877	−56	0
Sirinhaém	Bare soils	803	212	249	−554	−69	1
	Mangrove	1,860	2,215	2,241	+381	+20	—
	Urban	145	145	199	+54	+37	0
	Other vegetation	1,323	275	2,091	+768	+58	2
Suape	Port	118	355	683	+565	+479	2
	Bare soils	489	153	47	−442	−90	0
	Mangrove	2,248	2,633	2,726	+478	+21	–
	Urban	79	299	1,220	+1,141	+1,454	0
	Other vegetation	2,916	4,576	3,521	+605	+21	0

This activity had the greatest impact on mangrove in the estuarine complex, spreading over 3% of the initial mangrove. The area dedicated to aquaculture increased sharply after 1994 but has remained stable since 2010, while the mangrove coverage gradually declined (Table 2). Furthermore, an increase of the urban area by 58%, in this estuary, was accompanied by a decrease in vegetation and an increase of bare soils.

Meanwhile, in Sirinhaém, 3% of the mangrove was removed for the benefit of agriculture (2% by other vegetation and 1% by bare soils). Thus, the total area of exposed soils underwent an abrupt reduction after 1991 and was replaced by vegetation, while the total mangrove coverage increased. After this period, the areas of exposed soil, vegetation, and mangrove remained relatively stable.

During the period of the study, the port in Suape substituted 2% of the initial mangrove. The most significant expansion of the port area occurred between 1996 and 2000, increasing from 159 ha to 355 ha along with a reduction of the mangrove. Meanwhile, the urban/industrial area sharply raised by 1494%, becoming the most significant human activity in the area. Nevertheless, even if Suape was the estuary where the anthropization was the most striking, the mangrove area still increased 21% from 2,248 ha to 2,726 ha.

Discussion

Protection measures for a mangrove area can only be pertinent if it is known how, where, and when mangrove coverage has changed. This study is the first application of the grid technique for local-scale monitoring of mangrove, and it was effective for synthesizing changes in mangrove cover which can help implement conservation strategies. It is a parsimonious way to map the spatio-temporal patterns as they can be seen in two maps, one for the appearances and one for the disappearances. Unlike the gain–loss maps generated through the polygons overlapping technique, the grid technique enables to assess the modification rates for the overall period. Moreover, grids are more precise than polygons as the unit of study consists in smaller cells. They are therefore more adaptable for local-scale monitoring. The ability to alter cell sizes also allows a good way to see modifications at different scales, which can then be attributed to processes operating at different scales. Further research is needed to test whether or not this technique can effectively determine scales of threats.

The grid technique is appropriate for spatially representing landscape processes and has previously been used for integrated river basin models (Rathjens, Oppelt, Bosch, Arnold, & Volk, 2015). The precision of the technique, coupled with in situ data, allows to model several spatial aspects of animal or vegetation population structure (Royle, Fuller, & Sutherland, 2018). However, few studies are using this approach.

On another level, the distribution of different biotic elements is key to establishing connectivity, which is an omnipresent theme in current studies (Olds et al., 2017). The grid technique could therefore bring powerful insights to seascape ecology as the hypothesis of coastal habitats being separate is giving way to the concept of an interconnected mosaic of habitats (Nagelkerken, Sheaves, Baker, & Connolly, 2015).

Mangrove coverage in the three estuaries fluctuated noticeably over the three decades of the study. An overall increase can be noted in Suape and Sirinhaém, whereas after a similar initial increase, the Santa Cruz channel mangrove has decreased during the past 5 years. Mangroves in all three estuaries are strongly influenced by salinization, and the phases of mangrove increase can be linked to this. Salinization can stimulate mangrove growth by enabling its development in areas that were previously colonized only by freshwater plants (Teh et al., 2008). Wetland salinization is a process occurring at a global scale (Herbert et al., 2015; Nachshon, Ireson, Van Der Kamp, Davies, & Wheater, 2014; Williams, 2001) and is related to climate change (Meiggs & Taillefert, 2011). During droughts, alterations in soils properties enable mangrove landward expansion (Rogers, Wilton, & Saintilan, 2006). Sea level rise also causes salt water intrusion into the estuaries (Church & White, 2006). In regions close to our study sites, such as the Metropolitan region of Recife, a 5.6-mm/year increase in sea level was observed between 1946 and 1988 (Neves & Muehe, 1995), which was above the mean world annual sea level rise at that time (Hay, Morrow, Kopp, & Mitrovica, 2015).

Furthermore, an increase in salinity can also be directly linked to anthropogenic activities (Cañedo-Argüelles et al., 2013). This secondary salinization occurred in the Santa Cruz Channel between 1989 and 1994 due to extensive damming leading to the spread of mangroves (Lacerda & Marins, 2002). Another example of human-induced salinization is the establishment of the port in Suape, which altered the geomorphological and hydrodynamic conditions of the area, particularly in the Ipojuca River estuary (Koening et al., 2003). Muniz et al. (2005) sampled parameters including salinity, pH, dissolved oxygen, transparency, and temperature from three fixed stations in this estuary. They compared summer and winter before (1978) and after (1986–1987) the port's construction, to analyze the effects of port-induced changes. A general increase of all the chemical parameters was observed postconstruction, with an increase in salinity having the greatest ecological impact. This increase is due to the Ipojuca river being diverted toward the ocean, causing a reduction in river-flow speed, sedimentation, and a diminution in the circulation and depth. As a result, the mouth of the river has been transformed into a coastal lagoon where evaporation and consequently salinity were high. This increase in salinity is likely to have favored mangrove development and enhanced the appearance of a recovery zone (Souza & Sampaio, 2001).

Because spatial patterns in each estuary are different, conservation measures must also take into account specific localized conditions within each of the three zones. Differences in mangrove dynamics between the estuaries can be explained by differences in anthropogenic activities and by the morphological shapes of the estuaries.

The Santa Cruz Channel is the only estuary that has suffered a loss in mangroves over the three decades. It is thus important to understand the evolution of land use in this estuary. The mapping anthropogenic activities demonstrated that aquaculture between 1994 and 2001 was significant in the center of the estuary. It adversely affected mangroves not just through direct encroachment leading to deforestation but also by induced physicochemical changes (Naylor et al., 2000). Toxic effluent and the alteration of sediment rates are two possible reasons for mangrove disappearance in zones that were not directly exposed to aquaculture.

The mangrove diminution observed in the Santa Cruz Channel between 2010 and 2015 could be attributed to pollution. Mercury contamination was already registered in the area as a consequence of the effluent from a chlorine and caustic soda plant which settled in 1963 and released between 22 and 35 tons of mercury by 1987 (Meyer, 1996). However, high mercury concentrations were encountered recently in the sediment of the Santa Cruz Channel (De Moura & Candeias, 2009). The values were higher than those found in 1981 by the Companhia de Tecnologia de Saneamento Ambiental. Those results indicate recent inputs or a recent movement of the accumulated mercury toward the Santa Cruz Channel. As mercury is accumulated in sediment, its transport depends on environmental factors. As a consequence, the aquatic system can remain contaminated for many years after any mercury input in the area (De Moura & Candeias, 2009). Yet, mercury can prevent the survival of mangrove seedlings (Huang & Wang, 2010) and can also affect mangrove growth by causing a depletion of nutrients, as high concentrations threaten the microorganisms that guarantee the biological conversion of minerals (Ravikumar et al., 2007).

The port construction is the main driver in the spatial dynamic of the Suape mangroves, whether through direct deforestation, salinization, and sedimentation that would explain the fast recovery of the mangrove. The other drastic anthropic landscape modification during the studied period is the growth of urban areas and the concomitant removal of other vegetation. Indeed, urban area is currently the predominant land use around the mangrove. This constitutes an additional chemical threat, as more and more urban effluent will be discharged near the mangroves. A recovery area in the center and the instability of the coastal zone were observed, emphasizing that the modifications were localized. This configuration enables priority zones for protective measures to be determined.

Mangrove forest shapes could also, in part, explain the differences observed in mangrove spatiotemporal coverage patterns as the stability of a forest, and its susceptibility to external pressures, depends on its geometrical features (Hessburg, Agee, & Franklin, 2005).

Mangrove appearances and disappearances in the Santa Cruz Channel were distributed all along the estuary and formed several small aggregates. This is possibly due to its geometrical properties, which presented as elongated shapes that could increase its vulnerability to salinization, resulting in the appearance and disappearance of areas suitable for mangrove growth. This highlights a lack of stability in this zone and suggests that the protection of targeted zones might be inappropriate: Conservation measures need to be implemented at the estuary scale.

Mangrove stability in Sirinhaém could also be related to mangrove structural characteristics. Patch size and proximity to mangrove cells influence mangroves' ability to resist habitat loss because where the core forest is sizable, the stand as a whole is less vulnerable (Harper, Steininger, Tucker, Juhn, & Hawkins, 2007). Hence, the important patch size of Sirinhaém mangrove can confer a resistance to anthropogenic threats because the main part of the forest is not exposed–unlike in a more parceled landscape. The compactness of Sirinhaém's mangroves thus explains their overall growth, despite surrounding threats, as the survival of the vegetation is a function of the density of the initial forest (Kéfi et al., 2007).

Implications for Conservation

In this study, the spatiotemporal patterns of changes in the mangrove cover were mostly determined by anthropic activity. Aquaculture, agriculture, urbanization, and harbor construction caused deforestation, released toxic effluents inhibiting mangrove growth, and induced salinization enhancing mangrove development. Despite of all these threats, our results showed a trend of mangrove growth in two of the three estuaries over the past three decades. Therefore, it seems that their resilience to disturbance was linked to their geometrical properties and to a favorable change of salinity conditions.

Indeed, salinization, mostly induced by port construction, damming, and sea-rise level, had a positive effect on mangrove. Nevertheless, it could endanger the nursery function of mangrove because freshwater or estuarine fish can be adversely affected by increases in salinity. This physiological stress could force them to migrate toward less saline areas (Nielsen, Brock, Rees, & Baldwin, 2003). Salinization could also constitute an ecological threat, decreasing inorganic processes of nitrogen removal, reducing carbon storage, and increasing the release of toxic sulfides which impact nutrient cycles and climate regulation (Herbert et al., 2015). Because the secondary salinization that occurred in our study area could have an impact at individual, population, community, and ecosystem levels (Cañedo-Argüelles et al., 2013), mitigation measures should be contemplated.

Moreover, before the implementation of any conservation measures or, indeed, any further anthropogenic activities, the geometrical properties of the mangrove should be considered to evaluate future impacts. Although our study among others discussed the importance of geometrical features, few works on wetlands have focused on their implications for conservation planning to date (Boström, Pittman, Simenstad, & Kneib, 2011). Mangrove spatial configurations (patch size and patch dispersion) can also determine faunal distribution (Pittman, McAlpine, & Pittman, 2004); therefore, alterations to those features could impact biodiversity.

As these habitats are important for artisanal fisheries, our results can be used to elaborate concrete conservation strategies. Our mapping of the landscape modifications can indeed be useful because the anthropization had a relevant impact on the mangrove patterns. Aquaculture, agriculture, and port construction were established as three anthropogenic modifications that directly threaten the mangrove. The expansion of the activities that do not encroach on the mangrove was quantified, and in Suape, the urbanization was identified as a potential threat. Avoiding the spatial expansion of these activities should be a priority.

The grid technique was an effective method to investigate mangrove coverage changes at a local scale. Its precision helped to follow adequately spatiotemporal cover changes and relate them to anthropogenic activities. Hence, this method appears to be useful for conservation planning and future remote sensing work at local scale.

Appendix

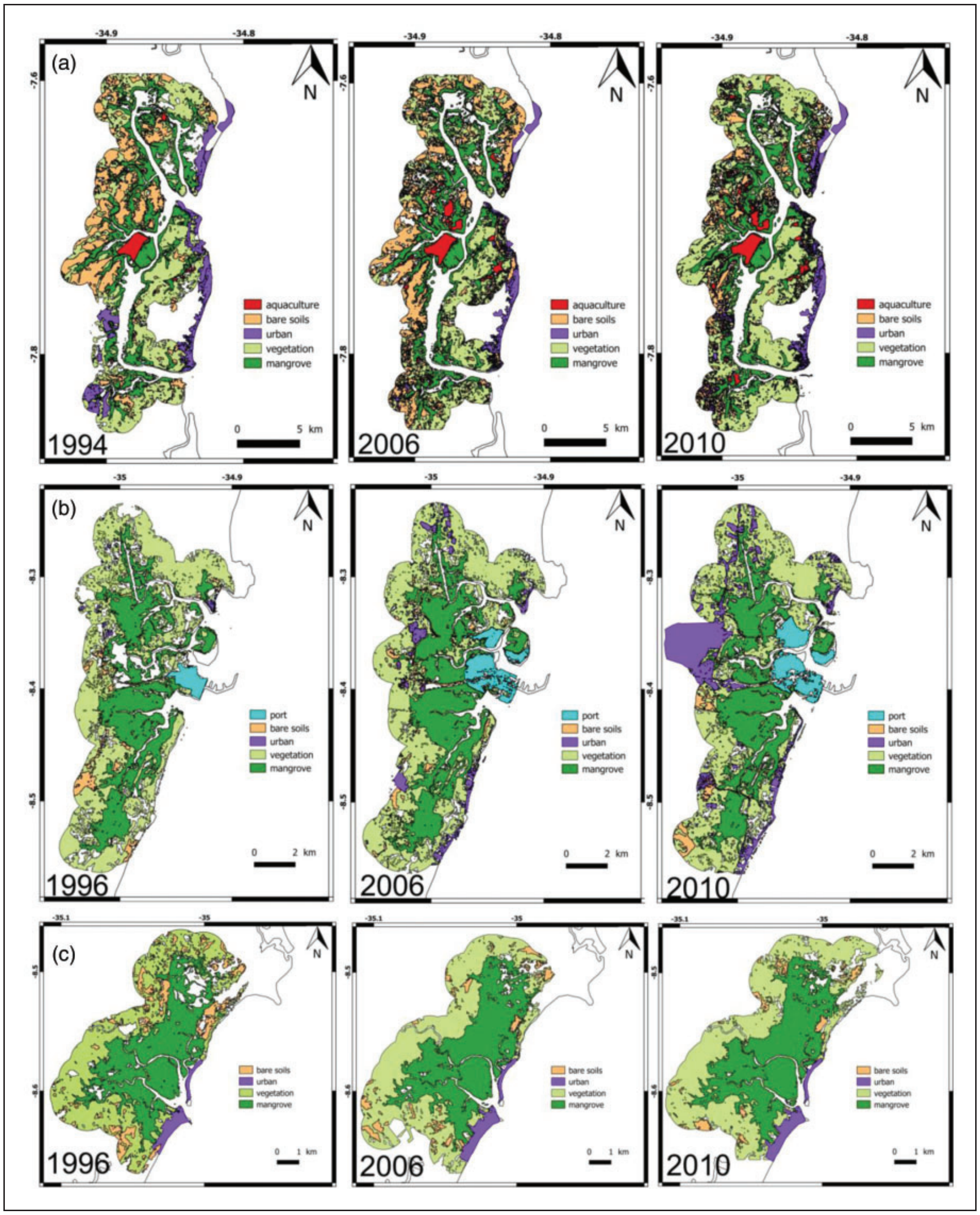

Figure A1. Evolution of mangrove area and anthropic activities spatial occupation for the images that were not presented in the results in (a) the Santa Cruz Channel, (b) Suape, and (c) Sirinhaém.

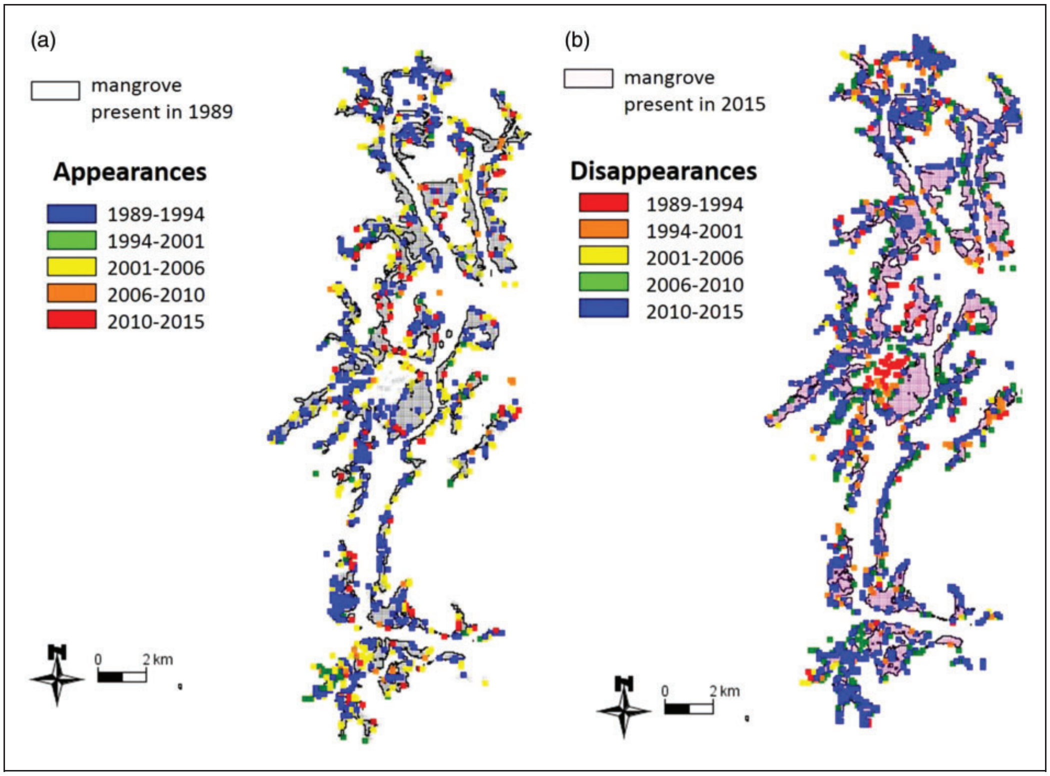

Figure A2. Maps of the mangrove areas that (a) appears and (b) disappears for each period in the Santa Cruz Channel.

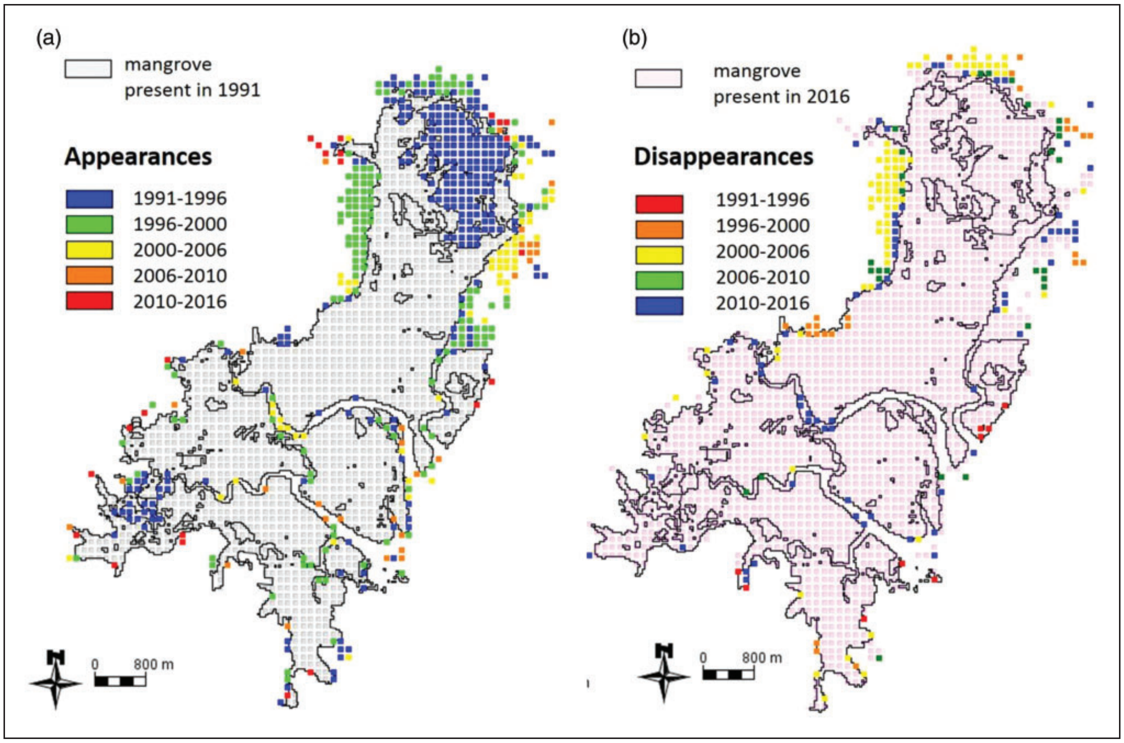

Figure A3. Maps of the mangrove areas that (a) appears and (b) disappears for each period in Sirinhaém.

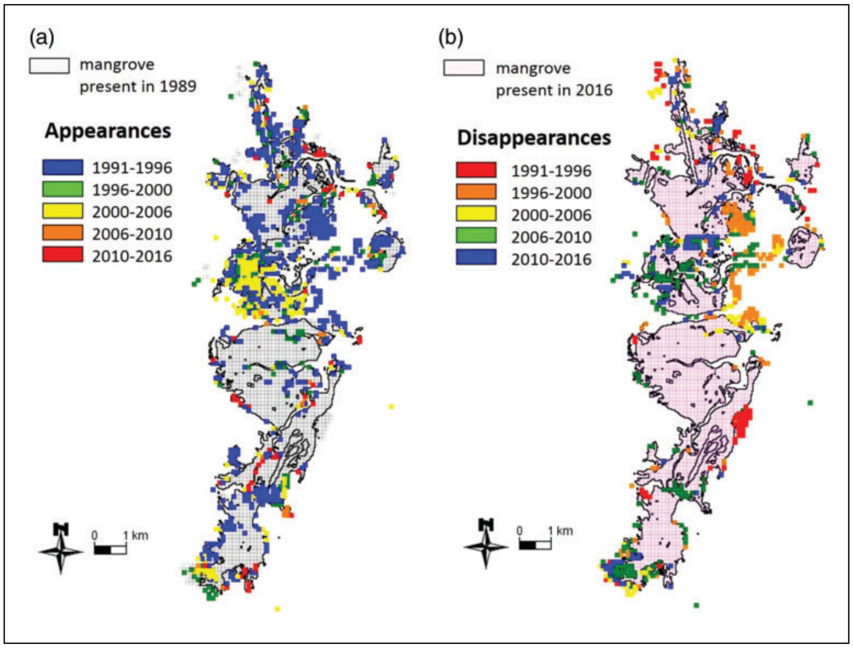

Figure A4. Maps of the mangrove areas that (a) appears and (b) disappears for each period in Suape.

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