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ASPECTOS DA BIODIVERSIDADE ÍCTIA E VULNERABILIDADE DE ESTUÁRIOS TROPICAIS EM PERNAMBUCO, BRASIL

RECIFE,
2017

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

ASPECTOS DA BIODIVERSIDADE ÍCTIA E VULNERABILIDADE DE ESTUÁRIOS TROPICAIS EM PERNAMBUCO, BRASIL

Carlos Antonio Beserra da Silva Júnior

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.

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Recife,
Agosto/2017

Dados Internacionais de Catalogação na Publicação (CIP)
Sistema Integrado de Bibliotecas da UFRPE
Nome da Biblioteca, Recife-PE, Brasil

S586a Silva Júnior, Carlos Antonio Bezerra da

Aspectos da biodiversidade íctia e vulnerabilidade de estuários tropicais em Pernambuco, Brasil / Carlos Antonio Bezerra da Silva Júnior. – 2017.

171 f. : il.

Orientador: Thierry Frédou.

Coorientadora: Bastien Mérigot

Tese (Doutorado) – Universidade Federal Rural de Pernambuco, Programa de Pós-Graduação em Recursos Pesqueiros e Aquicultura, Recife, BR-PE, 2017.

Inclui referências e anexo(s).

1. Produtividade 2. Biodiversidade 3. Ecossistema estuarino 4. Pesca artesanal 5. Monitoramento I. Frédou, Thierry, orient. II. Mérigot, Bastien, coorient. III. Título

CDD 639.3

UNIVERSIDADE FEDERAL RURAL DE PERNAMBUCO
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ASPECTOS DA BIODIVERSIDADE E VULNERABILIDADE DE ESTUÁRIOS TROPICAIS DO ESTADO DE PERNAMBUCO

Carlos Antonio Beserra da Silva Júnior

Tese julgada adequada para obtenção do título de doutor em Recursos Pesqueiros e Aquicultura. Defendida e aprovada em 30/08/2017 pela seguinte Banca Examinadora.

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Dedicatória

Dedico este trabalho a minha pequena Maria Luiza, que apesar de ainda estar a caminho, já tem sido para mim uma grande fonte de inspiração.

Agradecimentos

Agradeço primeiramente a Deus, pelo dom da perseverança. Por me dar forças para enfrentar as dificuldades e transforma-las em vitórias e aprendizados. Sem Ele eu não seria absolutamente nada.

À minha família, meu pais e minha irmã, por sempre me apoiarem em cada decisão tomada, e me aconselharem nos momentos mais importantes. Vocês são tudo pra mim!

À minha querida esposa Débora, pela dedicação, incentivo e amor. E por sempre compreender e me dar apoio nos momentos de ausência, principalmente nas horas em que "a coisa apertou". Te amo.

Ao chefe e orientador, Dr. Thierry Frédou. Obrigado pela orientação, ensinamentos e pelas grandes ajudas, tanto no período do mestrado, como agora no doutorado. Também não esquecendo da "chefa", a Dr.ª Flávia Lucena, muito obrigado pelas muitas correções e valiosas sugestões durante todo o processo de escrita dessa tese. Aos dois, deixo o meu muito obrigado por toda confiança depositada e por acreditarem em mim. Obrigado de coração!

Ao Dr. Bastien Mérigot, pela coorientação. Mesmo estando distante, sempre foi presente nos momentos necessário, sempre com valiosas sugestões e correções. Muito obrigado.

À Profª. Drª. Maria Elisabeth, a Prof.ª Dr.ª Patrícia Pinheiro, ao Prof. Dr. William Severi e ao Prof. Dr. Paulo Guilherme, por aceitarem fazer parte da minha banca e pelas valiosas sugestões, as quais enriqueceram ainda mais esta tese.

A todos que já fizeram e fazem parte do laboratório BIOIMPACT, pelas ajudas nas coletas, pelos intermináveis dias de processamento e pela parceria e amizade nesses últimos cinco anos e meio.

A CAPES, pela concessão da bolsa de estudos, e ao programa de Pós-Graduação em Recursos Pesqueiros e Aquicultura.

E finalmente a todos que direta ou indiretamente participaram dessa caminhada. Apesar do nome não está aqui, sou igualmente agradecido a todos. Muito obrigado!

Resumo

A presente tese teve como principal objetivo descrever a ictiofauna estuarina do estado de Pernambuco, considerando indicadores clássicos de diversidade, complementada com técnicas que avaliam a diversidade funcional. Por meio de um estudo de caso, avaliou-se a vulnerabilidade das principais espécies capturadas pela pesca de camboa, através da metodologia de Análise de Produtividade e Susceptibilidade (PSA), que tem como base a produtividade biológica relacionada às características de vida de um estoque, e a susceptibilidade dos organismos à captura durante um determinado tipo de pescaria. Os dados foram coletados trimestralmente entre o período de outubro de 2012 a dezembro de 2014, nos estuários de Itapissuma, Suape, Sirinhaém e Rio Formoso; utilizando-se apetrechos de pesca ativos (arrastão de praia e mangote) e passivos (camboa). Foram amostrados um total de 48.754 indivíduos, pertencentes a 122 espécies, 80 gêneros e 36 famílias. Os indicadores de diversidade, representados pelos índices de dominância e equitabilidade, revelaram que, apesar de Itapissuma possuir uma grande diversidade de espécies, essas são bastante próximas entre si do ponto de vista taxonômico, quando comparadas com as de outros estuários. Em relação a diversidade funcional, Suape, Sirinhaém e Rio Formoso apresentaram uma maior relação funcional entre si, com espécies bastante próximas, o que pode estar relacionado ao fato das áreas serem geograficamente próximas e apresentarem características físicas bastante semelhantes. O oposto foi observado para Itapissuma, que apresentou indivíduos funcionalmente distantes entre si, fator associado às características do ambiente, como o substrato lamoso e a grande incidência de mangues, que proporcionam uma maior diversidade de habitats e uma assembleia de peixes morfologicamente distinta. Para a vulnerabilidade, as espécies pertencentes às famílias Centropomidae e Lutjanidae foram classificadas como de alto risco, ou seja, altamente vulneráveis à arte de pesca (neste caso, a camboa), devido às caraterísticas da história de vida – espécies com tamanho relativamente grande e elevada longevidade – e à maior suscetibilidade, considerando principalmente o elevado percentual de captura de juvenis. Estudos que combinam a utilização de índices de diversidade, assim como aqueles relacionados aos padrões ecológicos e biológicos, fornecem um melhor conhecimento do estado em que se encontram as assembleias de peixes em estuários.

Palavras-chave: Produtividade; biodiversidade; ecossistema estuarino; pesca artesanal; monitoramento.

Abstract

The main objective of this thesis was to describe the estuarine ichthyofauna of the State of Pernambuco, considering classical indicators of diversity, complemented with techniques that evaluate functional diversity. Additionally, with a case study, the vulnerability of the main species captured by block net was evaluated through the methodology of Productivity and Susceptibility Analysis (PSA), which is based on biological productivity, related to the life characteristics of a stock, and the susceptibility of organisms to be caught by a particular type of fishing gear. For this study, data were collected quarterly between October 2012 and December 2014 in the estuaries of Itapissuma, Suape, Sirinhaém and Rio Formoso; Using active (beach seine) and passive (block net) fishing gear. A total of 48,754 individuals belonging to 122 species, 80 genera and 36 families were sampled. The diversity indexes, through the indexes of dominance and evenness, revealed that despite Itapissuma has a great diversity of species, its species are taxonomically closer when compared to the other estuaries. In relation to the functional diversity, Suape, Sirinhaém and Rio Formoso, had the lowest levels of functional diversity Q and species richness S, with species very close in the functional typology, which can be related to the fact that the areas are geographically close and present similar physical characteristics. The opposite was observed for Itapissuma, which presented functionally distant individuals, a factor associated with environmental characteristics, such as muddy substrate and high incidence of mangrove, which provide a greater diversity of habitats and a fish assemblage with species morphologically distinct from the other three estuaries. For the vulnerability, the species belonging to the family Centropomidae and Lutjanidae were classified as high-risk species, being highly vulnerable to fishing gear (in this case, block net), given the life history characteristics - species with relatively large size and high longevity - and the greater susceptibility, especially considering the high percentage of juvenile capture. Studies combining the use of diversity indexes, as well as those relating ecological and biological patterns, provide us with a better understanding of the state of fish assemblages as well as their ecological patterns.

Key-Words: Productivity; Biodiversity; Estuarine ecosystem; Fishing; Monitoring.

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Sumário

1- Introdução

Dentre os vários tipos de ecossistema marinhos localizados na zona costeira, destacam-se os ambientes estuarinos e os manguezais, habitats essenciais para o desenvolvimento de vários organismos aquáticos. Estas regiões desempenham papel importante no ciclo biológico das espécies, servindo como berçário tanto para espécies tipicamente estuarinas, como para as marinhas que fazem uso dessas áreas durante seu ciclo reprodutivo (Silva et al., 2008). Estuários tropicais são há muito tempo o ponto focal para muitas atividades humanas. Dentre os vários serviços ecossistêmicos de relevância, estas zonas de transição, entre o continente e o mar, são estratégicas para atividades portuárias, além de servirem como fonte de alimento, sendo de importância vital para seus habitantes (Blaber, 1997). Entretanto, estas zonas são sujeitas a uma gama de impactos, que ameaçam a biodiversidade estuarina (LIRA et al., 2010 a, b) e, a necessidade de se monitorar a diversidade dos peixes que vivem nesses ambientes, principalmente para fins de manejo e uso sustentável, é de grande relevância para a manutenção da zona estuarina.

Com este propósito, as assembleias de peixes podem ser monitoradas de várias maneiras, destacando-se aquelas metodologias que abordam a diversidade através da avaliação de múltiplos componentes (Gaertner et al., 2010; Lefcheck et al., 2014; Mérigot et al., 2007; Wilsey et al., 2005), como a número de espécies e equitabilidade, além daquelas que consideram a relação taxonômica entre as espécies de peixes, resultando em quais índices podem se mostrar mais sensíveis a fatores ambientais e/ou antrópicos (CLARKE e WARWICK, 1998). Desta forma, o monitoramento de um componente de diversidade usando um ou poucos índices de diversidade pode induzir a uma descrição incompleta da biodiversidade, acarretando uma possível perda de respostas importantes diante de fatores ambientais e/ou humanos (D’Agata et al., 2014; Stuart-Smith et al., 2013).

Dentre os efeitos dos impactos antrópicos, é importante identificar aqueles decorrentes da adição ou perda de organismos sobre as funções ecossitêmicas (HALPERN e FLOETER, 2008). Além da perda de espécies, a perda de funções específicas é tida como uma das principais ameaças, podendo colocar em risco os processos e serviços dos ecossistemas (Bellwood et al., 2012; Naeem et al., 2012). Entretanto, pouco se conhece sobre os efeitos dessas modificações na diversidade funcional das comunidades (Naeem, 2002). A função específica exercida pelas espécies em um determinado ambiente está associada à redundância funcional, ou seja, algumas espécies possuem funções semelhantes dentro das comunidades e ecossistemas em que vivem, podendo então ser substituídas por outras, causando pouco ou nenhum impacto sobre os processos do ecossistema em caso de perda de espécies (ROSENFELD, 2002; MICHELI e HALPERN, 2005). Diante disto, as avaliações voltadas para o estudo da biodiversidade devem se concentrar principalmente em identificar quais são as espécies, grupos funcionais e ecossistemas mais vulneráveis a processos de deficiência funcional, ou seja, a perda de espécies que exercem uma determinada função, ajudando assim a compreender como funciona a organização da diversidade entre diferentes comunidades e ecossistemas (Rosenfeld, 2002), de modo que estes possam ser priorizados pela gestão pesqueira.

Atualmente, estudos ecológicos têm focado nas implicações do declínio da biodiversidade sobre o funcionamento dos ecossistemas, propondo medidas que atuem diretamente sobre o ambiente (PETCHEY e GASTON, 2002, 2006; MOUCHET et al., 2010; NAEEM et al., 2012; MOUILLOT et al., 2013, 2014). Entretanto, grande parte destes são realizados em ambientes temperados (Albouy et al., 2011; Brind'Amour et al., 2011; Buisson et al., 2013; Dumay et al., 2004; Eros et al., 2009; Mason et al., 2007; Mouillot et al., 2007; Santoul et al., 2005). Poucos são aqueles voltados para a avaliação da diversidade funcional de peixes nas regiões tropicais, destacando-se os trabalhos realizados por Halpern e Floeter

(2008), Bender et al. (2013b) e Mouillot et al. (2014), que avaliaram a diversidade e estrutura funcional de espécies de peixes recifais em ambientes tropicais; Ferreira e Petrere (2009) e Pease et al. (2012) que estudaram espécies de peixes lagunares; e Nicolas et al. (2010), Villéger et al. (2010), Villéger et al. (2012) e Mendez et al. (2012) que consideraram espécies de peixes estuarinas. Quando analisados de maneira mais detalhada, comparando-se aos estudos realizados nos mais variados tipos de ecossistemas (e.g. lagoas, ambientes marinhos, riachos), apenas 10% dos trabalhos são realizados em regiões estuarinas (Manna et al., 2013).

A atividade pesqueira em ambiente estuarino vem modificando a ictiofauna local, uma vez que a sobreexplotação, a degradação ambiental e a perda do habitat natural estão entre as principais ameaças (BLABER et al., 2000; HALL e MAINPRIZE, 2004, REYNOLDS et al., 2005). Apesar de sua baixa produtividade, a pesca do Nordeste do Brasil tem uma grande importância socioeconômica, e a pesca artesanal estuarina de Pernambuco é responsável pela maior parte da produção do pescado do estado, uma vez que a mesma está associada a regiões ricas em manguezais (Luiz Lira et al., 2010). Dentre as várias características apresentadas pelas artes de pesca utilizadas nos estuários (rede de camboa, arrastão de praia, mangote, etc.), destaca-se a baixa seletividade, a qual ocasiona a captura de uma variedade de espécies e elevado número de indivíduos ainda em fase juvenil (GIGLIO e FREITAS, 2013; SILVA-JÚNIOR et al., 2016; MÉRIGOT et al., 2017). Apesar dessa grande captura de juvenis, são poucos os estudos focados nas espécies capturadas, sejam elas provenientes da pesca estuarina ou costeira (Feitosa et al., 2008; Silva-Júnior et al., 2013, 2015).

Estudos sobre padrões ecológicos de espécies capturadas como fauna acompanhante são relevantes para a obtenção de dados sobre os impactos causados por determinado tipo de apetrecho, assim como para a compreensão do equilíbrio do ecossistema que está sendo explotado (Branco et al., 2006). Para que se tenha uma melhor compreensão da sustentabilidade biológica, da dinâmica e do potencial produtivo de um determinado ecossistema, é importante compreender a forma de atuação da atividade pesqueira sobre o mesmo (Rodrigues-Filho et al., 2015), uma vez que o conhecimento desse conjunto de fatores implica em um conhecimento taxonômico e ecológico dos componentes bióticos que vivem no ambiente. Na região Nordeste, informações sobre o uso e composição das pescarias ainda são insuficientes (SANTOS, 1996; SANTOS et al., 1998; BRAGA et al., 2001; TISCHER e SANTOS, 2003; DANTAS et al., 2012; SILVA-JÚNIOR et al., 2013). De acordo com Santos et al. (1998), para esta região, as espécies que são rejeitadas geralmente são aquelas que não fazem parte da dieta alimentar da população pobre, uma vez que as pescarias dessa região são em sua grande maioria artesanal e em alguns casos voltadas para a subsistência.

Diante desse quadro, avaliar os riscos ou ameaças que uma pescaria impõe sobre os vários aspectos de um ecossistema – principalmente aqueles que se baseiam em características ecológicas, econômicas ou sociais – está entre os pontos essenciais para o desenvolvimento de planos de gerenciamento pesqueiro (Astles et al., 2006). Pois, a avaliação da extensão dos impactos causados pelas atividades antrópicas abrange, estudos dos habitats, biodiversidade, espécies ameaçadas ou grupos de espécies, além dos processos ecológicos (Astles et al., 2009; Ward et al., 2002).

As Análises de Riscos Ecológicos (*Ecological Risk Assessment* – ERA) estão entre as ferramentas mais utilizadas para a realização dessas avaliações (Astles et al., 2009; M.G. Bender et al., 2013; Braccini et al., 2006; Hobday et al., 2011; Lucena-Frédou et al., 2017; Stobutzki et al., 2001a; Villa et al., 2002; Zhou et al., 2016). Nos últimos anos, um dos grandes desafios é a realização de estudos relacionados à análise de risco em pescarias com deficiência de dados (FLETCHER et al., 2002; ASTLES et al., 2006), a qual se baseia em atributos ou propriedades dos organismos aquáticos, necessários para avaliar o estado em que se encontram os estoques explotados (ASTLES et al., 2006, 2009). Devido à ausência de dados, os impactos sobre as populações pesqueiras muitas vezes podem passar despercebidos, uma vez que essa problemática pode mascarar possíveis declínios populacionais em determinadas regiões (Dulvy et al., 2000).

Este é o caso de pescarias do nordeste do Brasil, como a pesca de camboa, na qual há uma total ausência de avaliação dos estoques explotados por esta modalidade, decorrente principalmente da carência de dados de captura, esforço e dos principais parâmetros da história de vida das espécies. Sendo assim, as Análises de Produtividade e Susceptibilidade (PSA) e as Análises de Risco Ecológicos (ERA) são vistas com uma alternativa bem adaptada para situações em que ocorre ausência de dados, gerando resultados que podem ser utilizados na avaliação da vulnerabilidade e risco ecológicos (Mcfadden, 2007; Walker et al., 2002).

Nos últimos anos, as análises de risco foram aplicadas em várias áreas, como para elasmobrânquios (ZHOU e GRIFFITHS, 2008; PATRICK et al., 2010; GALLAGHER et al., 2012; CORTÉS et al., 2010, 2015), cetáceos (BROWN et al., 2013) e teleósteos (STOBUTZKI et al., 2001a, 2001b; WAYTE et al., 2006; HOBDAY et al., 2011; ZHOU et al., 2013, 2016; VISINTIN e PEREZ, 2016; LUCENA-FRÉDOU et al., 2017). No Brasil, tais métodos ainda são pouco utilizados, apesar da grande quantidade de estudos envolvendo zonas costeiras, destacando-se o estudo realizado por Visintin e Perez (2016), os quais avaliaram através de uma PSA, a vulnerabilidade das espécies capturadas através da pesca de emalhe de fundo na região sul do país. Apesar de ter uma abordagem mais global, Lucena-Frédou et al. (2017) também estimaram a vulnerabilidade de espécies de atuns capturados no Atlântico Sul e Oceano Índico, através de uma análise de produtividade e susceptibilidade. Embora a PSA esteja ganhando bastante destaque em várias regiões do mundo nos últimos anos, ainda são poucos os estudos voltados para as espécies capturadas em áreas tropicais, como os estuários. Recentemente, Davoodi et al. (2017) desenvolveram uma análise de risco ecológico através de índices de qualidade sedimentar em uma zona estuarina do Golfo Pérsico. Apesar de não ser voltado para a fauna aquática, este trabalho é um dos poucos com essa metodologia realizada em estuários, o que demonstra a importância de se conhecer mais sobre estas áreas tão importantes para os organismos que dela dependem, e que estão sofrendo uma série de efeitos decorrentes dos impactos antrópicos, em ambientes tropicais (SILVA-JÚNIOR et al., 2016; MÉRIGOT et al., 2017).

As Análises de Risco Ecológico (ERA) e Análises de Produtividade e Susceptibilidade (PSA) ajudam a identificar características relacionadas às espécies de peixes, relacionando-as com as ameaças extrínsecas, para que sejam fornecidas informações sobre o quanto estes organismos estão vulneráveis à arte de pesca (M.G. Bender et al., 2013). Devido à escassez de dados bioecológicos das espécies distribuídas nos mais diversos tipos de ambientes, principalmente para aquelas capturadas em países em desenvolvimento (Kelleher, 2005), as análises de riscos ecológicos mostram-se como uma boa alternativa para o desenvolvimento de planos de manejo, uma vez que trabalham com incertezas para a identificação de áreas de possíveis riscos (Small et al., 2013).

Pernambuco é um bom local para aplicar o estudo de caso da avaliação da biodiversidade, considerando que os estuários no estado apresentam um elevado nível de impacto antrópicos, sendo um deles proveniente da atividade pesqueira, com grande importância socioeconômica. Em Pernambuco, a pesca artesanal estuarina é responsável pela maior parte da produção do pescado do estado, uma vez que a mesma está associada a regiões ricas em manguezais (Luiz Lira et al., 2010). O estado de possui uma região costeira com aproximadamente 34 comunidades pesqueiras, distribuídas entre 15 municípios ao longo dos seus 187 km de extensão (FIDEM, 1987). As suas 17 zonas estuarinas são caracterizadas por uma forte presença de manguezais (PAIVA e ARAÚJO, 2010), dos quais muitos estão sofrendo grandes impactos causados pelo desmatamento, processos erosivos, poluentes provenientes da agricultura da cana-de-açúcar e esgotos domésticos, assim como os causados pela aquicultura (SOUZA e SAMPAIO, 2001; CPRH, 2003a, 2003b). Tais fatores antrópicos podem afetar as comunidades aquáticas, causando perturbações nos cíclos biológicos de espécies marinhas e estuarinas, perda na biodiversidade e redução na produtividade pesqueira (PAIVA e ARAÚJO, 2010).

Diante do exposto, a presente pesquisa tem como objetivo avaliar o estado da ictiofauna de estuários do estado de Pernambuco, considerando a avaliação da diversidade desses ambientes, com o intuito de observar quais desses índices podem ser mais sensíveis a variações ambientais ou antrópicas, e da diversidade funcional, como forma de complementar os índices clássicos de diversidade e ter um melhor entendimento de como as espécies de peixes estão relacionadas ao funcionamento dos ambientes onde são capturadas. Adicionalmente foi realizada uma avaliação da produtividade e susceptibilidade das várias espécies de peixes capturadas pela pesca da camboa, visando estimar a vulnerabilidade das mesmas ao apetrecho utilizado durante as pescarias, descrevendo assim os principais padrões ecológicos observados para as áreas em questão, além de identificar possíveis espécies prioritárias para o manejo pesqueiro, considerando os impactos advindos da atividade pesqueira e os níveis de resiliência das mesmas.

A presente tese é composta de três capítulos. No primeiro capítulo, intitulado “Fish assemblages in tropical estuaries of northeast Brazil: A multicomponent diversity approach”, o principal objetivo foi investigar as assembleias de peixes com base em uma análise de múltiplos componentes, comparando-se os estuários pernambucanos submetidos a diferentes condições ambientais e impactos antrópicos. O segundo capítulo, intitulado “Functional diversity of fish in tropical estuaries: A traits-based approach of communities in Pernambuco, Brazil”, foi focado na diversidade funcional dos peixes, a qual funciona como uma maneira de complementar as análises mais clássicas ou índices ecológicos. O último capítulo, intitulado “Análise de produtividade e susceptibilidade das espécies capturadas em estuários tropicais”, consiste em um estudo de caso da análise dos efeitos da pesca da camboa, importante arte de pesca em Pernambuco, abordando uma metodologia semiquantitativa, sob o enfoque ecossistêmico. A metodologia aplicada neste capítulo é uma abordagem exploratória que visa estimar a vulnerabilidade das espécies capturadas, com base em atributos ecológicos e parâmetros da história de vida, sendo descritos os principais padrões ecológicos observados para as espécies, de acordo com a região e método de captura aplicado.

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3- Artigos científicos

3. 1 - Artigo científico I

Artigo científico publicado na Revista ***Ocean & Coastal Management*.**

Ver Anexo 1

Fish assemblages in tropical estuaries of northeast Brazil: A multicomponent diversity approach

Silva-Júnior, C. A. B

Abstract

Biodiversity in estuarine ecosystems suffers from the impact of environmental changes and human activities. This mainly involves changes in temperature, salinity, pollution, habitat degradation or loss and fishing activities. The diversity of species communities is traditionally assessed on the basis of their species richness and composition. However, there is growing interest in taking into account complementary components dealing with species differences (e.g. taxonomic relatedness). In spite of their social, ecological and economic importance, the diversity of tropical estuarine fish assemblages has rarely been monitored by means of a multi-component approach under different human pressure and environmental conditions. We analysed the diversity of exploited fish communities (both target and nontarget species) sampled during scientific surveys within four estuarine complexes in the state of Pernambuco, Brazil: Itapissuma, Suape, Sirinhaém, and Rio Formoso. A total of 122 species were collected within 34 samples. Overall, diversity indices and species models fitting dominance-evenness profiles mainly revealed differences between assemblages from Itapissuma, being the largest estuary with wide areas of mangrove, and the other estuaries. While assemblages from Itapissuma generally encompassed more species and individuals than the other estuaries, species were more closely related from a taxonomic point of view. In addition, a Double Principal Coordinate Analysis (DPCoA) established a typology of assemblages, useful for management purposes, and linked to particular fish families: it highlighted differences between Itapissuma, Suape, Sirinhaém and Rio Formoso. This method combines matrices of species abundances and differences (here taxonomic distances according to the Linnean classification). It was particularly accurate with a first factorial plane explaining 73% of the total inertia, while only 17% was achieved by a traditional Principal Component Analysis (PCA). Overall, this study provides an assessment of the state of fish assemblage diversity in Pernambuco estuaries where contrasted human and environmental conditions occur. It underscores the accuracy of using a multi-component diversity approach, with a multivariate analysis that is not yet widely used, for monitoring the diversity of estuaries for ecosystem-based fisheries management purposes.

Keywords: Estuaries; Fishes; Taxonomic diversity; Monitoring; Diversity indices; Double principal coordinate analysis; Ecosystem-based fisheries management

1. Introduction

Estuaries, interfaces between land and sea, are the site of a range of hydrological (e.g. river discharge), oceanographic and anthropogenic processes. These complex processes create habitats with intense thermal and salinity gradients and variable nutrient and pollutant concentrations (Wolanski, 2007). They notably drive fish patterns of distribution in space and time, and shape the diversity of assemblages (Elliott et al., 2007; Potter et al., 2015). Estuaries are ecologically essential as feeding and breeding grounds, providing appropriate habitats for different stages of the fish life cycle. In addition, they are a migratory pathway for both anadromous and catadromous species (Elliott et al., 2007; Potter et al., 2015). Furthermore, the high socio-economic value of their products, especially as a source of income and food, is widely recognized (Glaser and Diele, 2004; Isaac et al., 2009). However, global environmental change and direct human pressures impact the water quality and biodiversity of estuarine ecosystems (Garcia et al., 2003; Halpern et al., 2008; Borja et al., 2010; Li and de Jonge, 2015). Notably, major changes in fish diversity from the effects of mangrove removal, shrimp farming and agricultural development have been recently documented for estuaries worldwide in terms of assemblage composition, decrease in number of species, feeding groups and larger fishes (Blaber, 2008, 2009, 2013; Weerts and Cyrus, 2002; Singkran and Sudara, 2005; Oribhabor and Ogbeibu, 2010; Viana et al., 2010, 2012).

Aquatic resources are of fundamental importance for human food and for the development of Brazil, which has one of the longest coastlines in the world (9th longest worldwide) (Burke et al., 2001). Recently, Pinheiro et al. (2015) stated that the Brazilian aquatic biodiversity is in peril, notably with regard to fishes, in many cases due to habitat degradation. However, it is widely acknowledged that the success of many fisheries resources is dependent on the quantity and quality of the habitat (Lindall and Thayer, 1982; Auster et al., 1996). The northeast is one of the most densely populated coastal regions in Brazil. In

Pernambuco state, the degradation of coastal ecosystems is most severe around the main urban centre, Recife, mainly due to domestic pollution, industrial activity and habitat degradation and loss. These coastal ecosystems are also threatened by fishing (Elfes et al., 2014) because of the impact on the habitat (Turner et al., 1999) and/or fishes at the top of the trophic networks (e.g. trophic downgrading, Estes et al., 2011). Along the Pernambuco coast are located 34 fishing communities, with ca. 12 000 fishermen (Lessa et al., 2006).

To supply food and satisfy economic needs, aquatic resources are intensively exploited using multiple techniques and multispecies artisanal fisheries (Frédou et al., 2006, 2009a, b). The majority of stocks are either fully or over-exploited, and there is little room for expansion into new fisheries (MMA, 2006). Despite all these threats, the degree of dependency on and use of estuarine ecosystems, even in urban areas, remains very high (MMA, 2002). The diversity of the threats and their intensification in the Pernambuco estuaries (Lira et al., 2010a, b) highlight the need to monitor fish biodiversity for management and sustainable use purposes. While several studies dedicated to the diversity of fish assemblages have been carried out in this area (Paiva et al., 2009; Paiva and Araújo, 2010), there is still a need to assess and quantify their state in the main estuaries subjected to contrasting human pressure and environmental conditions. For this purpose, the diversity of fish assemblages can be monitored more exhaustively by a multi-component diversity approach (e.g. Wilsey et al., 2005; Mérigot et al., 2007; Gaertner et al., 2010; Lefcheck et al., 2014). This approach consists in assessing different components, or facets, such as species number and evenness, and also components that explicitly take into account the degree of difference among species. This reflects the taxonomic relatedness between species, on the basis of which diversity indices can be more sensitive to environmental and/or human drivers (Clarke and Warwick, 1998). Monitoring a single diversity component using one or few diversity indices leads to an incomplete description of the diversity of assemblages, and in turn important responses in the face of environmental and/or human drivers could be missed (e.g. Stuart-Smith et al., 2013; D’agata et al., 2014).

In this context, the aim of this work is to investigate the fish diversity assemblages on the basis of a multi-component approach, comparing tropical estuaries subjected to contrasting environmental conditions and human pressure. Firstly, complementary diversity indices were quantified for each estuary. Secondly, the typology of fish assemblages among these estuaries was characterized using a recent multivariate analysis, the Double Principal Coordinates Analysis (DPCoA), that explicitly considers species differences (here, taxonomic relatedness). Overall, this study provides an assessment of the state of assemblages among the studied tropical estuaries of northeast Brazil, and demonstrates that a multi-component approach of this kind can be useful for ecosystem-based fishery management in estuarine ecosystems.

2. Materials and methods

2.1. Study area

Four estuaries located along the coast of Pernambuco State were studied: Itapissuma, Suape, Sirinhaém and Rio Formoso (Fig.1). The estuarine complex of Itapissuma is located to the north of and closest to the Recife and Olinda urban centres, and is the most strongly influenced by landscape alterations, industrial discharges, shrimp farming and high fishing pressure (CPRH, 2003; Lira et al., 2010a; Gondim, 2015). The other three estuaries are located south of Recife: the Suape estuary has suffered the high impact of deforestation and land reclamation due to the construction of a large industrial port complex (chemical, shipping and logistics companies), with predictable effects on both the biodiversity and fisheries productivity (Paiva and Araújo, 2010). Sirinhaém and Rio Formoso are located in the southernmost area, and although located within an environmental protected area (APA

Guadalupe), they are also impacted by agricultural pollution (mainly sugar cane), shrimp farming and fishing (Lira and Fonseca, 1980). Rio Formoso is also a popular tourism destination (Lira et al., 2010b).

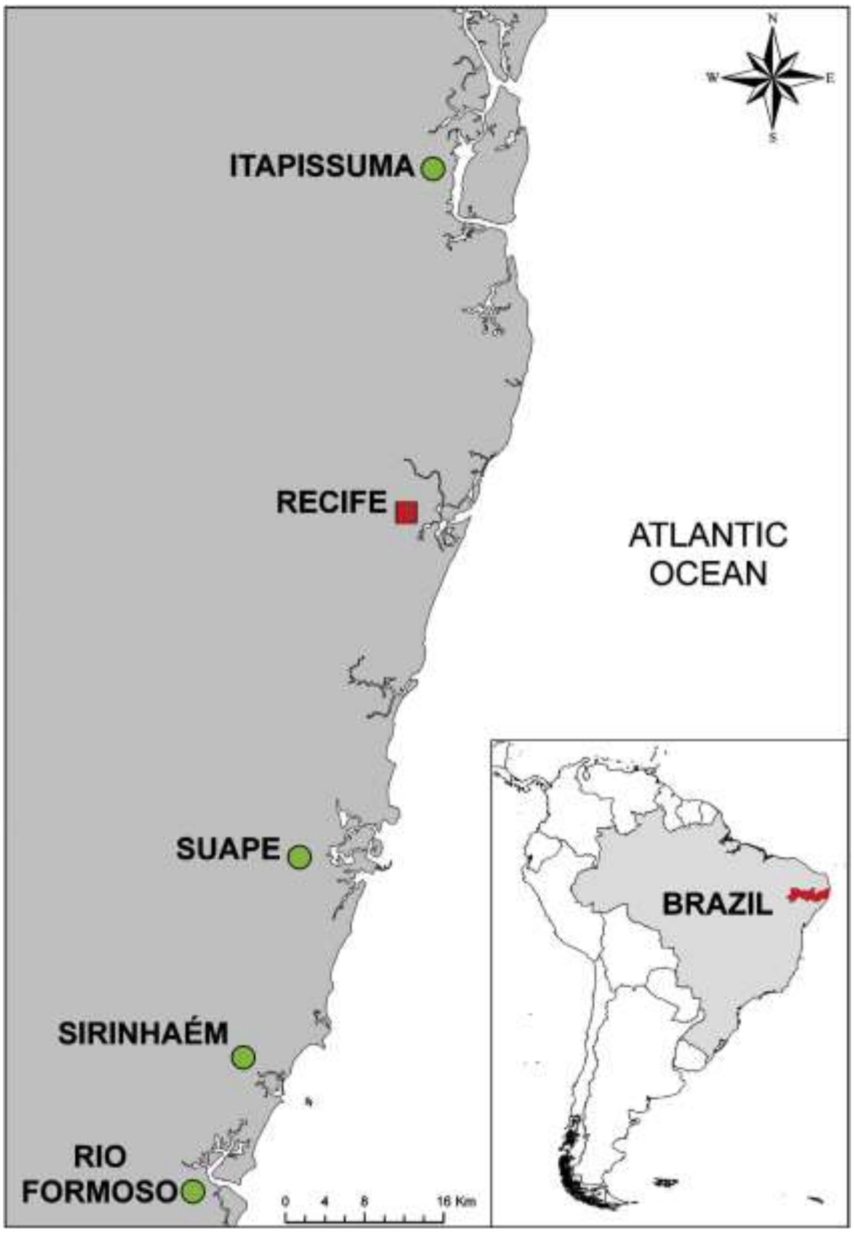

Fig. 1. The study area, Pernambuco, Brazil.

2.2. Sampling design

Fish were sampled from the end of 2012-2014 (see Table S1 in Supplementary Material for sampling dates). A total of 34 samples were obtained using block net and beach seine (seine) gear: 17 in Itapissuma, 9 in Sirinhaém, 4 in Rio Formoso and 4 in Suape. These types of fishing gear were selected as, together, they can cover most fish habitats present in the studied estuaries: shallow coastal areas and sand/mud banks (beach seine), and flooded mangrove and channels (block net). The number of samples per estuary varied according to the characteristics of each type of gear, the influence on tidal variation, and the gear setting time. Beach seine nets measured 80e180mlong, 5mhigh with a mesh size of 20 mm. The operation lasted no more than 20 min and was repeated several times. Block nets were set along the mangrove forest and channels and measured 70-90 m long, 2.5 m high with a mesh size of 50 mm. At low tide, the net was anchored to the bottom. At slack high-water, the net was deployed and attached to stakes and pulled taut 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).

2.3. Statistical analyses

Firstly, we computed a species accumulation curve to assess whether the fish community in the study area was exhaustively sampled, with the gear used (Gotelli and Colwell, 2001). We used a randomized method without replacement. It enables calculation of a mean number of species for a given number of samples, with a 95% confidence interval, as well as obtaining the total number of species caught for the total number of samples considered. We also computed the Chao2 index (Chao, 1987), which extrapolates, from the occurrence of rare species, the total expected number of species in the area, for the given sampling gear. In addition, because the number of samples differed among estuaries (see above), we computed the total rarified number of species for each estuary for a common number of four samples (i.e. minimum sampling size), following the same randomized procedure as described above. To investigate the overall dominance/evenness pattern among the four estuaries, raw rank-abundance curves and species abundance models were also computed (Fig. S1 in Supplementary Materials).

Secondly, for each assemblage (i.e. each of the 34 samples), we computed diversity indices related to three major components of species diversity (species richness, evenness and species taxonomy; see Table 1 for indices properties that justify a priori their complementary use, further details in Mérigot et al., 2007). Then, we studied the correlations among these indices to identify a set providing complementary patterns/responses for the studied estuaries (Mérigot et al., 2007; Gaertner et al., 2010, 2013). This latter step was performed using a principal component analysis (PCA) based on Spearman rank correlation coefficient (see Mérigot et al. (2007) for further details). A correlation coefficient of approximately <0.5 meant that indices were complementary. Subsequently, each of the complementary indices identified above were compared between the four estuaries with a Kruskal-Wallis test (i.e. a non-parametric ANOVA), due to the relatively low and uneven number of samples. When the null hypothesis of this test was rejected (i.e. homogeneity in distributions of rank values between estuaries), the non-parametric multi-comparison test of Siegel and Castellan (1988) was computed to assess the pairwise differences between estuaries (i.e. post-Hoc test).

Finally, a multivariate analysis named Double Principal Coordinate Analysis (DPCoA) (Pavoine et al., 2004) was implemented to provide a typology of fish assemblages (i.e. particular assemblages identified from the species, their taxonomy and number of individuals sampled). Compared to traditional multivariate methods such as the Principal Component Analysis (PCA), the DPCoA has the advantage of explicitly taking into account the species difference. It is computed from both a species abundance matrix and a species distance matrix (here based on taxonomic relatedness between species, e.g. Clarke and Warwick, 1998). The DPCoA provides a multidimensional space assembling the species and the samples to establish a typology of assemblages. We identified the contribution of the taxonomic family to assemblages among studied estuaries on the two first factorial axes (i.e. those which explained the main variability). Then, the optimum number of family groups that differentiate assemblages on these axes was defined by means of the Gap statistic (Tibshirani et al., 2001). All the statistical analyses and diversity indices mentioned above were performed using R software (R development Core R Core Team, 2015).

Table 1. Species diversity components and indices studied. x_i (i = 1, …, S) denotes the abundance of the ith species, N (= $\Sigma_i x_i$) is the total number of individuals in the sample, p_i (= x_i/N) is the proportion of all individuals belonging to species i, N_{max} is the number of individuals of the most abundant species. d_{ij} is the distance given to the path length linking species i to the first common node with species j in the hierarchical taxonomic classification.

Component	**Descriptor name**	**Formula**	**Expected properties**	**Reference**
Number of species	Species richness	*S* = Number of species in a sample	Species richness per sample	
	Margalef	$D_{mg} = \frac{(S-1)}{\ln(N)}$	Adjusted species richness by *N*	Margalef (1958)
Evenness	Heip	$E_{Heip} = \frac{\exp(H')-1}{S-1}$	Sensitive to rare species	Heip (1974)
	Simpson evenness	$E_{1-D} = \frac{1-(\sum_{i=1}^{S} p_i^2}{1-1/S}$	Sensitive to dominant species	Smith and Wilson (1996)
	Berger Parker	$\frac{1}{d} = \frac{N}{N_{max}}$	Sensitive to dominant species	Berger and Parker (1970)
Number of species + evenness	Shannon-Wiener	$H' = \sum_{i=1}^{S} pi \log pi$	Sensitive to rare species	Shannon and Weaver (1949)
	Simpson diversity	$1 - D = 1 - (\sum_{i=1}^{S} p_i^2)$	Sensitive to dominant species	Simpson (1949)
Species taxonomy	Quadratic entropy	$Q = \sum_{i=1}^{S} \sum_{j=i+1}^{S} d_{ij} p_i p_j$	Extension of 1−*D* including taxonomic relatedness	Rao (1982)

3. Results

3.1. Patterns of fish assemblage indices

A total number of 122 fish species were identified in the 34 samples of the study area. The species accumulation curve did not stabilise towards asymptotic values (Fig. 2), as is usually the case in ecological studies, due to the large number of rare species present in natural communities. However, the slope at the end of the curve suggested that the number of additional samples necessary to collect the species more exhaustively with the fishing gear used should only be slightly higher. It is congruent with the Chao2 index which estimated the total number of species in the studied area sampled with these types of gear at 139.29 ± 8.58 (standard deviation) species. For the same number of samples per estuary (here four samples), the total rarified number of species revealed that Itapissuma was the richest, with 58.88 ± 8 species, while the other estuaries encompassed fewer species, with 36.56 ± 4.9, 35 and 32 species for Sirinhaém, Suape and Rio Formoso, respectively. The identity and the proportion of the most dominant species differed across the four estuaries (Table 2). *Diapterus auratus* was the most dominant at Rio Formoso (60.4%), followed by *Gobionellus stomatus* in Itapissuma (50%), and *Sphoeroides testudineus* at Sirinhaém and Suape (29% and 23.6%, respectively (Table 2). The cumulative percentage of the 10 most abundant species in the study area varied according to the estuary: the highest value was recorded at Rio Formoso (91.7%) and the lowest at Sirinhaém (78.7%, Table 2). Furthermore, the whole dominance/evenness profile differed between estuaries according to the raw rank-abundance curves and species abundance models (Fig. S1 in Supplementary Materials).

Table 2 - Abundance, proportion and cumulative proportion for the ten most dominant species in the four studied estuaries: Itapissuma ITA, Sirinhaém SIR, Suape SUA, Rio Formoso RIO.

	Rank	**Abundance**	**Proportion (%)**	**Cumulative prop. (%)**
ITA				
Gobionellus stomatus	1	21207	50	50
Eucinostomus argenteus	2	2158	5.1	55.1
Ctenogobius stigmaticus	3	2064	4.9	60
Mugil curvidens	4	1966	4.6	64.6
Gobionellus oceanicus	5	1921	4.5	69.1
Sphoeroides testudineus	6	1773	4.2	73.3
Eucinostomus gula	7	1673	3.9	77.3
Cetengraulis edentulus	8	1597	3.8	81
Archosargus rhomboidalis	9	1259	3	84
Achirus lineatus	10	981	2.3	86.3
SIR				
Sphoeroides testudineus	1	237	29	13.8
Sciades herzbergii	2	222	31.1	26.7
Diapterus auratus	3	145	15.3	35.2
Achirus lineatus	4	132	15.2	42.9
Centropomus undecimalis	5	132	12.9	50.6
Centropomus parallelus	6	119	12.1	57.5
Diapterus rhombeus	7	116	12.9	64.3
Bairdiella ronchus	8	87	9.3	69.3
Mugil curvidens	9	83	8.9	74.2
Aspistor luniscutis	10	78	12.9	78.7
SUA				
Sphoeroides testudineus	1	256	23.6	23.6
Diapterus auratus	2	195	18	41.6
Centropomus parallelus	3	113	10.4	52
Lutjanus alexandrei	4	101	9.3	61.3
Centropomus mexicanus	5	65	6	67.3

Mugil curema	6	46	4.2	71.5
Eugerres brasilianus	7	45	4.1	75.7
Mugil curvidens	8	37	3.4	79.1
Caranx latus	9	36	3.3	82.4
Centropomus undecimalis	10	36	3.3	85.7
RIO				
Diapterus auratus	1	1835	60.4	60.4
Sphoeroides testudineus	2	277	9.1	69.5
Eugerres brasilianus	3	147	4.8	74.4
Diapterus rhombeus	4	146	4.8	79.2
Mugil curema	5	104	3.4	82.6
Caranx latus	6	66	2.2	84.8
Centropomus undecimalis	7	64	2.1	86.9
Centropomus parallelus	8	60	2	88.9
Centropomus mexicanus	9	44	1.4	90.3
Centropomus pectinatus	10	41	1.4	91.7

The correlation analysis revealed that abundance N, species richness S and Quadratic Entropy Q were complementary to describe patterns of the studied assemblages, providing proxies of the five others indices (see Materials and Methods section). Thus, only these three indices were considered for further analyses. N, S and Q revealed that estuaries had different levels of diversity (Fig. 3, Kruskal-Wallis tests, $12.19 < \text{Chi2} < 18.33$, d.f.=3, $p < 0.01$), especially between Itapissuma and Sirinhaém (multiple comparisons tests, $p < 0.05$). The mean number of individuals and species per sample were higher at Itapissuma (i.e. N=2494.59 ± 1467.31 individuals and S=29.82 ± 9.62 species, respectively). In contrast, the mean taxonomic diversity at Itapissuma (Q=2.14 ± 0.47) was lower compared to Sirinhaém and Suape (i.e. 2.72 ± 0.42 and 2.73 ± 0.19, respectively, see also quartiles distributions in Fig. 3). Q corresponds to the mean taxonomic distances between two randomly chosen individuals in a sample. Thus, while Itapissuma had a higher mean number of species per sample, it encompassed more closely related individuals than the other two estuaries.

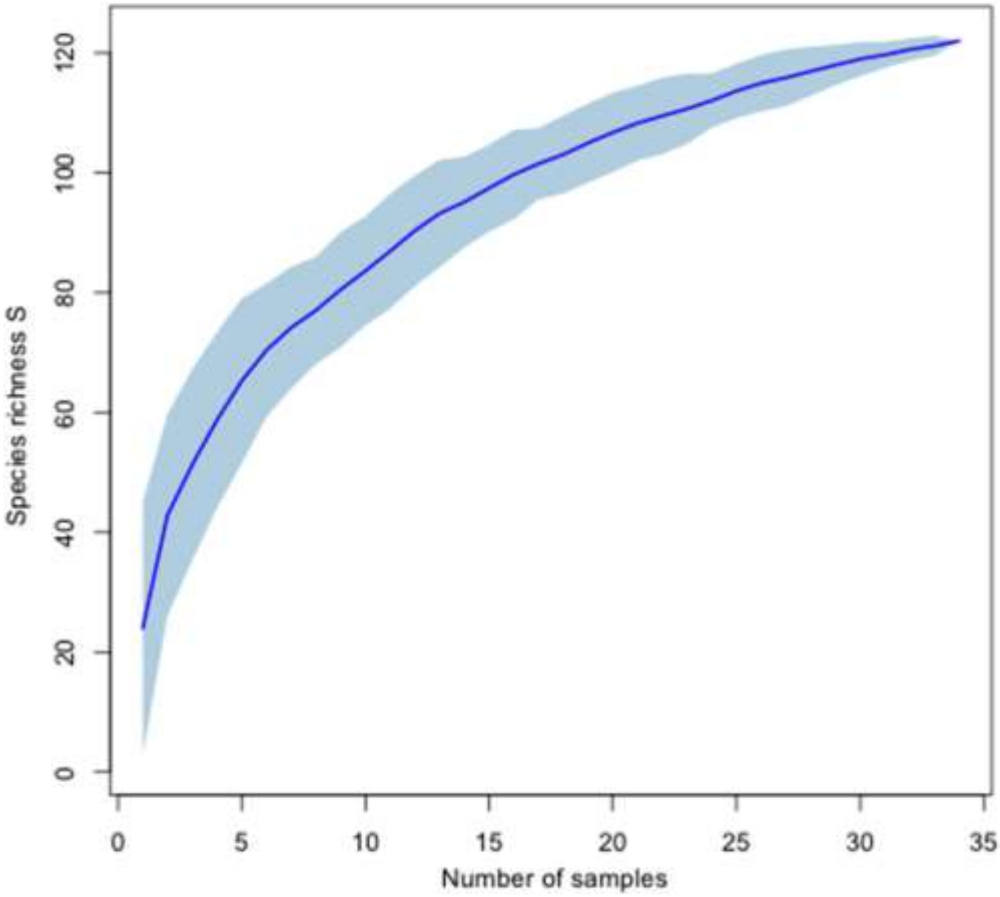

Fig. 2. Species accumulation curve of the studied area, computed by a random method without replacement. Mean species richness value ± 95% confidence interval.

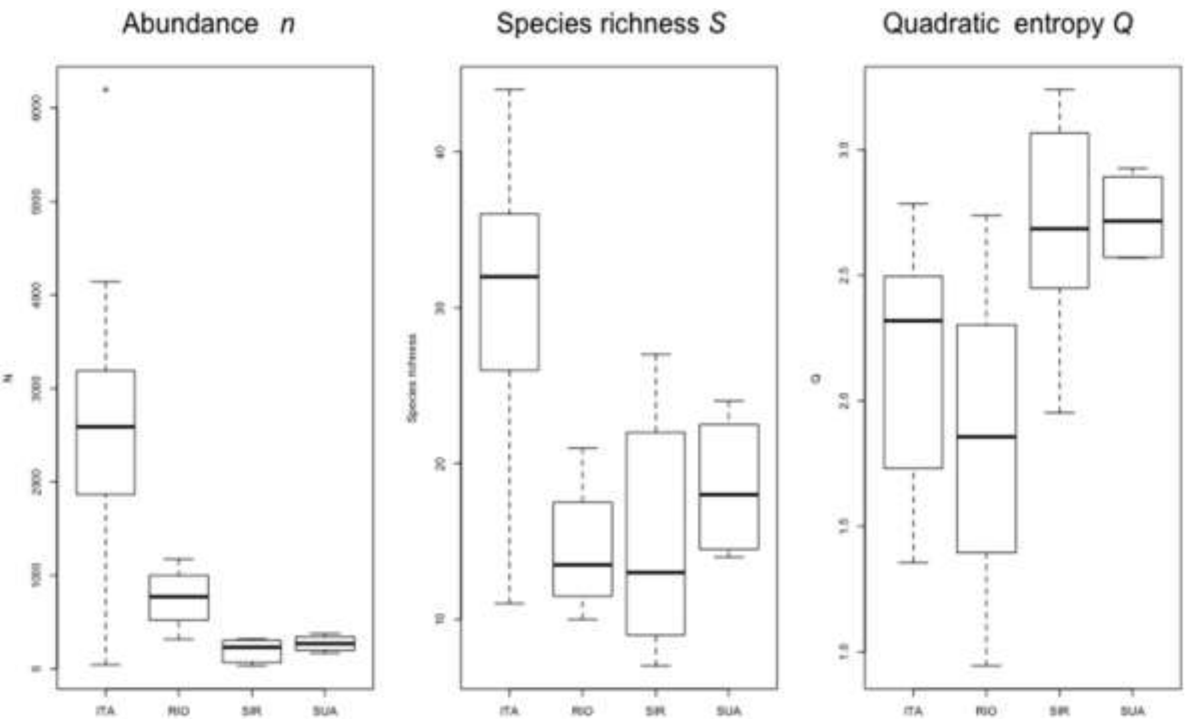

Fig. 3. Boxplots of species richness S, abundance n and quadratic entropy Q (see Table 1) according to the estuary: Itapissuma ITA, Sirinhaém SIR, Suape SUA, Rio Formoso RIO.

3.2. Typology of fish assemblages

The first factorial plane of the DPCoA was particularly accurate for studying the typology of assemblages, as it explained 73.20% of the total variability (axis 1: 54.19% and axis 2: 19.01%), while a PCA (i.e. without taking into account taxonomic relatedness) explained only 17.39%. Sampled assemblages were different between estuaries, especially between Itapissuma and the other three, as highlighted by axis 1, explaining most of the variability (see barycentre locations in Fig. 4a; supported by a Kruskal Wallis test performed on the samples coordinates of axis 1, Chi2=25.84, d.f.=3, $p < 0.001$, followed by a non-parametric multiple comparison test). To a lesser extent on axis 2, assemblages from Rio Formoso were differentiated from the other ones (Chi2=9.90, d.f.=3, $p < 0.05$).

Species families among assemblages of estuaries were not strictly the same, even if some were shared. More specifically, five groups of species families were identified on the first factorial plane (Fig. 4b, Gap statistic computed from 1000 bootstraps). On the one hand, Mugilidae and Gerreidae were associated with Suape, Sirinhaém and Rio Formoso assemblages (Fig. 4b, families with the highest contributions on the 1st and 2nd axes, see the left part of the figure). On the other hand, some species of Gobiidae were only sampled at Itapussima (Fig. 4b, contributions of the right part of the 1st axis).

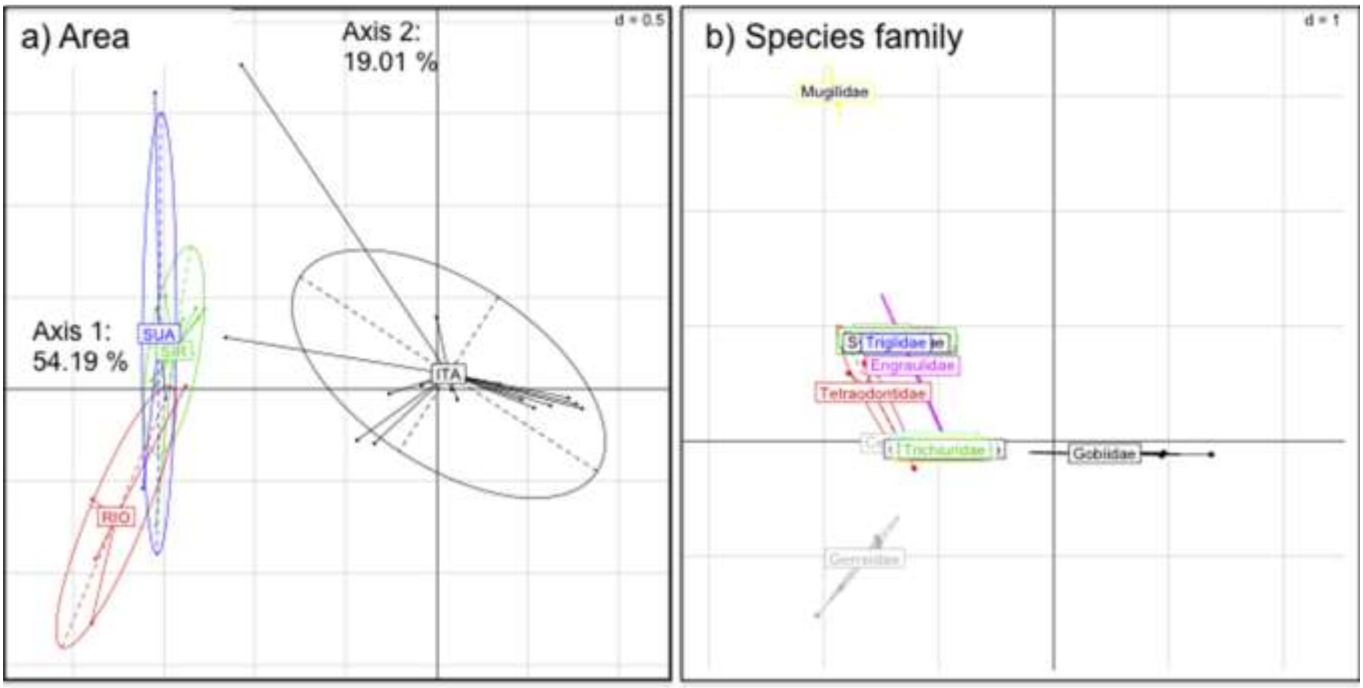

Fig. 4. Double Principal Coordinate Analysis (DPCoA), first factorial plane explaining 73.20% of the total inertia. Projections of a) samples according to the estuary: Itapissuma ITA, Sirinhaém SIR, Suape SUA, Rio Formoso RIO; b) species family.

4. Discussion

4.1. Patterns of fish diversity in estuaries

Our results highlighted that fish assemblages differ among the four estuaries studied in Pernambuco state. Dominance-evenness profiles varied between Itapissuma and Suape compared to those of Rio Formoso and Sirinhaém (Table 2 and Fig. S1 in Supplementary Materials). In addition, Itapissuma had higher mean number of species and individuals per sample than the other estuaries (Fig. 3), but individuals were more closely related from a taxonomic point of view, especially compared to Sirinhaém and Suape (Fig. 3). Different typologies of species assemblages were highlighted by the DPCoA (Fig. 4), taking into account both their abundances and species taxonomic relatedness. Differences between i) Itapissuma ii) Rio Formoso and iii) Sirinhaém, Suape were indeed related to particular species families (Fig. 4). Estuaries within a given region are likely to encompass similar fish assemblages (Whitfield, 1999; Harrison and Whitfield, 2006). However, the four estuaries studied here are of different morphology and size, and are subject to varying degrees of human pressures and abiotic conditions, which in turn lead to particular fish assemblages. Firstly, the overall results highlighting that the diversity of assemblages from Itapissuma was mainly different from that of the other estuaries corroborates the fact that this estuary, composed of different rivers, is considered as the largest estuarine ecosystem in Pernambuco state (Silva et al., 2011). In contrast, the others estuaries are of more similar size and are geographically closer (south of Recife, Fig.1). The size of the estuary influences fish assemblages (e.g. Selleslagh et al., 2009; Nicolas et al., 2010; Cardoso et al., 2011; Pasquaud et al., 2015), notably by increasing the availability of species niches.

Secondly, Itapissuma is a homogeneous muddy estuary, in contrast to the other estuaries. Assemblages differed especially due to the relatively high abundance of some species of Gobiidae, only sampled at Itapissuma (note that the other estuaries featured different species of this family) (Table 2, Figs. 3 and 4). Gobiidae represent a large part of larval fish assemblages in most tropical estuaries (Blaber, 1997), with long larval durations influencing dispersal and colonization (Shen and Tzeng, 2008). They play an important role in estuarine ecosystems (Bruno and O'Connor, 2005), notably as omnivores in the trophic structure of Brazilian tropical estuaries (Joyeux et al., 2004) and worldwide (Whitfield, 2005; Sanvicente-Anorve et al., 2003; Polgar and Bartolino, 2010). The fish larvae of the Santa Cruz Channel (Itapissuma) was investigated by Ekau et al. (2001) and Gobiidae were among the dominant group. Thus, these results highlight the potential importance of several species of this family in the diversity of Itapissuma assemblages, which should contribute to their trophic structure in a significant way, notably due to their dominance (Table 2).

In addition, while Mugilidae were sampled in all estuaries, they were more abundant at Itapissuma (Table 2, Figs. 3 and 4). This is congruent with the fact that this family, encompassing iliophagous species, is well represented in tropical estuaries (Blaber, 1997). These areas are known to be detritus traps, with an abundance of micro-algae and -fauna, especially in mangrove areas and near mudflats (Hutchings and Saenger, 1987). Likewise, Gerreidae were sampled in all estuaries, but more predominantly at Rio Formoso (Table 2, Figs. 3 and 4). Paiva et al. (2009) also found Gerreidae among the dominant families in Rio Formoso. This family is also characteristic of tropical estuaries (Blaber, 2008). They have a particular morphology and feeding pattern of eating slower moving or small invertebrates, but they can adapt to feed on other resources such as bivalve siphon tips (see Blaber, 2008 and references therein). The fact that Gerreidae were more abundant at Rio Formoso than in the other areas, could suggest a particular resource availability in this estuary that is part of a multiple use state environmental protection area, and is also located in proximity to protected reef areas (Ferreira et al., 2006), which could enhance connectivity (Ramos et al., 2016).

Finally, assemblages among the studied estuaries were dominated by few species (Table 2, Fig. S1 in Supplementary Materials), as observed in other areas worldwide (Cabral et al., 2001; Akin et al., 2005; Maes et al., 2005; Elliott et al., 2007). Most of these species are partly or totally dependent on mangroves, the dominant habitat in tropical estuaries (dependence was relatively predominant within Itapissuma compared to the other estuaries), used for all or part of their life cycle (Blaber, 2008). In addition, the fact that juveniles of marine species were also sampled during our surveys is particularly interesting with regard to the connectivity, production and management of sea fisheries (Blaber, 2009). It notably underlines the need to avoid mangrove habitat loss, which is also usually followed by an increase in fishing pressure with the use of more efficient gear.

Furthermore, the use of the estuary at different stages in the species life cycle also underlines the importance of performing temporal surveys. Throughout this study, seasonal variations could not be assessed yet because of the insufficient number of samples in some estuaries, but further surveys would be useful to investigate this. Different aspects of fish assemblages, such as diversity, species composition, ecological and feeding guilds distribution, can have contrasted patterns in relation to seasons, depending on the use of the estuaries, such as, for instance, permanent occupancy, temporal nursery or migration (Gordo and Cabral, 2001; Pombo et al., 2007; Selleslagh and Amara, 2008; Cardoso et al., 2011). Such temporal assessment should thus offer a more in-depth extension of this study that constitutes a starting point by characterizing the state of fish assemblages in estuaries subjected to contrasting human pressure and environmental conditions. It provides a first step towards monitoring assemblages as a basis for subsequently assessing the potential future human impact and environmental changes as an aid to fisheries management.

4.2. Implications for fisheries management

In order to implement the most widely accepted fisheries approaches, such as the code of conduct of responsible fisheries (Pitcher et al., 2009), Brazil is expected to incorporate ecosystem based fishery management (EBFM), for which the non-exploited species are an essential part of the ecosystem (Pikitch et al., 2004). The multi-component approach we used to quantify fish assemblage diversity is thus directly part of the EBFM (Mérigot et al., 2007; Gaertner et al., 2010, 2013). In addition, the DPCoA was more powerful than a traditional PCA to identify assemblages among estuaries (73% and 17% of total variation explained by the first factorial plane, respectively). It underscores the advantage of including species taxonomic differences in this multivariate analysis to take into account more realistically of the species ecology. However, the DPCoA has to date been poorly used for studying fish assemblages (see Pavoine et al., 2009; based on phylogenetic data). Our results suggest that this method could be of particular interest for EBFM.

Many fish species inhabit or occur in estuaries during juvenile phase, with estuary-coastal ontogenetic migrations after the first few years of life (Sheaves et al., 2015). This is notably the case in the four investigated estuaries, as underlined by the presence of juveniles sampled during surveys (see above, and Aschenbrenner and Ferreira (2015), more specifically for Lutjanid species in the study area). This is a key component of nursery habitat value (Sheaves et al., 2015) that enhances the importance of considering habitat and ecosystem connectivity in management strategies. The tropical estuaries in the present study are part of ecosystem mosaics with seagrasses, mangroves and costal reefs (Ferreira et al., 2016) that together are supposed to maintain species life cycles and biodiversity. Mosaics of Marine Protected Areas (MPAs) have increased potential to favor EBFM in fisheries management and to extend its range of application (Seixas and Veira, 2015). However, MPA areas have only been implemented on the southern part of the study area (Rio Formoso and Sirinhaem complexes), and measures taking into account EBFM in fisheries management processes are generally lacking in the studied region. Community-based management options, such as extractive reserves, where both sustainable fisheries and conservation objectives are targeted, although facing challenges Santos and Schiavetti, 2014), remain promising approaches to reduce conflicts and maintain biodiversity.

Acknowledgments

We thank three anonymous reviewers for their constructive comments on an earlier version of the manuscript. We thank all students and researchers who participated in field collections. We are also grateful to the Conselho Nacional de Desenvolvimento Científico e Tecnológico (CNPq) for providing a research grant to F. Lucena Frédou, T. Frédou and B. Ferreira. This work was carried out under the National Institute on Science and Technology in Tropical Marine Environments - INCT-AmbTropic (CNPq process 565054/ 2010-4). We thank Michael Paul for correcting the English of the paper.

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SUPPLEMENTARY MATERIAL

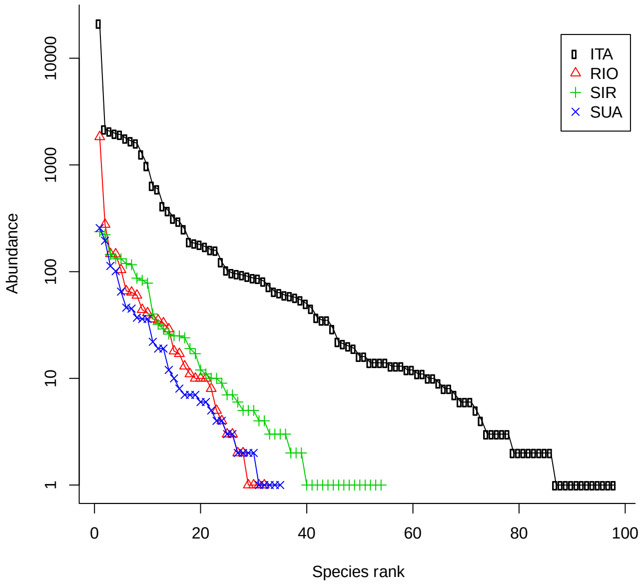
Fish assemblages in tropical estuaries of Northeast Brazil: a multi-component diversity approach.

Table S1: Data collection dates according to the estuary and type of fishing gear.

Estuary	Fishing gear/Date
Itapissuma	**Block net**
Itapissuma	January-13
Itapissuma	May-13
Itapissuma	June-13
Itapissuma	August-13
Itapissuma	November-13
Itapissuma	March-14
Itapissuma	May-14
	Seine
Itapissuma	January-13
Itapissuma	May-13
Itapissuma	August-13
Itapissuma	October-13
Itapissuma	November-13
Itapissuma	December-13
Itapissuma	January-14
Itapissuma	February-14
Itapissuma	March-14
Itapissuma	April-14
Itapissuma	May-14
Itapissuma	June-14
Itapissuma	July-14
Itapissuma	August-14
Itapissuma	September-14

Estuary	Fishing gear/Date
Sirinhaém	**Beach seine**
Sirinhaém	March-13
Sirinhaém	May-13
Sirinhaém	August-13
Sirinhaém	November-13
Sirinhaém	February-14
Sirinhaém	May-14
	Block net
Sirinhaém	October-12
Sirinhaém	February-13
Sirinhaém	May-13
Sirinhaém	August-13
Sirinhaém	November-13
Sirinhaém	February-14
Sirinhaém	June-14
Suape	**Block net**
Suape	June-13
Suape	August-13
Suape	November-13
Suape	March-14
Rio Formoso	**Block net**
Rio Formoso	May-13
Rio Formoso	August-13
Rio Formoso	November-13
Rio Formoso	March-14

Fig. S1: Species rank-abundance curves for each estuary: Itapissuma ITA (17 samples), Sirinhaém SIR (9), Suape SUA (4), Rio Formoso RIO (4).



We computed raw rank-abundance curves (Whittaker, 1965; Magurran, 2004) for each estuary in order to provide a graphical assessment of the dominance/evenness pattern of fish assemblages. The log rank-abundance curves of each estuary were then fitted by the Broken stick, pre-emption, log-Normal, Zipf and Zipf-Mandelbrot models in order to assess if the structure of the assemblages differed between estuaries (Wilson, 1991). The best model for a given estuary was kept according to the lowest Bayesian's Information Criterion (BIC) value (Etienne and Wolff, 2005).

The whole dominance/evenness profile differed between estuaries according to the raw rank-abundance curves (Fig. S1). Likewise, and species abundance models, selected according to the BIC criterion, supported different assemblage structures between estuaries: log-normal models for Itapissuma and Suape, whereas a Zipf and Pre-emption models best fitted data of Rio formoso and Sirinhaém, respectively.

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3. 2 - Artigo científico II

Artigo científico publicado na Revista **Estuarine, Coastal and Shelf Science**.

Ver Anexo 2

Functional diversity of fish in tropical estuaries: a traits-based approach of communities in Pernambuco, Brazil

Silva-Júnior, C. A. B.

Abstract

Environmental changes and human activities may have strong impacts on biodiversity and ecosystem functioning. While biodiversity is traditionally based on species richness and composition, there is a growing concern to take into account functional diversity to assess and manage species communities. In spite of their economic importance, functional diversity quantified by a traits-based approach is still poorly documented in tropical estuaries. In this study, the functional diversity of fishes was investigated within four estuaries in Pernambuco state, northeast of Brazil. These areas are subject to different levels of human impact (e.g. mangrove deforestation, shrimp farming, fishing etc.) and environmental conditions. Fishes were collected during 34 scientific surveys. A total of 122 species were identified and 12 functional traits were quantified describing two main functions: food acquisition and locomotion. Fish abundance and functional dissimilarities data were combined into a multivariate analysis, the Double Principal Coordinate Analysis, to identify the functional typology of fish assemblages according to the estuary. Results showed that Itapissuma, the largest estuary with a wider mangrove forest area, differs from the other three estuaries, showing higher mean values per samples of species richness S and quadratic entropy Q. Similarly, it presented a different functional typology (the first two axes of the DPCoA account for 68.7% of total inertia, while those of a traditional PCA based solely on species abundances provided only 17.4%). Conversely, Suape, Sirinhaém, and to a lower extent Rio Formoso, showed more similarity in their diversity. This result was attributed to their predominantly marine influenced hydrological features, and similar levels of species abundances and in morphological traits. Overall, this study, combining diversity indices and a recent multivariate analysis to access species contribution to functional typology, allows to deepen diversity assessment by providing additional information regarding the functional pattern of fish assemblages.

Key words: Monitoring, Diversity indices, Double Principal Coordinate Analysis, Estuarine ecosystems, Species function.

1 - Introduction

Estuaries are highly productive and important providers of ecosystem services, included among the most biologically valuable areas of the world (COSTANZA, 1997). However, human activities, including physical and chemical transformations, habitat destruction and changes in biodiversity impacted these environments in recent years (HALPERN et al., 2007, 2008; BORJA et al., 2010). Thus, the assessment of the biodiversity in estuarine regions through functional diversity is crucial, which relates to the species role in the ecosystem and potentially to its resistance and/or resilience (Villéger et al., 2010; Wolanski and Donald, 2011; Mouchet et al., 2013; Vilar et al., 2013).

Different indexes describe and quantify the diversity of a particular community, such as number of species, evenness and/or more complex variations between species like taxonomy, phylogeny and function (Devictor et al., 2010; Granger et al., 2015; Pavoine and Bonsall, 2011; Stuart-Smith et al., 2013). Recent studies showed that traditional indexes are limited as they consider all species equivalent (Magurran, 2004; Mouchet et al., 2010; Villéger et al., 2010). The incorporation of functional features of species have generated different patterns and responses, sometimes more sensitive in detecting changes in communities (Bender et al., 2013; Buisson et al., 2013; D'agata et al., 2014; Martins et al., 2012). A functional traits-based approach may be valuable, even for applied studies focused on conservation and restoration decisions (Cadotte et al., 2011). In this context, functional diversity of communities assumes mechanistic links with niche and functional aspects of species, and thus provides a pragmatic assessment of niche mismatch/overlap between species (Cadotte et al., 2011). Functional diversity can be defined as the value and range of the functional traits of an organism present in a community (Petchey and Gaston, 2006). Functional traits are defined as “morpho-physiophenological traits which impact fitness indirectly via their effects on growth, reproduction and survival, the three components of individual performance” (Violle et al., 2007).

Despite the growing interest for functional diversity of fishes (Resetarits and Chalcraft, 2007; Halpern and Floeter, 2008; Albouy et al., 2011; Granger et al., 2015), few studies considered a traitbased approach for fish communities in tropical estuaries (Mouchet et al., 2013; Villéger et al., 2010, 2012). In addition, the general lack of standardized fisheries databases prevents the assessment of the effects of natural and/or anthropogenic factors on the fish diversity in these ecosystems. This is notably the case for estuaries in Pernambuco state, northeastern region of Brazil, with a coastline extension of 187 km encompassing 34 fishing communities distributed along 15 coastal municipalities (FIDEM, 1987). There are 17 estuarine zones, characterized by the strong presence of mangroves (Paiva and Araújo, 2010), many of which are impacted mainly by deforestation, erosion, pollutants from the sugar industry and sewage, as well as shrimp farming (Souza and Sampaio, 2001; CPRH, 2003a, 2003b). Such factors may affect aquatic communities with perturbations of the biological cycle of marine and estuarine species, loss of biodiversity and reduction in fisheries productivity (Paiva and Araújo, 2010). However, there is no study providing a traits-based approach on the functional diversity of estuarine fish in these environments.

In this context, this study assesses the functional diversity among estuaries with different levels of impacts and environmental features, using a fish functional traits-based approach. It highlights the usefulness of a recent multivariate analysis (Double Principal Coordinate Analysis - DPCoA) (Pavoine et al., 2004) to assess the species contribution in functional typology and deepen the diversity assessment of estuarine fishes. The analysis was carried out in two steps: firstly, diversity indices of species richness and the mean functional differences between individuals were quantified per sample for each estuary. Secondly, functional typology of fish assemblages among these estuaries was characterized by this multivariate analysis and compared to a traditional Principal Component Analysis to provide a baseline diagnostic for future monitoring in an ecosystem-based fishery management.

2 - Material and methods

2.1 - Study Site

Four estuaries located along the coast of Pernambuco State were studied: Itapissuma, Suape, Sirinhaém and Rio Formoso. The estuarine complex of Itapissuma is located to the north and closest to Recife and Olinda urban centres, and the most influenced by landscape modifications, industrial discharges, shrimp farming and high fishing pressure (CPRH, 2003a; Lira et al., 2010b; Gondim, 2015). The other three estuaries are located south of Recife: the Suape estuary has recently suffered the high impact of deforestation and land reclamation due to the construction of a large industrial port complex, with expected effects on both biodiversity and fisheries productivity (Paiva and Araújo, 2010). Sirinhaém and Rio Formoso are located in the southernmost area, and although located within an environmental protected area (APA Guadalupe), they are also impacted by agricultural pollution (mainly sugar cane), shrimp farming and fishing (Lira and Fonseca, 1980). Rio Formoso is also a popular tourism destination (Lira et al., 2010b).

2.2 - Data collection

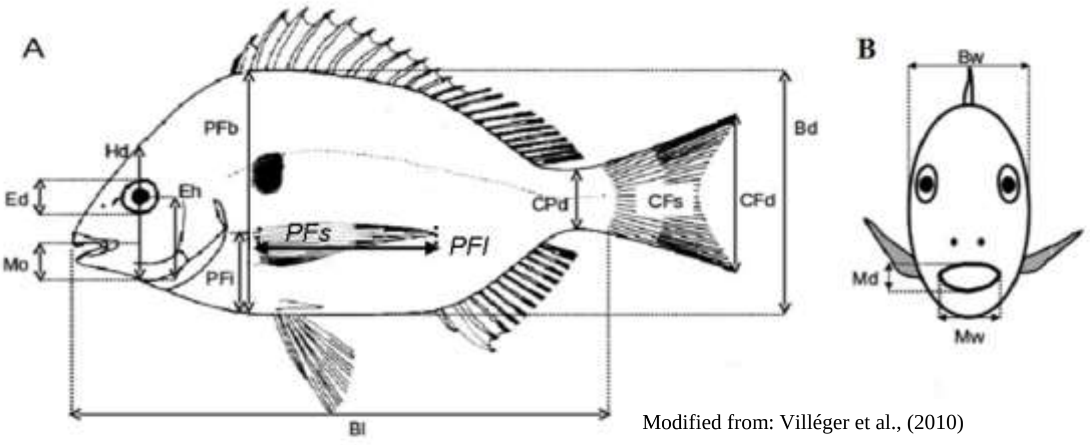
Thirty-four samples were collected at four estuaries from the Pernambuco coast (Fig. 1): Itapissuma, Suape, Sirinhaém and Rio Formoso (17, 4, 9 and 4 samples, respectively) from October 2012 to December 2014 (see dates of each sampling in the Table 2 in supplementary material). Two complementary fishing gears were used: block net and beach seine. These fishing gears were selected as, together, they cover most fish habitats: shallow coastal areas and sand/mud banks (beach seine) and flooded mangrove and channels (block net) (Mérigot et al., 2016). Due to their geomorphology and floor, in Sirinhaém and Itapissuma both fishing gear were used and, in Suape and Rio Formoso, only the block net was utilized. The number of samples per estuary varied due to the characteristics of each gear, the dependence of tidal variation, and the sampling time for a given gear. Block nets were set along the mangrove forest and channels measured from 70 to 90 m long and 15mhigh and used along the mangrove forest. Blocking initiated at the end of the high tide and continued throughout the entire ebb tide cycle (6 h). The beach seine ranged from 80 to 180 m long and 5 m high. The seining lasted no more than 20 min and it was repeated several times.

Figure 1 – Estuaries sampled along the Pernambuco Coast - Brazil

2.3 - Functional characterization of species

Two key functions, food acquisition and locomotion, that relate to twelve ecomorphological traits, assimilated to functional traits, were considered (Villéger et al., 2010). The functional traits, based on morphoanatomical measurements, of 122 species were calculated. A maximum of 10 individuals by species was randomly selected for measurement. When the 10 individuals were not available, pictures from the internet were supplied ("Fishbase website"; Froese and Pauly, 2016). The traits were computed as ratios (Albouy et al., 2011; Mason et al., 2007; Villéger et al., 2010) of each individual (Fig. 2; Table 1), assuming that intraspecific variations were smaller than interspecific variations (Dumay et al., 2004; Villéger et al., 2012). Body width (Bw), mouth width (Mw) and mouth depth (Md) were measured with an electronic calliper (precision of 0.1 mm). The other measures were performed with digital photography (Sony DSC-W830, 20.1 Mega Pixels) and the free software Image J (Schneider et al., 2012), with 0.1 mm of accuracy.

Each functional trait was standardized so that the mean was equal to 0 and standard deviation 1. Functional pairwise-distances between species were computed on standardized traits using the Euclidean distance. Only adult specimens, dominants in the samples, were used for measurements, to avoid bias associated to ontogenetic variations, since such differences could affect some species. Relationships between the functional traits and species obtained with a Principal Component Analysis are provided as Figs. 1 and 2 in the supplementary material.



Modified from: Villéger et al., (2010)

Figure 2 – Morphological measures (A): Bl body standard length, Bd body depth, CPd caudal peduncle minimal depth, CFd caudal fin depth, CFs caudal fin surface, PFi distance between the insertion of the pectoral fin to the bottom of the body, PFb body depth at the level of the pectoral fin insertion, PFl pectoral fin length, PFs pectoral fin surface, Hd head depth along the vertical axis of the eye, Ed eye diameter, Eh distance between the center of the eye to the bottom of the head, Mo distance from the top of the mouth to the bottom of the head along the head depth axis; and with an electronic caliper (B): Bw body width, Md mouth depth, Mw mouth width. Figure modified from Villéger et al. (2010).

Table 1: Functional traits of fishes. See codes of morphometric measurements in Figure 2.

	Functional traits	Formula
Food Acquisition	Oral gape surface	$\frac{Mw \ x \ Md}{Bw \ x \ Bd}$
	Oral gape shape	$\frac{Md}{Mw}$
	Oral gape position	$\frac{Mo}{Hd}$
	Eye size	$\frac{Ed}{Hd}$
Locomotion	Eye position	$\frac{Eh}{Hd}$
	Body transversal shape	$\frac{Bd}{Bw}$
	Pectoral fin position	$\frac{PFi}{PFb}$
	Aspect ratio of the pectoral fin	$\frac{PFl^2}{PFs}$
	Caudal peduncle throttling	$\frac{CFd}{CPd}$
	Aspect ratio of caudal fin	$\frac{CFd^2}{CFs}$
	Fin surface ratio	$\frac{2 \ x \ PFs}{CFs}$
	Fin surface to body size ratio	$\frac{(2 \ x \ PFs) + CFs}{{}^{\pi}/_{4} \ x \ (Bw \ x \ Bd)}$

2.4 - Statistical analyses

The diversity indices computed were species richness (*S*) and functional diversity through Rao's quadratic entropy (*Q*) (BOTTA-DUKÁT, 2005; RAO, 2010; PAVOINE, 2012) which is the mean functional distance between two randomly chosen individuals in an assemblage. It incorporates both the relative abundance of species and the functional pairwise distances between species: $Q = \sum_{j=1}^{S-1}\sum_{j=j+1}^{S} d_{ij}p_ip_j$, where: p_i is the proportion of all individuals belonging to the species *i* and *j*, and d_{ij} is the functional distance between species *i* and *j*.

The distributions and level of *S* and *Q* between the estuaries were analysed by mean of a Kruskal-Wallis test due to low sampling size and as normality assumptions were not met. When the null hypothesis of homogeneity in distributions between estuaries was rejected, the non-parametric multi-comparison post-hoc test proposed by Siegel and Castellan (1988) assessed the pairwise differences between estuaries.

To provide a functional typology of fish assemblages a multivariate analysis named *Double Principal* Coordinate Analysis (DPCoA) was used. This method was developed to compare several samples containing species that differ according to their taxonomic, phylogenetic or functional features (PAVOINE et al., 2004). Following the description of the method provided by Pavoine et al. (2004, 2013), a key step of this approach is the definition of a common Euclidean space that embeds both species and samples. To obtain this common space, a principal coordinate analysis (PCoA) is first applied to the species distances matrix (Euclidean) according to their functional traits, where each species is weighed by its global proportion over all samples. This PCoA generates a cloud of points in a geometric (Euclidean) space of orthogonal axes (i.e. principal axes), where each point represents a species. Each axis explains a certain part of the total variance (inertia) of the data, in a decreasing order from axis 1. Thus, the first axes of the space optimize the representation of the dissimilarities among species in few dimensions. Then, the samples are positioned in this space at the centroids of the species they contain (centroids are defined in terms of means of species' coordinates on the principal axes). Finally, species that mainly characterized samples by their higher proportions can be deduced according to their position on each axis.

This analysis gives additional information regarding the overall diversity in the study area (i.e. γ-diversity) decomposed in between-assemblages (i.e. β-diversity) and within-assemblages (i.e. α-diversity), where these diversities can directly be quantified by the quadratic entropy *Q* (PAVOINE et al., 2004). A previous application of this method on fish dealt with phylogenetic (PAVOINE et al., 2009) and taxonomic distances (Mérigot et al., in press), and this is the first use on functional diversity of fishes, though it has been applied on other groups (e.g. birds, Pavoine et al. 2004). Differences between the four estuaries in the typology of DPCoA (centroid locations drawn from samples belonging to each estuary) have been assessed by a Kruskal Wallis test on sample coordinates of the axes. For comparison purposes, a Principal Component Analysis (PCA) based only on species abundance was also performed (abundance data were log transformed prior to the analysis due to the relative high values of some species).

All the statistical analyses and diversity indices mentioned above were performed using R version 3.2.1 (YAN et al., 2011). The packages used were “ade4’’ (DRAY and DUFOUR, 2007), “vegan” (OKSANEN et al., 2010) and “FD” (LALIBERTÉ et al., 2015).

3 - Results

A total of 48,754 individuals were sampled, belonging to 122 species, 80 genera and 36 families. Regarding the capture in biomass, it was 459.8 kg in Itapissuma, followed by Sirinhaém with 177.8 kg, Rio Formoso with 125.8 kg and Suape with 108.5 kg. The 12 morphometric measurements were registered for 663 individuals.

Regarding species richness S, Itapissuma showed the highest values in mean per sample compared to the other estuaries ($Chi^2=13.08$, d.f.=3, $p < 0.05$) (see also quartiles distributions on Fig. 3A). The mean values of S per sample were 29.82 ± 9.62 species in Itapissuma, 18.5 ± 5 species in Suape, 16.2 ± 7.9 species in Sirinhaém and 14.5 ± 4.6 species in Rio Formoso. Interestingly, the quadratic entropy *Q* showed a quite similar overall pattern to S, but with lower variability within each estuary (Fig. 3B). The mean functional distances between two randomly chosen individuals in a sample in Itapissuma was the highest with *Q*=0.052 ± 0.02, whereas lower values were obtained in Sirinhaém (i.e. 0.004 ± 0.002, see also quartiles distributions in Fig. 3B) (Chi^2=18.33, d.f.=3, $p < 0.01$). This highlights that, while samples from Itapissuma showed a higher average richness, they encompassed more functional distant individuals than the three other estuaries.

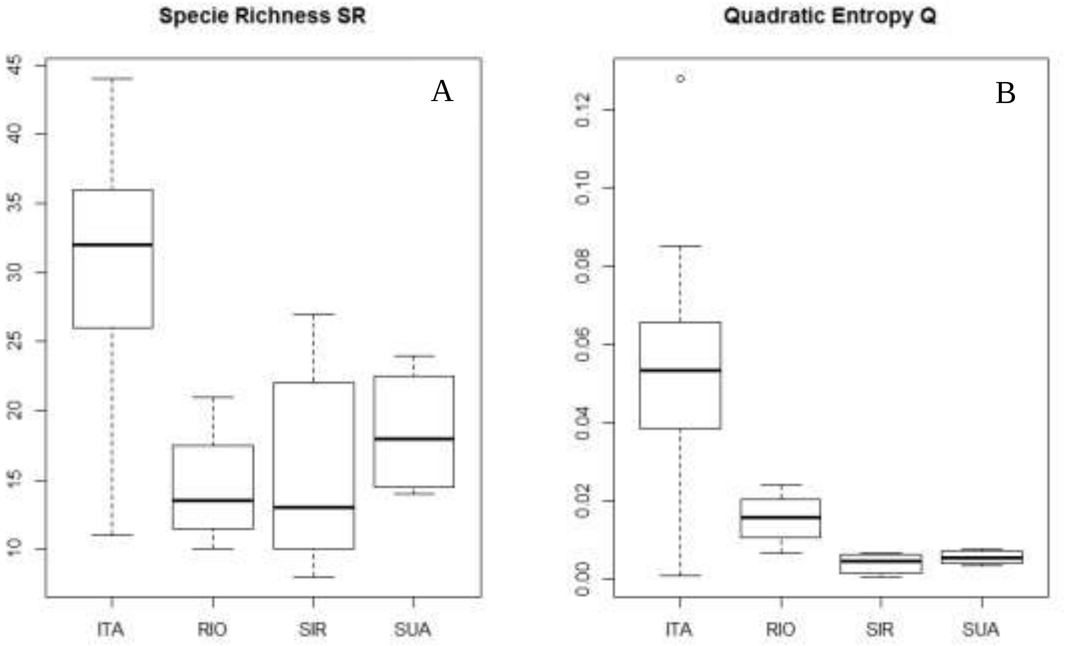


Figure 3 – Boxplots of specie richness *S* and functional diversity *Q* indices, according to the estuary: Itapissuma ITA, Rio Formoso RIO, Sirinhaém SIR, Suape SUA.

The first factorial plane of the DPCoA analysis, based on both species abundances and functional distances, explained 68.70% of the total inertia (axis 1: 55.0%, axis 2: 13.7%), whereas those of a traditional PCA (based only on species abundances) explained only 17.4%. The analysis revealed a difference in functional typology between Itapissuma assemblages and the other three estuaries on the first axis (see barycentre locations on Fig. 4); supported by a Kruskal-Wallis test performed on the samples coordinates of axis 1, Chi^2=26.04, d.f.=3, $p < 0.001$, followed by the non-parametric multiple comparison test proposed by Siegel and Castellan (1988). In contrast, axis 2 differentiated the Rio Formoso assemblages from the three other estuaries (Chi^2=10.65, d.f.=3, $p < 0.05$, Fig. 4).

The typology was related to the abundance and function of particular species (Fig. 5). The first axis discriminated Sirinhaém, Suape and Rio Formoso on the positive side, from Itapissuma located at the center of the axis (Fig. 4). The positions of the species related to the positions of estuaries and their habitats (Fig. 5). The list of species and their abundance are given in Supplementary material. *Gobionellus stomatus* (gob.sto) had negative coordinates on the first principal axis, and is therefore related to Itapissuma's habitat due to a relatively higher abundance and being only sampled there (Fig. 5, axis 1). Conversely, *Diapterus auratus* (dia.aur) had positive coordinates on the first and second axes, and thus could be associated with the Rio Formoso's estuary where it had the highest abundance (Fig. 5, axes 1 and 2). Finally, *Mugil curema* (mug.cur) contributed to the positive part of the axis 1 and the negative part of axis 2. Whereas sampled in Itapissuma, it also contributed to the Sirinhaém and Suape's functional typology where it appeared in most of the samples.

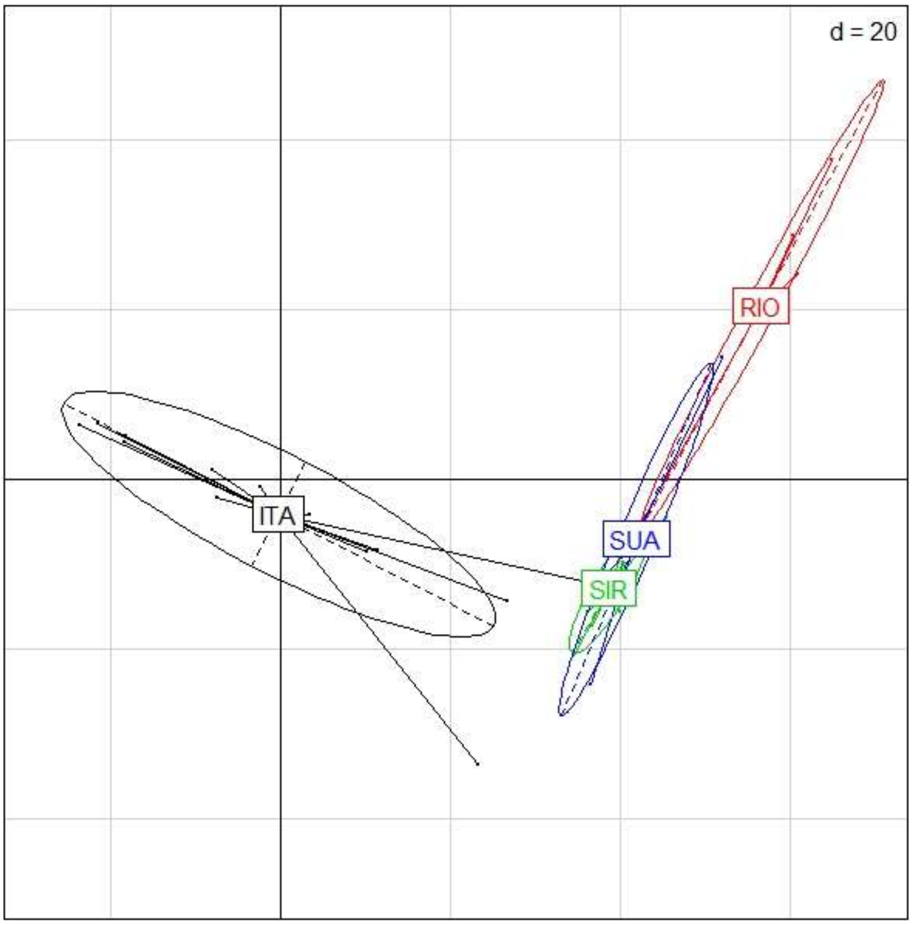


Figure 4 – Projections of samples from the Double Principal Coordinate Analysis (DPCoA), first factorial plane explaining 68.7 % of the total inertia (axis 1: 55.0%, axis 2: 13.7%) for the different estuaries: Itapissuma ITA, Sirinhaém SIR, Suape SUA, Rio Formoso RIO. Coloured lines link samples to centroid of the estuary they belong. An ellipse encompasses 67% of samples for a given estuary.

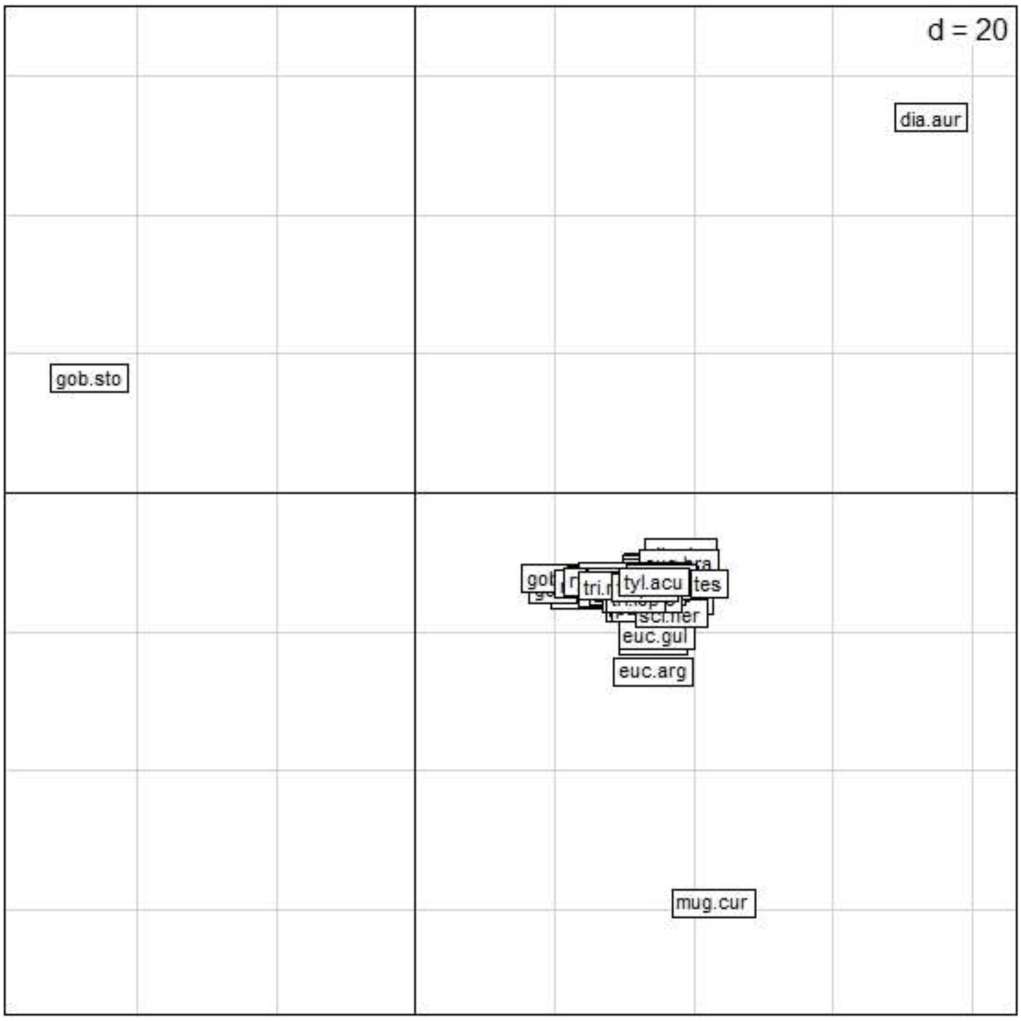


Figure 5 – Projections of species from the Double Principal Coordinate Analysis (DPCoA), first factorial plane explaining 68.7% of the total inertia (axis 1: 55.0%, axis 2: 13.7%). Species' codes are provided in Table 1 in the supplementary material.

4 - Discussion

One means of ensuring accurate representation of the species assemblage is through effective sampling design (Kwak and Peterson, 2007). The heterogeneity of the estuaries makes standardization of sampling strategies and gears difficult. According to Mourão et al. (2014), although differences in the fish composition between estuaries may arise due to differential effort (variability in sampling intervals, years and gears), clear trends are detected in biodiversity studies, as for in our analysis. In our study, geomorphology was a key factor in the choice of the fishing gears, which were adapted to most of the fish habitats in the area. The beach seine was used in shallow coastal areas and sand/mud banks whereas the block net was used in all estuaries within flooded mangrove and channels. Thereby, a multi-gear approach was used in order to provide the most complete assessment of fish assemblage (Kwak and Peterson, 2007).

Assessing both species richness and functional diversity can reveal the degree of redundancy and complementarity of species among community that may result from human impacts and environmental conditions (Villéger et al., 2010, 2012). In this study, the estuaries of Sirinhaém, Suape and Rio Formoso had the lowest levels of functional diversity Q and species richness S per sample (Fig. 3). It suggests that assemblages in these areas encompassed predominant functional closely related individuals compared to Itapissuma. This similarity may occur due to some similar features of these estuaries which are geographically close (Paiva and Araújo, 2010) and might result from environmental filtering.

An important structuring factor on the functional composition of assemblages could be the environmental geomorphology. Suape, Sirinhaém and Rio Formoso have a similar structure with a large inner region and a narrow outermost area (Silva et al., 2011), as well as have a high incidence of mangroves (CPRH, 2003c). These characteristics are quite different from those of Itapissuma, which present a very irregular morphology (da Silva et al., 2011), a width ranging from 400 to 1500 m, depths of 2-5 m at low tide and is affected by an important discharge of organic matter from different rivers (CPRH, 2003a). Furthermore, muddy substrate and mangrove vegetation predominate in the Itapissuma estuary (da Silva et al., 2011), enabling the formation of microhabitats that are fundamental for juveniles to settle, complete their life cycle and feed on specific prey available (Goodall, 1986). Overall, these characteristics may produce a more diverse habitat and lead ichthyofauna assemblages to be richer with species morphologically distinct from the other three estuaries.

The DPCoA analysis highlighted three species *Gobionellus stomatus*, *Diapterus auratus* and *Mugil curema* which are closely related to the estuaries Itapissuma, Rio Formoso and Sirinhaém/Suape, respectively. Itapissuma showed higher levels of functional diversity *Q* and species richness (Fig. 3), and differed in functional typology (Fig. 4). These trends were notably associated to the high level of abundance of *Gobionellus stomatus* (gob.sto) in this estuary when no specimen was sampled elsewhere. The Gobiidae family are directly associated with nutrients-rich areas, i.e. with high availability of food, and in muddy estuaries (Menezes and Figueiredo, 1985), characteristics found in Itapissuma, the largest and most productive estuary of the Pernambuco coast (Lira et al., 2010a). In addition, the Gobiidae family occupies an important place in the trophic chain of coastal environments, due to their role at the initial links in the food chain (Bemvenutti, 1987; Andreata et al., 1997; El-Deir, 2005).

Diapterus auratus (dia.aur) is an abundant species and contributes to the functional typology of the estuary of Rio Formoso. This species can be classified as a marine, coastal and estuarine fish, living near the bottom where they spawn and feed (Menezes and Figueiredo, 1980; Chaves and Otto, 1998; Nelson, 2006). Moreover, as a marine migrant, it is tolerant to variations in salinity and can be found in estuaries under strong marine influence (Figueiredo and Menezes, 1980; Paiva et al., 2008; Vilar et al., 2011). Rio Formoso is an estuary near the coast and thus is under an important influence of marine habitats nearby (reefs and bays).

Species associated to particular niches by having a specific combination of traits are named specialist species (e.g. Villéger et al., 2010). They are supposed to be more sensitive to environmental changes because an impact or loss of these niches can affect them, while generalist ones may be less sensitive by occupying the most common niches (Jiguet et al., 2007; Villéger et al., 2010). Thus, the difference in functional features observed between assemblages of Itapissuma compared to those of Sirinhaém, Suape and Rio Formoso may also result from past and current habitat/niche degradation, impacts of pollutants from the sugar industry and sewage, as well as from shrimp farming, that occurred for a longer time and higher levels (CPRH, 2003a, 2003b; Lira and Fonseca, 1980). The other areas are also subject to some common impacts (agriculture run-off, deforestation, shrimp farming) and have singular conditions, such as the port development in Suape and presence of protected areas in Rio Formoso (APA Guadalupe (Lira et al., 2010b)) that may affect biodiversity and fish production (Paiva and Araújo, 2010). These factors may determine different scenarios in the future for these areas, as well as new environmental policies, as the recent changes in the Brazilian Forest code and foreseen impacts to mangrove areas (Soares-Filho et al., 2014). Such impacts can influence water transparency, chemical characteristics (i.e. pH), type of organic matter and availability of food sources, which in turn affect estuarine fish communities (Camargo and Isaac, 2003). Furthermore, the sensitivity of specialist species to environmental changes suggests that particular caution of potential human impacts should be considered.

Overall, combining diversity indices and functional typology through a multivariate analysis, the DPCoA (Pavoine et al., 2004, 2013), allowed to deepen diversity assessment in functional patterns of assemblages for estuarine fishes. In estuarine ecosystems, a common approach is to consider fishes by ecological and feeding guilds (Elliott et al., 2007). It provides through functional groups a rough, synthetic and qualitative classifications of species (e.g. marine migrant; piscivorous etc.), that have been applied successfully worldwide (Bremner et al., 2003). While more time consuming, our results suggest that a quantitative traits-based approach can detect different patterns. This approach has allowed to highlight interesting responses of fish communities in a tropical estuary faced to perturbations (Villéger et al., 2010, 2012). In order to assess their complementary, it should thus be accurate in forthcoming studies to directly compare the results of DPCoAs computed from the two kind of information (functional distances between species could be computed on qualitative guilds based the Gower coefficient (Gower, 1971)).

The assessment of functional diversity of fish can be accurate in the context of environmental and anthropogenic perturbations (Villéger et al., 2010; D'agata et al., 2014). Considering that the Brazilian coast encompasses a wide range of contrasted conditions, such analyses could provide an interesting framework for comparison between the estuarine ecosystems along the coast. This will also be accurate at a global scale, for which species richness patterns and its predictors have been recently investigated (Pasquaud et al., 2015; Vasconcelos et al., 2015), and that should be extended to functional diversity.

5 – Acknowledgments

This study was partially funded by INCT (National Institute of Science and Technology) in Tropical Marine Environments - INCT-AmbTropic, and by CAPES (Coordination for the Improvement of Higher Education Personnel) for providing scholarship to the first author. We thank the members of Bioimpact Lab and Coastal Reefs group, which were essential for the proper conduct of this research.

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SUPPLEMENTARY MATERIAL

Functional diversity of fish in tropical estuaries: a traits-based approach of communities in Pernambuco, Brazil

Table S1 – Table with information about species composition, their respective code and their abundance for each estuary.

Family	*Scientific name*	Code	Itapissuma	Sirinhaém	Suape	Rio Formoso
Acanthuridae						
	Acanthurus chirurgus	aca.chi	3	-	-	-
Achiridae						
	Achirus lineatus	ach.lin	996	132	1	-
	Achirus declivis	ach.dec	16	7	-	-
	Trinectes microphthalmus	Tra.goo	-	1	-	-
	Trinectes paulistanus	tri.pau	59	19	-	-
Ariidae						
	Sciades herzbergii	sci.her	13	222	8	10
	Aspistor luniscutis	asp.lun	1	78	-	-
	Notarius grandicassis	not.gra	-	1	-	-
	Cathorops agassizii	cat.aga	2			
	Cathorops spixii	cat.spi	-	5	-	-
Atherinopsidae						
	Membras dissimilis	mem.dis	1	-	-	-
	Atherinella brasiliensis	ath.bra	10	-	-	-
Batrachoididae						
	Thalassophryne nattereri	tha.nat	37	-	-	-
	Batrachoides surinamensis	bat.sur	19	-	-	-
Belonidae						
	Strongylura timucu	str.tim	1	-	-	1
	Tylosurus acus acus	tyl.acu	3	-	-	-
Carangidae						
	Caranx latus	car.lat	50	29	36	66
	Caranx crysos	car.cry	2	-	-	29
	Oligoplites saurus	oli.sau	15	-	2	18
	Caranx hippos	car.hip	158	10	10	11
	Oligoplites palometa	oli.pal	8	5	-	10
	Selene vomer	sel.vom	-	-	6	4
	Selene setapinnis	sel.set	-	-	3	-
	Trachinotus falcatus	tra.fal	-	2	-	-
	Carangoides bartholomaei	car.bar	-	3	-	-
	Selene vomer	sel.vom	3	1	-	-
	Chloroscombrus chrysurus	chl.chr	72	1	-	-
	Oligoplites saliens	oli.sal	6	-	-	-
Centropomidae						
	Centropomus undecimalis	cen.und	57	132	36	64

	Centropomus parallelus	cen.par	90	119	113	60
	Centropomus mexicanus	cen.mex	-	-	65	44
	Centropomus pectinatus	cen.pec	5	25	7	41
	Centropomus ensiferus	cen.ens	-	26	-	-
Chaetodontidae						
	Chaetodon ocellatus	cha.oce	1	-	-	-
Clupeidae						
	Opisthonema oglinum	opi.ogl	124	25	-	-
	Sardinella brasiliensis	sar.bra	11	-	-	-
	Rhinosardinia bahiensis	rhi.bah	191	-	-	-
	Harengula clupeola	har.clu	45	-	-	-
	Chirocentrodon bleekerianus	chi.ble	61	-	-	-
	Lile piquitinga	lil.piq	1	-	-	-
Cynoglossidae						
	Symphurus tessellatus	sym.tes	35	-	-	-
Echeneidae						
	Echeneis naucratis	ech.nau	-	-	-	1
Eleotridae						
	Guavina guavina	gua.gua	13	4	-	-
Elopidae						
	Elops saurus	elo.sau	2	-	-	13
Engraulidae						
	Anchoa lyolepis	anc.lyo	8	-	-	-
	Cetengraulis edentulus	cet.ede	1622	1	-	-
	Anchovia clupeoides	anc.clu	175	-	-	-
	Anchoa marinii	anc.mar	13	-	-	-
	Lycengraulis grossidens	lyc.gro	97	-	-	-
	Anchoa spinifer	anc.spi	63	1	-	-
	Anchoa tricolor	anc.tri	259	-	-	-
Ephippidae						
	Chaetodipterus faber	cha.fab	21	7	4	8
Gerreidae						
	Diapterus auratus	dia.aur	296	145	195	1835
	Eugerres brasilianus	eug.bra	9	-	45	147
	Diapterus rhombeus	dia.rho	652	116	12	146
	Eucinostomus melanopterus	euc.mel	14	10	-	33
	Eucinostomus argenteus	euc.arg	2194	25	2	5
	Eucinostomus gula	euc.gul	1687	32	2	-
	Eucinostomus havana	euc.hav	35	-	-	-
Gobiidae						
	Gobionellus stomatus	gob.sto	21430	-	-	-
	Gobionellus oceanicus	gob.oce	1927	-	-	-
	Bathygobius soporator	bat.sop	91	-	-	-
	Ctenogobius boleosoma	cte.bol	179	-	-	-
	Gobionellus stigmaticus	gob.sti	2073	-	-	-
	Ctenogobius shufeldti	cte.shu	99	-	-	-

	Microgobius meeki	mic.mee	88	-	-	-
	Ctenogobius smaragdus	cte.sma	686	-	-	-
	Evorthodus lyricus	evo.lyr	1	-	-	-
Haemulidae						
	Pomadasys crocro	pom.cro	2	12	-	3
	Pomadasys corvinaeformis	pom.cor	-	1	-	1
	Haemulon plumierii	hae.plu	-	-	2	-
	Genyatremus luteus	gen.lut	3	2	-	-
	Conodon nobilis	con.nob	-	1	-	-
	Haemulon parra	hae.par	16	-	-	-
Hemiramphidae						
	Hyporhamphus unifasciatus	hyp.uni	94	-	-	-
	Hemiramphus brasiliensis	hem.bra	20	-	-	-
Lobotidae						
	Lobotes surinamensis	lob.sur	1	-	-	-
Lutjanidae						
	Lutjanus alexandrei	lut.ale	-	-	101	10
	Lutjanus jocu	lut.joc	82	42	19	2
	Lutjanus synagris	lut.syn	66	-	7	-
	Lutjanus alexandrei	lut.ale	190	3	-	-
	Lutjanus analis	lut.ana	315	-	4	-
	Lutjanus cyanopterus	lut.cya	-	-	1	-
Megalopidae						
	Megalops atlanticus	meg.atl	-	1	-	-
Mugilidae						
	Mugil curema	mug.cur	1970	83	37	104
	Mugil curvidens	mug.curv	-	6	46	17
	Mugil liza	mug.liz	29	2	-	1
	Mugil hospes	mug.hos	2	-	-	-
	Mugil brevirostris	mug.bre	-	-	7	-
Muraenidae	*Gymnothorax funebris*	gym.fun	-	1	-	2
	Gymnothorax ocellatus	gym.oce	1	-	-	-
Paralichthyidae	*Paralichthys brasiliensis*	para.bra	6	-	22	-
	Etropus longimanus	etr.lon	14	-	-	-
	Citharichthys spilopterus	cit.spi	377	5	-	-
	Etropus crossotus	etr.cro	104	-	-	-
Polynemidae	*Polydactylus virginicus*	pol.vir	3	5	-	-
Scaridae	*Sparisoma radians*	spa.rad	160	-	-	-
Sciaenidae	*Bairdiella ronchus*	bai.ron	54	87	-	36
	Stellifer brasiliensis	ste.bra	-	-	-	35
	Stellifer microps	ste.mic	1	1	-	-

	Micropogonias furnieri	mic.fur	1	17	-	-
	Menticirrhus americanus	men.ame	-	3	-	-
	Cynoscion leiarchus	cyn.lei	12	-	-	-
	Cynoscion acoupa	cyn.aco	-	1	-	-
	Ophioscion punctatissimus	oph.pun	-	1	-	-
	Isopisthus parvipinnis	iso.par	1	11	-	-
	Cynoscion virescens	cyn.vir	12	-	-	-
	Stellifer stellifer	ste.ste	2	-	-	-
	Macrodon ancylodon	mac.anc	2	-	-	-
Serranidae	*Mycteroperca bonaci*	myc.bon	3	-	1	-
	Rypticus randalli	ryp.ran	4	-	-	-
	Epinephelus itajara	epi.ita	-	-	3	-
	Epinephelus adscensionis	epi.ads	1	-	1	-
	Epinephelus marginatus	epi.mar	1	-	-	-
Sparidae	*Archosargus rhomboidalis*	arc.rho	1266	1	1	3
	Archosargus probatocephalus	arc.pro	23	-	-	-
Sphyraenidae	*Sphyraena guachancho*	sph.gua	10	-	6	-
	Sphyraena barracuda	sph.bar	15	1	19	-
Stromateidae	*Peprilus paru*	pep.par	19	11	-	-
Synodontidae	*Synodus foetens*	syn.foe	8	-	-	-
Tetraodontidae	*Sphoeroides testudineus*	sph.tes	1778	237	256	277
	Colomesus psittacus	col.psi	6	-	-	-
	Lagocephalus laevigatus	lag.lae	2	3	5	-
	Sphoeroides greeleyi	sph.gre	412	-	-	-
Trichiuridae	*Trichiurus lepturus*	tri.lep	-	9	-	-
Triglidae	*Prionotus punctatus*	pri.pun	2	-	-	-
TOTAL			42912	1720	1085	3037

Table S2 - Data collection dates

Estuary	Fishing gear/Date
Itapissuma	**Block net**
Itapissuma	January-13
Itapissuma	May-13
Itapissuma	June-13
Itapissuma	August-13
Itapissuma	November-13
Itapissuma	March-14
Itapissuma	May-14
Itapissuma	June-14
	Beach Seine
Itapissuma	January-13
Itapissuma	May-13
Itapissuma	August-13
Itapissuma	October-13
Itapissuma	November-13
Itapissuma	December-13
Itapissuma	January-14
Itapissuma	February-14
Itapissuma	March-14
Itapissuma	April-14
Itapissuma	May-14
Itapissuma	June-14
Itapissuma	July-14
Itapissuma	August-14
Itapissuma	September-14
Itapissuma	December-14

Estuary	Fishing gear/Date
Sirinhaém	**Beach seine**
Sirinhaém	March-13
Sirinhaém	May-13
Sirinhaém	August-13
Sirinhaém	November-13
Sirinhaém	February-14
Sirinhaém	May-14
	Block net
Sirinhaém	October-12
Sirinhaém	February-13
Sirinhaém	May-13
Sirinhaém	August-13
Sirinhaém	November-13
Sirinhaém	February-14
Sirinhaém	June-14
Suape	**Block net**
Suape	June-13
Suape	August-13
Suape	November-13
Suape	March-14
Rio Formoso	**Block net**
Rio Formoso	May-13
Rio Formoso	August-13
Rio Formoso	November-13
Rio Formoso	March-14

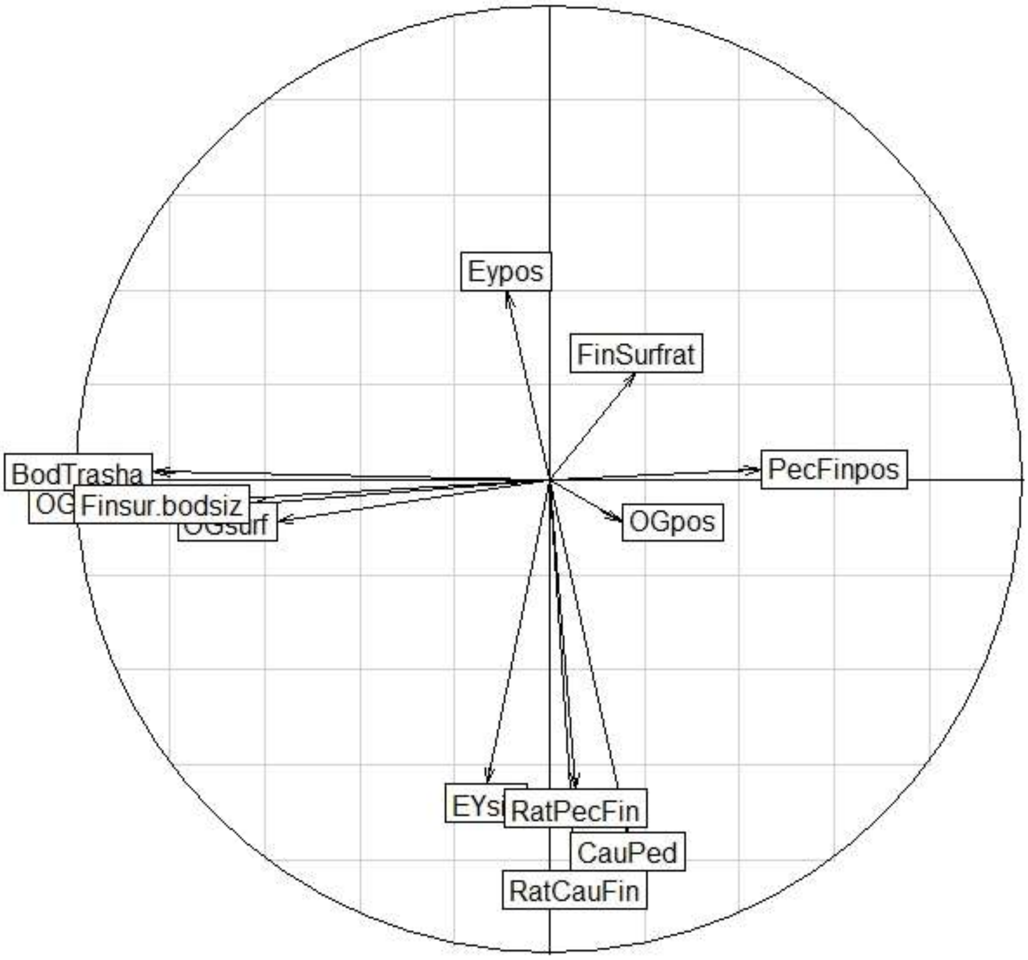

Figure S1 - Projections of species traits from the Principal Component Analysis (PCA), first factorial plane explaining 39.82% of the total inertia (axis 1: 20.64%, axis 2: 39.82%) for the different traits: OGsurf: Oral gape surface; OGshap: Oral gape shape; OGpos: Oral gape position; EYsiz: Eye size; Eypos: Eye position; BodTrasha: Body transversal shape; PecFinpos: Pectoral fin position; RatPecFin: Aspect ratio of the pectoral fin; CauPed: Caudal peduncle throttling; RatCauFin: Aspect ratio of caudal fin; FinSurfrat: Fin surface ratio; Finsur.bodsiz: Fin surface to body size ratio.

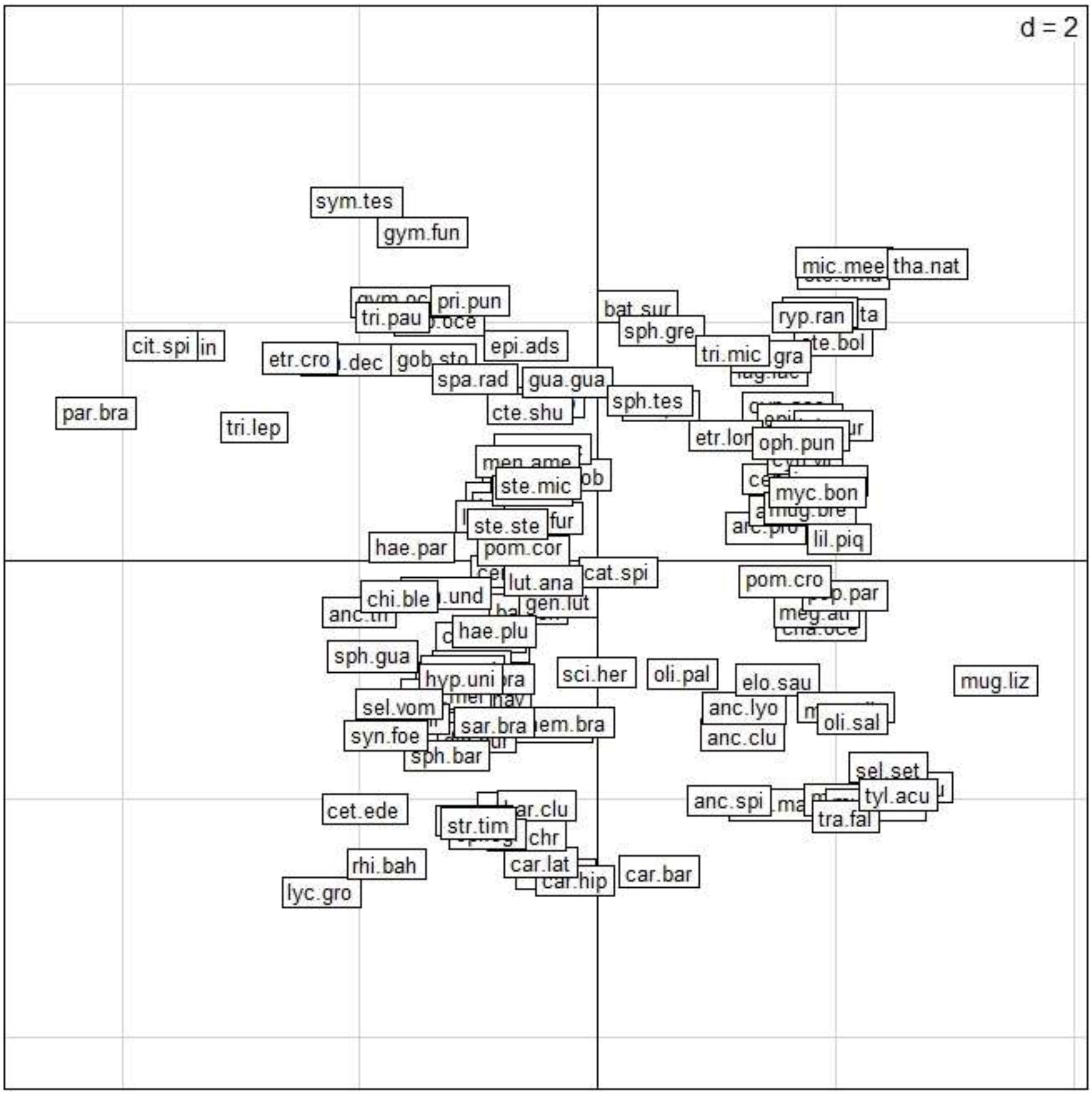

Figure 2 - Projections of species from the Principal Component Analysis (PCA), first factorial plane explaining 39.82% of the total inertia (axis 1: 20.64%, axis 2: 39.82%) for the different traits. The species codes are provided in the Table 1 of supplementary material.

3. 3 - Artigo científico III

Artigo científico a ser encaminhado a Revista **Fisheries Research**

Análise de produtividade e susceptibilidade para espécies de peixes capturadas em estuários tropicais

Silva-Júnior, C. A. B.

Resumo

Analisar a sustentabilidade de pescarias com caráter multiespecífico vem sendo um grande desafio para os pesquisadores uma vez que a natureza indiscriminada dessas pescarias tem um grande potencial para afetar a população das espécies capturadas como bycatch. As pescarias realizadas em ambientes tropicais são as mais afetadas, sendo relatada a captura de uma grande diversidade de organismos, que vão desde invertebrados móveis ou sésseis, elasmobrânquios, tartarugas e várias espécies de peixes. Nos últimos anos, a Análise de Produtividade e Susceptibilidade ou "*Productivity and Susceptibility Analysis (PSA)"* vem sendo amplamente utilizada, principalmente para pescarias "pobre em dados", com o objetivo de se estimar a vulnerabilidade das espécies exploradas nas mais diversas atividades pesqueiras. Este trabalho teve como objetivo estimar a vulnerabilidade da comunidade de peixes capturados pela pescaria de camboa em dois estuários da costa do estado de Pernambuco, utilizando o PSA como método, identificando também as espécies prioritárias de manejo, considerando-se os impactos advindos da atividade pesqueira e os níveis de resiliência das mesmas. Para isso, foram selecionados onze parâmetros relacionados com a história de vida das espécies analisadas, sendo cinco referentes à susceptibilidade (%>L50, Categoria comercial, Grupos funcionais, Frequência de ocorrência e Z/k) e seis à produtividade (L50, Lmax, k, Nível trófico, r e L50/Lmax). Foram capturados um total de 5.375 indivíduos. As espécies menos vulneráveis foram *Cetengraulis edentulus*, *Eucinostomus argenteus* (para as áreas do Canal de Santa Cruz e Barra de Sirinhaém) e *Harengula clupeola* que, por serem espécies de crescimento rápido e tamanho de primeira maturação relativamente baixo, possuem uma maior resiliência. Considerando o efeito da pescaria de camboa na vulnerabilidade dos peixes, as espécies de Centropomidae (*C. undecimalis* e *C. parallelus*) mostraram 100% de probabilidade de serem classificadas como espécies de alto risco, seguidas de *Lutjanus alexandrei* com 50%. O oposto pode ser observado para *Harengula clupeola* a qual obteve uma probabilidade de 80% de ser de baixo risco. A PSA é um método promissor para a análise de riscos ecológicos na atividade pesqueira. Considerando todos os aspectos discutidos neste trabalho, a elaboração das estratégias de manejo deve sempre ser baseadas na estrutura de todo o ecossistema, não sendo aconselhável levar em consideração apenas algumas espécies. Para as áreas estuarinas aqui estudadas, sugere-se a implementação de mecanismos com o intuito de diminuir o impacto da pesca sobre as populações que ainda não atingiram sua maturidade sexual – assim como para aquelas que vem sofrendo um forte impacto causado pela atividade pesqueira.

Palavras-chaves: Análise de risco ecológico; PSA; vulnerabilidade; ictiofauna; estuário.

1. **INTRODUÇÃO**

Estuários e manguezais estão entre os habitats costeiros com maior importância ecológica, sendo reconhecidos como relevantes zonas de berçário (NAGELKERKEN et al., 2001) e importantes zonas de refúgio e migração de várias espécies de peixes em diferentes estágios do seu ciclo de vida (BARLETTA e BLABER, 2007). Entretanto, dados os múltiplos impactos antrópicos a que estão sujeitos esses ambientes, os estuários apresentam um elevado nível de degradação refletido em alterações na composição e funcionamento da biodiversidade (HALPERN et al., 2007, 2008; BORJA et al., 2012). A pesca excessiva configura-se com um dos grandes impactos nas áreas estuarinas, uma vez que a sobreexplotação, a degradação e a perda do habitat natural estão entre as principais ameaças (BLABER et al., 2000; HALL e MAINPRIZE, 2004, REYNOLDS et al., 2005).

Apesar de apresenta uma baixa produtividade pesqueira, a pesca do Nordeste do Brasil tem uma grande importância socioeconômica e, a pesca artesanal nos estuários de Pernambuco é responsável pela maior parte da produção do pescado do estado, uma vez que a mesma está associada a regiões ricas em manguezais (LIRA et al., 2010a). Dentre os diversos tipos de modalidades de pesca, a pesca de camboa é tradicionalmente praticada em zonas estuarinas, tanto no estado de Pernambuco (SILVA-JÚNIOR et al., 2016; MÉRIGOT et al., 2017), quanto em outras áreas das regiões norte e nordeste do Brasil (PAIVA ET AL., 2009; GIGLIO e FREITAS, 2013). Dentre as características apresentadas por essa modalidade de pesca, destaca-se a baixa seletividade (GIGLIO e FREITAS, 2013), ocasionando a captura de uma variedade de espécies e de elevado número de indivíduos ainda em fase juvenil (MÉRIGOT et al., 2017).

Atualmente, as estratégias de manejo pesqueiro são voltadas principalmente para manutenção de níveis ótimos de captura exclusivamente das espécies alvo (HERNÁNDEZ AGUADO et al., 2016). A aplicação de uma Abordagem Ecossistêmica das Pescarias (EAF)

(GARCIA e COCHRANE, 2005; CORRALES et al., 2015), é um quadro bastante efetivo para o gerenciamento ecossistêmico, uma vez que é levado em consideração o conhecimento e as incertezas sobre os componentes bióticos, abióticos e humanos dos ecossistemas, assim como suas interações, aplicando uma abordagem integrada das pescarias dentro de limites ecologicamente significativos (GARCIA et al., 2003). Enfoque como estes, que considera não somente as espécies alvo, mas também a fauna acompanhante, são raros (REYNOLDS et al., 2001; DULVY et al., 2004), principalmente no Nordeste do Brasil.

A Avaliação de Risco Ecológico (ERA), quadro hierárquico de avaliação de risco, foi adotada pelo Conselho de Manejo Marinho (Marine Stewardship Council – MSC) e é uma das ferramentas que faz parte da Abordagem Ecossistêmica da Pesca (FAO-EAF) (DUNSTAN et al., 2016). Um dos grandes desafios nos últimos anos é a realização de estudos relacionados à análise de risco em pescarias com deficiência de dados (FLETCHER et al., 2002; ASTLES et al., 2006), a qual se baseia em atributos ou propriedades dos organismos aquáticos, necessários para avaliar o estado em que se encontram os estoques explotados (ASTLES et al., 2006, 2009). Devido à falta de informações, os impactos sobre as populações pesqueiras muitas vezes passam despercebidos, chegando a mascarar declínios populacionais em determinadas regiões (DULVY et al., 2000).

O conceito de sustentabilidade dentro da atividade pesqueira tem sido bastante utilizado como referência para estabelecer níveis adequados de captura (KOOIMAN et al., 2005), mas também para avaliar riscos de um recurso sujeito à pesca, considerando vários aspectos ecológicos, econômicos ou sociais (ASTLES et al., 2006). Nos últimos anos, métodos como a Análise de Produtividade e Susceptibilidade ou “*Productivity and Susceptibility Analysis* (PSA)”, um procedimento de análise de risco semiquantitativa, vem sendo amplamente utilizados, principalmente para pescarias pobre em dados, com o objetivo de se estimar a vulnerabilidade das espécies exploradas nas mais diversas atividades pesqueiras (STOBUTZKI et al., 2001a; FAO, 2008; ASTLES et al., 2009; HOBDAY et al., 2011; ZHOU et al., 2016; LUCENA-FRÉDOU et al., 2017). A PSA tem como base a produtividade biológica, a qual está relacionada às características de vida de um estoque, e à susceptibilidade dos organismos serem capturados por um determinado tipo de pescaria. Além disso, esta análise prioriza o monitoramento, a avaliação e a gestão dos estoques pesqueiros daquelas espécies que possuem um maior risco de um possível colapso do estoque (HOBDAY et al., 2011). No Brasil, esta metodologia foi utilizada por Visintin e Perez (2016) para espécies capturadas na pesca de emalhe-de-fundo na região sudeste do país, e por Lucena-Frédou et al. (2017) para a pesca de espinhel direcionado para atuns e afins no Atlântico Sul. Apesar do destaque deste método ao redor do mundo, o seu uso ainda é escasso no Brasil, principalmente quando se trata das zonas estuarinas da região nordeste do país, com pescarias de pequena escala e menor valor econômico, apesar de sua grande relevância social.

A pesca de camboa em Pernambuco é um exemplo típico de pesca artesanal em ambiente tropical, na qual há uma grande carência de avaliação dos estoques explotados, decorrente principalmente da ausência de informação sobre a captura, o esforço e os principais parâmetros da história de vida das espécies. Sendo assim, a PSA é vista com uma alternativa bem adaptada para condições pobre em dados, pois podem ser utilizados na avaliação da vulnerabilidade e risco de colapso perante a atividade pesqueira, mesmo não havendo uma compreensão mais completa das relações causa/efeito dos impactos causados pelas pescarias (WALKER et al., 2002; MCFADDEN, 2007).

Este trabalho tem como principal objetivo estimar a vulnerabilidade da comunidade de peixes capturados pela pescaria com rede de camboa através da uma Análise de Produtividade e Susceptibilidade, em duas localidades estuarinas da costa do estado de Pernambuco, identificando também espécies prioritárias de manejo, considerando os impactos advindos da atividade pesqueira e os níveis de resiliência das mesmas.

2. **Material e Métodos**

2.1 *Área de estudo*

A coleta de dados foi realizada em dois estuários localizados ao longo da costa do estado de Pernambuco: zona estuarina do Canal de Santa Cruz (Itapissuma) e na Barra de Sirinhaém (Figura 1). O complexo estuarino do Canal de Santa Cruz está localizado no litoral norte do estado, ficando próximo a grandes centros urbanos da Região Metropolitana do Recife, sofrendo grandes impactos causados pela alteração de paisagem, descarga de esgotos, fazendas de cultivo de camarão e grande pressão pesqueira (CPRH, 2003; LIRA et al., 2010b; GONDIM, 2015). Está região é caracterizada por ser um grande complexo estuarino em com seu canal em formato de U, o qual é conectado com o mar através de várias entradas. Na parte interna do canal o substrato é dominado principalmente pela lama, e à medida que atinge maiores profundidade próximo ao mar, o substrato vai mudando para areia fina (Silva, 2004).

A região estuarina de Barra de Sirinhaém fica localizada no litoral sul do estado, sendo formado pelo rio Sirinhaém e outros três pequenos afluentes (Arrumador, Trapiche e Aquirá). Este sistema apresenta uma baixa profundidade e uma pequena conexão ao oceano, formando algumas lagoas e planícies inundadas (CPRH, 2001). Além disso, a região costeira perto da boca do estuário exibe uma barreira de recife de arenito, sendo este responsável pelo aumento da taxa de sedimentação e por diminui o influxo de água (Silva 2009). Apesar de estar localizada dentro de uma área de proteção ambiental (APA de Guadalupe), este estuário vem sofrendo com os impactos causados por poluição decorrente de indústrias agrícolas (principalmente de cana de açúcar), fazendas de cultivo de camarão e atividade pesqueira (LIRA, 2010; LIRA et al., 2010a).

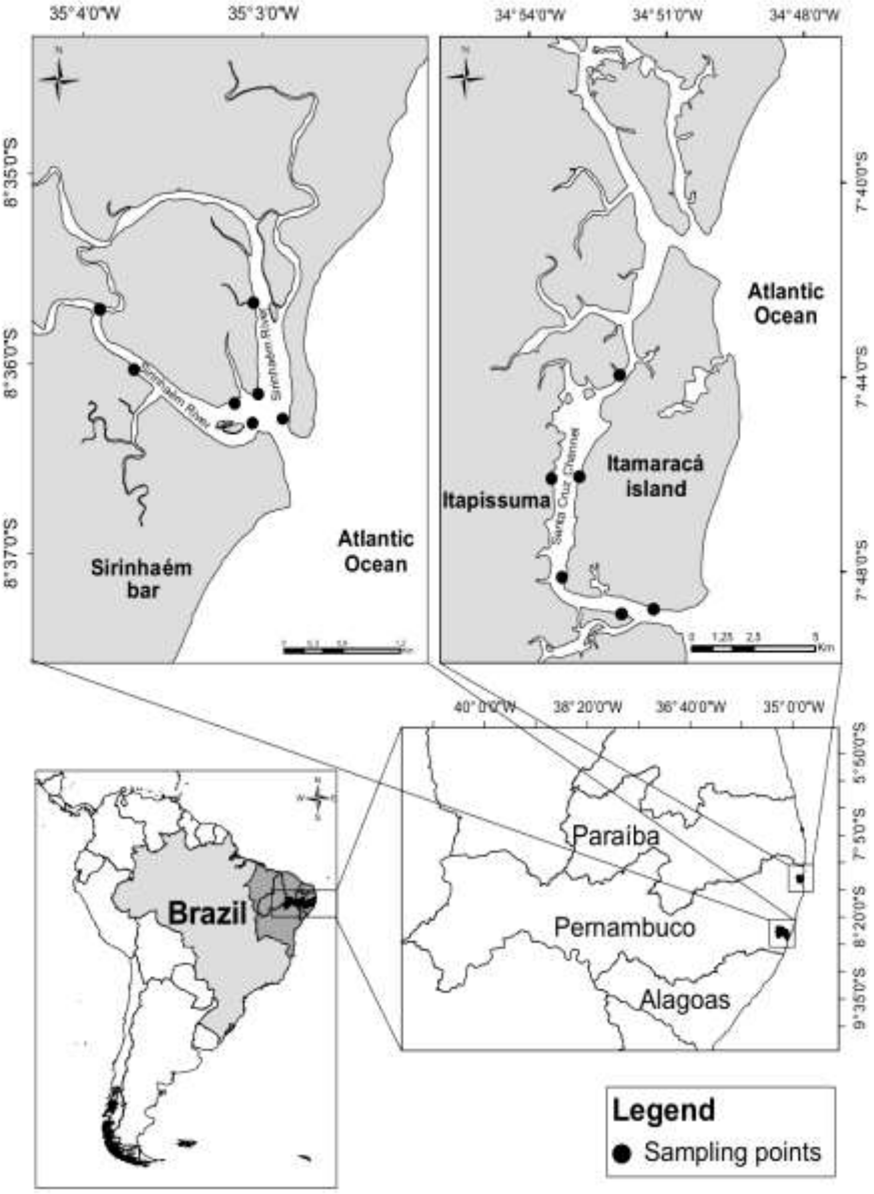

Figura 1 – Localização das regiões estuarinas do Canal de Santa Cruz (Itapissuma) e Barra de Sirinhaém no litoral norte e sul do estado de Pernambuco, Brasil. Fonte: (LIRA et al., 2017).

2.2 *Coleta dos dados*

As coletas foram realizadas durante o período de outubro de 2012 a junho de 2014, com uma frequência trimestral. O apetrecho utilizado foi a rede camboa, que possui aproximadamente 700 m de extensão, sendo um total de 6 redes, colocadas no mesmo momento, com duas panagens cada e malha variando entre 60 e 80 mm entre nós opostos. As despescas foram realizadas de acordo com a variação das marés (no período diurno ou noturno), com as redes sendo colocadas ao redor da margem dos manguezais durante a maré baixa, e sendo posteriormente levantadas na maré alta. Todos os indivíduos capturados foram medidos (comprimento total) e pesados (peso total e eviscerado). Em seguida, os peixes foram acondicionados em gelo e guardados para posterior análise e identificação em laboratório. A identificação das espécies foi realizada de acordo com literatura específica (e.g., (FIGUEIREDO e MENEZES, 1978, 1985, 2000; MENEZES e FIGUEIREDO, 1980)

2.3 *Espécies incluídas na análise*

A seleção das espécies para a análise de produtividade e susceptibilidade foi feita segundo os critérios propostos por Garcia e Vieira (2001), sendo as espécies classificadas como abundantes quando a sua captura percentual era maior que a razão 100/S, onde S é o número de espécies amostrada no ambiente; e frequentes quando a sua frequência de ocorrência (FO) era maior que 50% para a área em questão. Dessa forma, as espécies foram classificadas como: (1) abundantes e frequentes ($\%N > 100/S$ e $\%FO \geq 50\%$), (2) abundantes e pouco frequentes ($\%N > 100/S$ e $\%FO < 50\%$), (3) pouco abundantes e frequentes ($\%N > 100/S$ e $\%FO < 50\%$), e (4) pouco abundantes e pouco frequentes ($\%N < 100/S$ e $\%FO < 50\%$). Foram incluídas na PSA todas as categorias exceto aquelas classificadas como pouco abundantes e pouco frequentes.

2.4 *Análise dos dados*

A abordagem PSA baseia-se no pressuposto de que o risco para um componente ecológico/espécie depende de duas características: (1) a extensão do impacto devido à atividade pesqueira, a qual é determinada pela susceptibilidade dos organismos à pesca (Susceptibilidade – S) e (2) a capacidade de recuperação de um estoque pesqueiro após um estado de depleção ou danos causados pela pesca (Produtividade – P) (HOBDAY et al., 2007; ZHOU et al., 2016; LUCENA-FRÉDOU et al., 2017). De acordo com Stobutzki et al. (2002), uma espécie se torna vulnerável a uma determinada pescaria quando a mortalidade por pesca excede a sua capacidade biológica de se renovar.

Cada atributo relacionado à produtividade (P) e susceptibilidade (S) foi avaliado em uma escala dividida em três níveis: baixo (1), médio (2) e alto (3). Para o critério produtividade, o valor 1 indica que a espécie possui uma baixa produtividade e, consequentemente, um maior risco, enquanto o valor 3 está associado a uma alta produtividade e baixo risco. O oposto é observado para o atributo susceptibilidade, onde o nível 3 indica uma alta susceptibilidade e alto risco, e o 1 indica uma baixa susceptibilidade e baixo risco de captura. Para aqueles atributos faltantes não foi atribuída nenhuma pontuação e não foram usados para calcular os valores finais de P ou S. As informações sobre os critérios de produtividade e susceptibilidade foram obtidas através de literatura especializada e através do Fishbase (FROESE e PAULY, 2017).

Para avaliar a produtividade, foram analisados seis atributos:

(1) Tamanho de primeira maturação (L_{50}): espécies que apresentam uma maturação tardia são altamente vulneráveis a um possível risco de extinção (HUTCHINGS e REYNOLDS, 2004). Para aquelas que não possuíam o tamanho de primeira maturação disponível em literatura, o mesmo foi estimado através da seguinte fórmula: $L_{50} = 0,64 \times L\infty^{0,95}$, sendo $L\infty$ = comprimento assintótico (LE QUESNE e JENNINGS, 2012). Os valores referentes ao comprimento assintótico foram obtidos através do banco de dados do Fishbase (FROESE e PAULY, 2017);

(2) Tamanho máximo (L_{max}): o tamanho máximo de uma espécie é utilizado como um indicador relativo de sua capacidade de recuperação. As espécies que atingem grande porte tendem a viver mais, possuem crescimento lento e maturação tardia, portanto, suas populações recuperam-se mais vagarosamente do que aquelas que crescem menos

(ROBERTS e HAWKINS, 1999). O tamanho máximo considerado foi o maior obtido através da literatura ou do banco de dados utilizado neste estudo;

(3) Taxa de crescimento (k): velocidade com que uma determinada espécie alcança o seu tamanho máximo. Peixes com alto valor de k tendem a ter a vida mais curta do que aqueles que apresentam baixos valores (PATRICK et al., 2009). Quando não encontrado na literatura, a taxa de crescimento foi estimada de acordo com a fórmula empírica proposta por Le Quesne & Jennings, (2012): $k = 2{,}15 \times L\infty^{-0{,}46}$, sendo $L\infty$ = comprimento assintótico.

(4) Nível trófico: esse fator é inversamente proporcional à produtividade, isto é, quanto mais baixa for a produtividade, maior será o nível trófico de uma espécie (PATRICK et al., 2009). Os valores referente ao nível trófico foram obtidos através do banco de dados do Fishbase (FROESE e PAULY, 2017); e

(5) Taxa intrínseca de crescimento (r): esse parâmetro mede a taxa de crescimento de uma população sobre um intervalo de tempo curto, determinando se a população irá crescer exponencialmente, permanecer constante ou diminuir até a extinção. Para o cálculo desse parâmetro foi utilizada uma abordagem baseada no modelo proposto por Mertz (1970) considerando a tabela de vida, a qual geralmente tem uma aproximação de 10% do valor verdadeiro (STEARNS, 1992). A tabela de sobrevivência por idade foi calculada com base nas estimativas da mortalidade natural (M), conforme proposto por Gislason et al. (2010): $M = 0.55kL\infty^{1.44} * exp^{-1.61*logL}$, sendo L o comprimento médio da classe etária. Assumiu-se uma proporção sexual de 1: 1, e que a fertilidade das fêmeas é proporcional ao peso (αL^3). Os limites das pontuações de produtividade que definiram as três categorias ordenadas foram estabelecidos usando o método do tercil.

(6) L_{50}/L_{max}: essa proporção descreve o investimento relativo em crescimento somático e reprodutivo. Espécies de pequeno porte tendem a alcançar a maturidade sexual em tamanhos maiores quando relacionadas ao seu tamanho máximo, enquanto que espécies de

grande porte atingem a sua maturidade em tamanhos relativamente menores (JUAN-JORDÁ et al., 2013).

Os limites de todos os atributos de produtividade foram definidos pelo método do tercil (Tabela 1).

Tabela 1 – Atributos relacionados à produtividade, cada qual com seu respectivo peso, e os níveis utilizados para determinar a vulnerabilidade das espécies capturadas na pesca de camboa nos estuários de Barra de Sirinhaém e Canal de Santa Cruz, no estado de Pernambuco. Todos os valores que constam na tabela são provenientes dos resultados obtidos para as espécies analisadas.

Atributos	**Ranking**		
	Alta (1)	**Moderada (2)**	**Baixa (3)**
L_{50} (cm)	< 12,09	12,09 – 20,23	> 20,23
Lmax (cm)	< 31,66	31,66 – 59	> 59
k	> 0,41	0,30 – 0,41	< 0,30
Nível trófico	< 3,0	3,0 – 3,4	> 3,4
r	> 0,58	0,49 – 0,58	< 0,49
L_{50} /Lmax	< 0,29	0,29 – 0,43	> 0,43

Os critérios adotados para analisar a susceptibilidade das espécies à arte de pesca foram os seguintes (Tabela 2):

(1) Porcentagem de adultos (% > L_{50}): percentagem dos indivíduos capturados acima do tamanho de primeira maturação (STOBUTZKI et al., 2001a; LUCENA-FRÉDOU et al., 2017). Esse atributo foi calculado através da frequência de comprimento das espécies selecionadas considerando cada área (Canal de Santa Cruz e Barra de Sirinhaém) e o tamanho de primeira maturação obtido como atributo de produtividade descrito anteriormente. Os limites desse atributo foram definidos pelos tercis;

(2) Categoria comercial: as populações de peixes altamente valorizadas são mais susceptíveis à sobrepesca por pescadores comerciais devido ao aumento do esforço (PATRICK et al., 2009). Para este critério os dados foram obtidos através de pesquisa dos preços nos principais pontos de venda de pescado nas localidades amostradas. Os limites desse atributo foram definidos através do método do tercil;

(3) Grupos funcionais: zonas estuarinas possuem algumas funções bem definidas, como área de berçários, área de alimentação e rota de migração. As espécies foram classificadas conforme proposto por Elliott et al. (2007): marinhas migrantes (MM), estuarinas residentes (ER) e migrantes estuarinas (ME), com base em dados obtidos através da literatura. Por ser uma pescaria realizada em regiões estuarinas, as espécies classificadas como estuarinas residentes (ER) apresentam uma alta susceptibilidade (3), as migrantes estuarinas (ME) uma susceptibilidade média (2), enquanto que as marinhas migrantes (MM) se mostram menos susceptíveis (1);

(4) Frequência de ocorrência: quanto maior a frequência de captura de uma espécie, mais propensa a mesma está à sobrepesca. Os limites desse atributo foram definidos conforme a metodologia proposta por Garcia e Vieira (2001), na qual as espécies definidas como abundantes e frequentes foram classificadas com alta susceptibilidade (3), as abundantes e pouco frequentes com média (2) e as pouco abundantes e frequentes com uma baixa susceptibilidade (1);

(5) Razão Mortalidade/Taxa de Crescimento (Z/k): aqui assume-se que a cauda direita de uma distribuição de frequência de comprimento é determinada pelo $L\infty$, k e Z. A proporção entre a mortalidade total (Z) e a taxa de crescimento (k) foi estimada através do método de Powell-Wetherall (WETHERALL et al., 1987), na qual utilizou-se a distribuição da frequência de comprimento das espécies capturadas durante todo o período de amostragem. Espécies que apresentam um Z/k maior que 1 são consideradas mais susceptíveis (3)

(PAULY, 1984), entre 0,5 e 1 apresentam uma susceptibilidade média (2), e aquelas com proporção menor que 0,5 são consideradas a menos susceptíveis (1).

Tabela 2 - Atributos relacionados à susceptibilidade, cada qual com seu respectivo peso, e os níveis utilizados para determinar a vulnerabilidade das espécies capturadas na pesca de camboa nos estuários de Barra de Sirinhaém e Canal de Santa Cruz, no estado de Pernambuco. Todos os valores que constam na tabela são provenientes dos resultados obtidos para as espécies analisadas.

Atributos	**Ranking - Susceptibilidade**		
	Baixo (1)	**Moderado (2)**	**Alto (3)**
% > L_{50} (Adultos)	> 22,8	11,93 – 22,8	< 11,93
Categorial Comercial (R$)	< 6	6 - 18	> 18
Grupos funcionais	Marinhas Migrantes (MM)	Migrantes Estuarinas (ME)	Estuarinas Residentes (ER)
Frequência de Ocorrencia (FO)	Pouco abundantes e frequentes	Abundantes e pouco frequentes	Abundantes e frequente
Z/k	< 0,5	0,5 - 1	> 1

Os valores obtidos para cada conjunto de dados foram representados em um gráfico bidimensional, nos quais os índices de vulnerabilidade foram calculados de acordo com a distância Euclidiana da origem dos eixos da PSA, seguindo a seguinte fórmula: $v = \sqrt{(P - X_0)^2 + (S - Y_0)^2}$, sendo X_0 e Y_0 são as coordenadas de origem (x, y).

As pontuações foram representadas graficamente em um gráfico de dispersão, com P no eixo x e S no eixo y. As linhas de contorno separam as regiões de acordo com o seu nível de risco [baixo ($v \leq 1$), moderado ($1 < v \leq 2$) e alto ($v > 2$)] e agrupam aquelas espécies com níveis semelhantes (HOBDAY et al., 2007). O eixo x foi invertido (isto é, começa em 3 e termina em 1), de modo que a área próxima à origem (3, 1) corresponde a populações com alta produtividade e baixa susceptibilidade, portanto, estoques menos vulneráveis (baixo risco). Os estoques mais vulneráveis (alto risco) são os que se encontram mais distantes da origem (1, 3). A área intermediária do gráfico representa as espécies que estão sujeitas a um risco moderado.

Os pesos dos atributos de P e S foram atribuídos dentro de uma escala de 1, 2 e 3. Duas situações ocorreram:

(1) Foi considerado um caso de referência onde o peso 3 foi atribuído a três atributos de produtividade: L_{max}, k e r. Lucena-Frédou et al. (2017) mostraram que as diferenças entre os valores da produtividade são principalmente relacionadas ao L_{max} e o k. O fator r também recebeu um peso 3 pois é considerado um fator-chave para a resiliência, refletindo um conjunto de componentes referentes a história de vida (MUSICK et al., 2001). Os demais atributos de produtividade receberam o peso 2. No caso de referência da susceptibilidade, todos os atributos tiveram um peso 2. A produtividade total do estoque e os valores de susceptibilidade foram calculados através de uma média ponderada de todos os atributos.

(2) Considerando que a atribuição dos pesos é, de certa forma, subjetiva e potencialmente influente sobre o resultado final, uma análise de sensibilidade foi aplicada. Simulações com pesos dos atributos variando entre 1, 2 e 3 foram determinadas. Estimativas dos valores de produtividade e susceptibilidade foram calculadas com toda a gama de peso (1, 2 e 3). No final, cada espécie teve uma variação de valor de produtividade e de susceptibilidade representadas em um gráfico de contornos. Uma probabilidade de risco das espécies capturadas pela rede de camboa foi representada com um gráfico de barras de acordo com essas simulações.

Para testar se houveram diferenças significativas entre a produtividade, susceptibilidade e vulnerabilidade, levando-se em consideração o local e as famílias das espécies capturadas, foi utilizado o teste de Kruskal-Wallis. Todas as análises estatísticas foram feitas através do R Statistical Software v.3.4.0 (R CORE TEAM, 2017).

3 Resultados

3.1 *Composição de captura*

Foram capturados um total de 5375 indivíduos, sendo 4420 capturados no Canal de Santa Cruz, distribuídos entre 47 espécies e 22 famílias; e 955 capturados na Barra de Sirinhaém, distribuídos entre 38 espécies pertencentes a 19 famílias (Tabela 3). No Canal de Santa Cruz, as famílias que apresentaram maior número de indivíduos capturados foram Mugilidae (45,2%), Gerreidae (24,7%) e Engraulidae (17,6%). *Mugil curema* foi a espécies mais abundante em número, responsável por 44,5% do total de indivíduos capturados, seguida de *Cetengraulis edentulus* (13,7%) e *Eucinostomus argenteus* (12,1%). A família Gerreidae apresentou maior riqueza em número de espécies (6 espécies), juntamente com Carangidae (4 espécies).

Para a Barra de Sirinhaém, foram mais abundantes em número aquelas espécies pertencentes às famílias Gerreidae (26,7%), Tetraodontidae (21,4%), Centropomidae (19,5%) e Sciaenidae (11,5%). *Sphoeroides testudineus* foi a espécie com maior captura em número, sendo responsável por 21,4% do total amostrado, seguida de *Diapterus auratus* (10,7%) e *Diapterus rhombeus* (9,5%). Assim como no Canal de Santa Cruz, Gerreidae também apresentou uma alta riqueza de espécies na Barra de Sirinhaém, com 5 espécies capturadas, ficando atrás apenas da família Sciaenidae, que registrou um total de 6 espécies amostradas.

Para a análise de vulnerabilidade da captura pela rede de camboa, foram selecionadas as espécies classificadas como abundantes e frequentes, abundantes e pouco frequentes e pouco abundantes e frequentes. Para o Canal de Santa Cruz, as seguintes espécies foram consideradas: *Anchovia clupeoides*, *Bairdiella ronchus*, *Centropomus parallelus*, *Centropomus undecimalis*, *Cetengraulis edentulus*, *Diapterus auratus*, *Eucinostomus* *argenteus*, *Eucinostomus gula*, *Harengula clupeola*, *Lutjanus alexandrei*, *Mugil curema* e *Sphoeroides testudineus*. Para a Barra de Sirinhaém, as espécies consideradas foram: *B. ronchus*, *C. parallelus*, *Centropomus pectinatus*, *C. undecimalis*, *Diapterus auratus*, *D. rhombeus*, *E. argenteus*, *E. gula*, *M. curema*, *Sciades herzbergii* e *S. testudineus* (Tabela 3).

3.2 *Parâmetros de história de vida*

Os parâmetros de história de vida apresentaram uma ampla faixa de variação (Tabela 4). Dentre os parâmetros observados, poucos foram encontrados em trabalhos realizados na região nordeste (Couto e Guedes, 1981; Mendonça, 2004; Aschenbrenner e Ferreira, 2015; Leão, 2016). Dessa forma, a maioria dos dados tiveram que ser estimados para várias espécies, com destaque para a taxa intrínseca de crescimento (r), que foi estimada para todas as espécies analisadas. O maior tamanho de primeira maturação foi registrado para *Centropomus undecimalis* (45,5 cm), enquanto *Eucinostomus argenteus* apresentou o menor (8,3 cm). Já o tamanho máximo encontrado em literatura variou de 18 cm (*Harengula clupeola*) a 140 cm (*Centropomus undecimalis*). Em relação à taxa de crescimento (k), *C. unidecimalis* apresentou um menor valor (crescimento mais lento), com um k igual a 0,0712 cm/ano, e *C. edentulus* o mais elevado, com 1,05 cm/ano.

Em relação a captura de indivíduos juvenis (Tabela 3), as espécies que apresentaram uma maior captura de imaturos no Canal de Santa Cruz foram *C. parallelus* (97,6%), *M. curema* (96,5%), *C. undecimalis* (93,8%), *D. auratus* (93,8%), *E. argenteus* (87,3%), também se destacando *L. alexandrei* com 83,1% dos indivíduos capturados abaixo do seu tamanho de primeira maturação sexual. As espécies *H. clupeola* e *B. ronchus* foram as que tiveram menor incidência de juvenis, com 0% e 40,7%. No estuário de Sirinhaém as espécies *C. pectinatus* (95,8%), *C. undecimalis* (92,2%), *S. herzbergii* (88,4%), *D. rhombeus* (78,1%) e *C. parallelus* (76,6%) se destacaram como aquelas com maior captura de juvenis. *B. ronchus* (14,8%), *E. gula* (46,4%) e *S. testudineus* (49,5%) foram as com menor índice de juvenis capturados.

3.3 *Estimativa de vulnerabilidade*

A produtividade das espécies capturadas na pesca com rede de camboa nos estuários de Barra de Sirinharém e no Canal de Santa Cruz variou entre 1,13 e 2,73 (Tabela 5). *Eucinostomus argenteus* e *Anchovia clupeoides* apresentaram os maiores valores de produtividade, ambos com 2,73, enquanto *Centropomus parallelus* e *C. undecimalis* apresentaram os menores, 1,13 e 1,27, respectivamente.

Para os dois estuários em conjunto, os valores obtidos para o critério susceptibilidade variaram entre 1,80 e 2,40: *Sciades herzbergii* (Barra de Sirinhaém) e *Sphoeroides testudineus* (Canal de Santa Cruz) obtiveram o menor e o maior valor, respectivamente. *Eucinostomus gula*, *M. curema* e *H. clupeola* apresentaram menor susceptibilidade ao apetrecho no Canal de Santa Cruz; enquanto que *B. ronchus*, *C. edentulus* e *S. testudineus* se mostraram mais susceptíveis. Para a Barra de Sirinhaém, as espécies *S. herzbergii*, *C. undecimalis* e *C. pectinatus* foram as menos susceptíveis, e *B. ronchus, M. curema* e *D. auratus* se mostraram mais susceptíveis ao apetrecho (Tabela 5).

Quando comparados entres as duas áreas, os critérios de susceptibilidade (H=0,0010, p>0,05) e produtividade (H=1,0003, p>0,05) não apresentaram diferença significativa entre si. Quando comparados entre as famílias capturadas, os critérios de susceptibilidade (H=5,18, p<0,05,) e produtividade (H=9,06, p<0,05) se mostraram diferentes, e as espécies pertencentes à família Gerreidae obtiveram os menores escores para susceptibilidade, quando comparado com aquelas da família Centropomidae. O oposto foi observado para o critério produtividade, com os escores da família Centropomidae se mostrando inferiores aos de Gerreidae.

As espécies que apresentaram maior vulnerabilidade à captura pela rede de camboa foram: *Centropomus paralellus*, *Centropomus unidecimalis* para ambas as áreas e *Lutjanus alexandrei* para o Canal de Santa Cruz. As espécies *Sphoeroides testudineus* para Barra de

Sirinhaém, *Eucinostomus argenteus* (para ambas as áreas), *Anchovia clupeoides* e *Harengula clupeola* foram as menos vulneráveis (Figura 2). A vulnerabilidade não apresentou diferença significativa quando comparada entre as áreas (H=0,75, p>0,05). Entretanto, assim como para os critérios de susceptibilidade e produtividade, a mesma apresentou diferença entre as famílias (H=8,25, p<0,05), com as famílias Gerreidae e Centropomidae (p<0,05) sendo responsáveis por essa diferença, sendo a Centropomidae a mais vulnerável.

A simulação das possíveis combinações de peso dos atributos de produtividade e susceptibilidade mostrou que a maior parte das espécies está classificada como risco moderado, devendo se salientar que, mesmo com a realização destas simulações, a maioria das espécies analisadas não sofreram alterações na sua classificação de risco (Figura 3). Considerando o efeito da pescaria de camboa na vulnerabilidade das espécies, as espécies da família Centropomidae (*C. undecimalis* e *C. parallelus*) mostraram 60 e 100% de probabilidade de serem classificadas como espécies de alto risco, respectivamente, seguida de *L. alexandrei* com 75%. O oposto pode ser observado para a espécie *Harengula clupeola* a qual obteve uma probabilidade de 78% de ser de baixo risco.

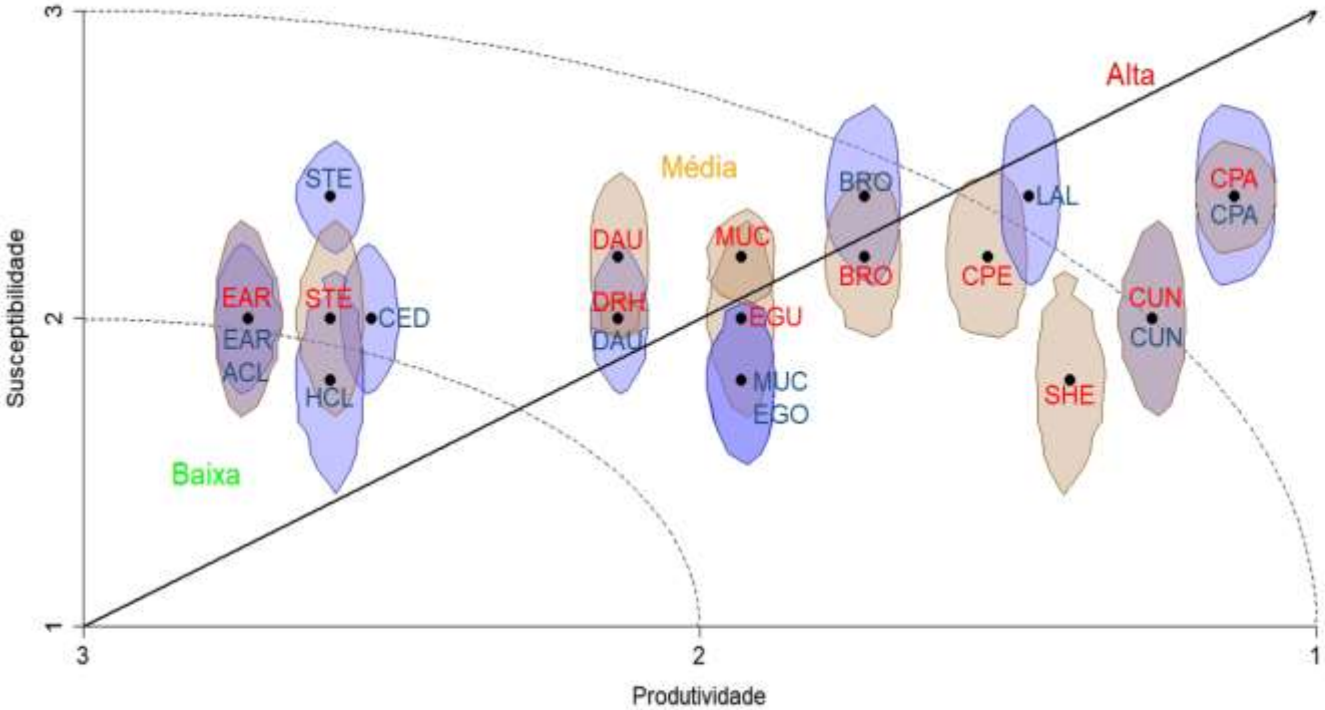

Figura 2 – Produtividade e susceptibilidade das principais espécies capturadas pela pescaria de camboa nos estuários de Barra de Sirinhaém e no Canal de Santa Cruz, PE, classificadas em baixo, médio e alto risco. As áreas em azul correspondem ao estuário do Canal de Santa Cruz, enquanto as vermelhas são referentes a Barra de Sirinhaém. Legenda: ACL: *Anchovia clupeoides*, BRO: *Bairdiella ronchus*, CPA: *Centropomus parallelus*, CPE: *Centropomus pectinatus*, CUN: *Centropomus undecimalis*, CED: *Cetengraulis edentulus*, DAU: *Diapterus auratus*, DRH: *Diapterus rhombeus*, EAR: *Eucinostomus argenteus*¸ EGU: *Eucinostomus gula*, HCL: *Harengula clupeola*¸ LAL: *Lutjanus alexandrei*¸ MCU: *Mugil curema*, SHE: *Sciades herzbergii*, STE: *Sphoeroides testudineus*.

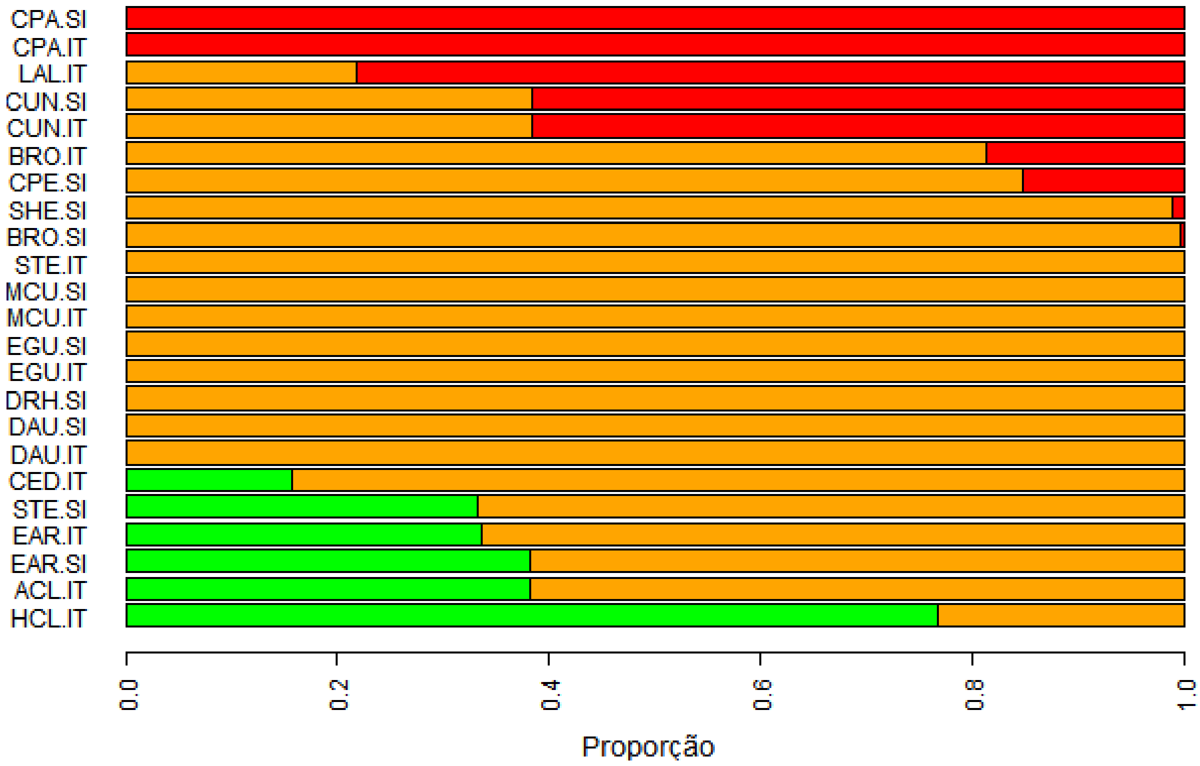

Figura 3 – Probabilidade de risco das principais espécies capturadas pela rede de camboa nos estuários de Barra de Sirinhaém e Canal de Santa Cruz, PE. As espécies foram ordenadas (de cima para baixo) de acordo com o rank de vulnerabilidade. Vermelho – alto risco; laranja – médio risco; verde – baixo risco. Legendas: IT: Canal de Santa Cruz, SI: Barra de Sirinhaém, ACL: *Anchovia clupeoides*, BRO: *Bairdiella ronchus*, CPA: *Centropomus parallelus*, CPE: *Centropomus pectinatus*, CUN: *Centropomus undecimalis*, CED: *Cetengraulis edentulus*, DAU: *Diapterus auratus*, DRH: *Diapterus rhombeus*, EAR: *Eucinostomus argenteus*¸ EGU: *Eucinostomus gula*, HCL: *Harengula clupeola*¸ LAL: *Lutjanus alexandrei*¸ MCU: *Mugil curema*, SHE: *Sciades herzbergii*, STE: *Sphoeroides testudineus*.

Tabela 3 – Espécies capturadas pela pescaria de camboa nos estuários do Canal de Santa Cruz e Barra de Sirinhaém, no estado de Pernambuco. Legendas: n (%) – abundância em porcentagem; FO (%) – frequência de ocorrência em porcentagem; % < L50 – porcentagem de indivíduos capturados abaixo do tamanho de primeira maturação sexual. As espécies marcadas com * foram as utilizadas na PSA.

Família/Espécie	**Estuário**							
	Itapissuma				**Sirinhaém**			
	n (%)	**FO (%)**	**Classificação**	**% < L50**	**n (%)**	**FO (%)**	**Classificação**	**% < L50**
Achiridae								
Achirus lineatus	**0,11**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**0,42**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Ariidae								
Cathorops agassizii	**0,05**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Cathorops spixii	**-**	**-**	**-**	**-**	**0,31**	**25**	**Pouco abundantes e não frequentes**	**-**
*Sciades herzbergii**	**0,29**	**37,5**	**Pouco abundantes e não frequentes**	**-**	**2,72**	**62,5**	**Abundantes e frequentes**	**88,4**
Atherinopsidae								
Membras dissimilis	**0,02**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Carangidae								
Caranx hippos	**0,27**	**25**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Caranx latus	**0,66**	**25**	**Pouco abundantes e não frequentes**	**-**	**1,26**	**37,5**	**Pouco abundantes e não frequentes**	**-**
Chloroscombrus chrysurus	**-**	**-**	**-**	**-**	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Oligoplites palometa	**-**	**-**	**-**	**-**	**0,21**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Oligoplites saurus	**0,05**	**25**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Selene vomer	**0,07**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Centropomidae								
Centropomus ensiferus	**-**	**-**	**-**	**-**	**2,26**	**37,5**	**Pouco abundantes e não frequentes**	**-**
*Centropomus parallelus**	**1,92**	**62,5**	**Pouco abundantes e frequentes**	**97,6**	**6,28**	**75**	**Abundantes e frequentes**	**76,6**
*Centropomus pectinatus**	**0,11**	**25**	**Pouco abundantes e não frequentes**	**-**	**2,51**	**75**	**Pouco abundantes e frequentes**	**95,8**

*Centropomus undecimalis**	**1,11**	**62,5**	**Pouco abundantes e frequentes**	**93,8**	**8,06**	**87,5**	**Abundantes e frequentes**	**92,2**
Clupeidae								
*Harengula clupeola**	**1,02**	**25**	**Pouco abundantes e frequentes**	**0**	-	-	-	-
Opisthonema oglinum	**0,02**	**12,5**	**Pouco abundantes e não frequentes**	-	-	-	-	-
Sardinella brasiliensis	**0,09**	**12,5**	**Pouco abundantes e não frequentes**	-	-	-	-	-
Eleotridae								
Guavina guavina	-	-	-	-	**0,42**	**25**	**Pouco abundantes e não frequentes**	-
Elopidae								
Elops saurus	**0,05**	**12,5**	**Pouco abundantes e não frequentes**	-	-	-	-	-
Engraulidae								
*Anchovia clupeoides**	**3,89**	**12,5**	**Abundantes e pouco frequentes**	**75,0**	-	-	-	-
*Cetengraulis edentulus**	**13,7**	**75**	**Abundantes e frequentes**	**84,1**	-	-	-	-
Ephippidae								
Chaetodipterus faber	**0,43**	**25**	**Pouco abundantes e não frequentes**	-	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	-
Gerreidae								
*Diapterus auratus**	**6,4**	**75**	**Abundantes e frequentes**	**93,2**	**10,7**	**62,5**	**Abundantes e frequentes**	**66,6**
*Diapterus rhombeus**	**0,25**	**12,5**	**Pouco abundantes e não frequentes**	-	**9,6**	**75**	**Abundantes e frequentes**	**78,1**
Eucinostomus argenteus *	**12,1**	**75**	**Abundantes e frequentes**	**87,29**	**2,51**	**50**	**Pouco abundantes e frequentes**	**66,7**
*Eucinostomus gula**	**5,27**	**62,5**	**Abundantes e frequentes**	**81,1**	**2,93**	**50**	**Abundantes e frequentes**	**46,43**
Eucinostomus havana	**0,38**	**12,5**	**Pouco abundantes e não frequentes**	-	-	-	-	-
Eucinostomus melanopterus	**0,32**	**12,5**	**Pouco abundantes e não frequentes**	-	**1,05**	**12,5**	**Pouco abundantes e não frequentes**	-
Haemulidae								
Genyatremus luteus	**0,07**	**12,5**	**Pouco abundantes e não frequentes**	-	**0,21**	**12,5**	**Pouco abundantes e não frequentes**	-
Haemulon parra	**0,07**	**12,5**	**Pouco abundantes e não frequentes**	-	-	-	-	
Pomadasys corvinaeformis	-	-	-	-	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	-
Pomadasys crocro	**0,05**	**25**	**Pouco abundantes e não frequentes**	-	**0,63**	**37,5**	**Pouco abundantes e não frequentes**	-
Hemiramphidae								
Hemiramphus brasiliensis	**0,02**	**12,5**	**Pouco abundantes e não frequentes**	-	-	-	-	-
Lobotidae								
Lobotes surinamensis	**0,02**	**12,5**	**Pouco abundantes e não frequentes**	-	-	-	-	-

Lutjanidae								
*Lutjanus alexandrei**	**3,08**	**62,5**	**Abundantes e frequentes**	**83,1**	**0,31**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Lutjanus analis	**0,05**	**25**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Lutjanus jocu	**0,25**	**25**	**Pouco abundantes e não frequentes**	**-**	**3,98**	**25**	**Pouco abundantes e não frequentes**	**-**
Megalopidae								
Megalops atlanticus	**-**	**-**	**-**	**-**	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Mugilidae								
*Mugil curema**	**44,5**	**62,5**	**Abundantes e frequentes**	**96,5**	**8,0**	**75**	**Abundantes e frequentes**	**78,9**
Mugil liza	**0,66**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**0,21**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Muraenidae								
Gymnothorax funebris	**-**	**-**	**-**	**-**	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Paralichthyidae								
Citharichthys spilopterus	**0,14**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**0,21**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Paralichthys brasiliensis	**0,02**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	
Polynemidae								
Polydactylus virginicus	**0,07**	**25**	**Pouco abundantes e não frequentes**	**-**	**0,42**	**37,5**	**Pouco abundantes e não frequentes**	**-**
Sciaenidae								
*Bairdiella ronchus**	**1,22**	**62,5**	**Pouco abundantes e frequentes**	**40,7**	**8,48**	**62,5**	**Abundantes e frequentes**	**14,81**
Cynoscion acoupa	**-**	**-**	**-**	**-**	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Isopisthus parvipinnis	**-**	**-**	**-**	**-**	**1,15**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Micropogonias furnieri	**-**	**-**	**-**	**-**	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Ophioscion punctatissimus	**-**	**-**	**-**	**-**	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Stellifer microps	**0,02**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Stellifer stellifer	**0,05**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Serranidae								
Epinephelus marginatus	**0,02**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Sparidae								
Archosargus rhomboidalis	**0,32**	**25**	**Pouco abundantes e não frequentes**	**-**	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	**-**
Sphyraenidae								
Sphyraena barracuda	**0,11**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**0,1**	**12,5**	**Pouco abundantes e não frequentes**	**-**

Sphyraena guachancho	**0,07**	**25**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Tetraodontidae								
Colomesus psittacus	**0,11**	**25**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
Sphoeroides greeleyi	**0,05**	**12,5**	**Pouco abundantes e não frequentes**	**-**	**-**	**-**	**-**	**-**
*Sphoeroides testudineus**	**0,52**	**75**	**Pouco abundantes e frequentes**	**86,9**	**21,4**	**37,5**	**Abundantes e pouco frequentes**	**49,5**
Trichiuridae								
Trichiurus lepturus	**-**	**-**	**-**	**-**	**0,94**	**25**	**Pouco abundantes e não frequentes**	**-**

Tabela 4 – Parâmetros de história de vida das espécies capturadas nos estuários de Barra de Sirinhaém e Canal de Santa Cruz.

Espécies	**L_{50}** (cm)	**Fonte**	**L_{max} (cm)**	**Fonte**	**k (cm.ano^{-1})**	**Fonte**	**Nível Trófico**	**Fonte**	**r**	**Fonte**
Anchovia clupeoides	11,7	**1**	30	**6**	0,575	⸸	3,4	**23**	0,6128	⸸
Bairdiella ronchus	15,8	**2**	35	**7**	0,410	⸸	3,5	**24**	0,5852	⸸
Centropomus parallelus	23,54	*	72	**8**	0,187	⸸	4,2	**25**	0,2723	⸸
Centropomus pectinatus	30,35	⸸	56	**9**	0,332	⸸	4	**26**	0,5399	⸸
Centropomus undecimalis	45,5	**3**	140	**10**	0,071	**17**	4,2	**23**	0,0969	⸸
Cetengraulis edentulus	12,88	*	18.2	**11**	1,05	**18**	2,1	**27**	1,0847	⸸
Diapterus auratus	18,7	*	42.8	*	0,375	⸸	2,4	**23**	0,5787	⸸
Diapterus rhombeus	14,98	*	42.3	*	0,377	⸸	3	**28**	0,5799	⸸
Eucinostomus argenteus	8,32	**4**	21	**1**	0,61	**19**	3,2	**29**	0,9130	⸸
Eucinostomus gula	11	**5**	25,5	**12**	0,29	**20**	2,7	**30**	0,2215	⸸
Harengula clupeola	8,86	⸸	18	**13**	0,69	**21**	3,3	**23**	0,7662	⸸
Lutjanus alexandrei	17,23	*	24,3	**14**	0,24	**22**	3,8	**31**	0,2132	⸸
Mugil curema	22,7	*	90	**15**	0,371	⸸	2	**28**	0,5664	⸸
Sciades herzbergii	27,14	*	94,2	**16**	0,262	⸸	3,3	**32**	0,4794	⸸
Sphoeroides testudineus	11,4	*	59	*	0,77	⸸	3,4	**29**	1,2562	⸸

* Banco de dados do autor; ⸸ - Estimado; **1** - (BOUCHON-NAVARRO et al., 2006); **2** - (CASTRO et al., 1999); **3** – (COUTO e GUEDES, 1981); **4** – (LEÃO, 2016); **5** - (MEXICANO-CÍNTORA, 1999); **6** - (BIGELOW et al., 1963); **7** - (CHAO, 1978); **8** - (IGFA, 2001); **9** - (CERVIGÓN et al., 1992); **10** - (LIESKE e MYERS, 1994); **11** – (GIARRIZZO et al., 2006); **12** - (AMADOR-DEL ANGEL et al., 2015); **13** – (LIESKE e MYERS, 1994); **14** - (MOURA e LINDEMAN, 2007); **15** - (HARRISON, 1995); **16** - (CHACON et al., 1994); **17** – (MENDONÇA, 2004); **18** - (SOUZA-CONCEIÇÃO e SCHWINGEL, 2011); **19** - (SILVA et al., 2014); **20** - (GARCIA e DUARTE, 2006); **21** - (POSADA, 1999); **22** – (ASCHENBRENNER e FERREIRA, 2015); **23** - (SIERRA et al., 1994); **24** - (GARCIA e NIETO, 1978); **25** - (BOUJARD et al., 1997); **26** – (FROESY e PAULY, 2017); **27** - (WHITEHEAD et al., 1988); **28** - (GÓMEZ-CANCHONG et al., 2004); **29** - (AUSTIN e AUSTIN, 1971); **30** - (MOTTA et al., 1995); **31** - (RANDALL, 1967); **32** - (LE BAIL et al., 2000).

Tabela 5 - Índices de produtividade, susceptibilidade e vulnerabilidade com os níveis classificação das espécies capturadas pela pescaria de camboa nos estuários de Barra de Sirinhaém e do Canal de Santa Cruz, no estado de Pernambuco.

Espécie	**Família**	**Estuário**	**Rank**	**Produtividade**	**Susceptibilidade**	**Vulnerabilidade**
Centropomus parallelus	Centropomidae	Canal de Santa Cruz	1	1,13	2,40	2,33
Centropomus parallelus	Centropomidae	Barra de Sirinhaém	2	1,13	2,20	2,22
Lutjanus alexandrei	Lutjanidae	Canal de Santa Cruz	3	1,47	2,40	2,08
Centropomus undecimalis	Centropomidae	Canal de Santa Cruz	4	1,27	2,00	2,00
Centropomus undecimalis	Centropomidae	Barra de Sirinhaém	5	1,27	2,00	2,00
Bairdiella ronchus	Sciaenidae	Canal de Santa Cruz	6	1,73	2,40	1,89
Sciades herzbergii	Ariidae	Barra de Sirinhaém	7	1,40	1,80	1,79
Centropomus pectinatus	Centropomidae	Barra de Sirinhaém	8	1,53	2,00	1,78
Bairdiella ronchus	Sciaenidae	Barra de Sirinhaém	9	1,73	2,20	1,74
Mugil curema	Mugilidae	Barra de Sirinhaém	10	1,93	2,20	1,61
Diapterus auratus	Gerreidae	Barra de Sirinhaém	11	2,13	2,20	1,48
Cetengraulis edentulus	Engraulidae	Canal de Santa Cruz	12	2,53	2,40	1,48
Eucinostomus gula	Gerreidae	Barra de Sirinhaém	13	1,93	2,00	1,46
Sphoeroides testudineus	Tetraodontidae	Canal de Santa Cruz	14	2,60	2,40	1,46
Eucinostomus gula	Gerreidae	Canal de Santa Cruz	15	1,93	1,80	1,33
Mugil curema	Mugilidae	Canal de Santa Cruz	16	1,93	1,80	1,33
Diapterus auratus	Gerreidae	Canal de Santa Cruz	17	2,13	2,00	1,32
Diapterus rhombeus	Gerreidae	Barra de Sirinhaém	18	2,13	2,00	1,32
Sphoeroides testudineus	Tetraodontidae	Barra de Sirinhaém	19	2,60	2,00	1,08
Anchovia clupeoides	Engraulidae	Canal de Santa Cruz	20	2,73	2,00	1,03
Eucinostomus argenteus	Gerreidae	Canal de Santa Cruz	21	2,73	2,00	1,03
Eucinostomus argenteus	Gerreidae	Barra de Sirinhaém	22	2,73	2,00	1,03
Harengula clupeola	Clupeidae	Canal de Santa Cruz	23	2,60	1,80	0,89

4. Discussão

Analisar a sustentabilidade de pescarias com caráter multiespecífico e com poucas informações disponíveis vem sendo um grande desafio para os pesquisadores (ZHOU e GRIFFITHS, 2008), uma vez que a natureza indiscriminada dessas pescarias tem um grande potencial para afetar a população das espécies capturadas como fauna acompanhante (STOBUTZKI et al., 2001a), na maioria das vezes carente de dados de captura e da história de vida. As pescarias realizadas em ambientes tropicais apresentam este problema, tendo vários trabalhos relatado a captura de uma grande diversidade de organismos, que incluem desde invertebrados móveis ou sésseis até elasmobrânquios, tartarugas e uma grande diversidade de peixes (MILTON, 2001; STOBUTZKI et al., 2001b, 2002; FEITOSA et al., 2008; SILVA JÚNIOR et al., 2013). O método de Análise de Produtividade e Susceptibilidade (PSA) pode ser facilmente utilizado em pescarias com este caráter, nas quais dados de capturas para aplicação de modelos tradicionais de avaliação são deficientes ou até mesmo inexistentes, sendo os parâmetros de história de vida e dados ecológicos utilizados como entrada de dados (PATRICK et al., 2010; HOBDAY et al., 2011; LUCENA-FRÉDOU et al., 2016; ZHOU et al., 2016). Dessa forma, este tipo de abordagem pode auxiliar na tomada de decisões referentes a estratégias para planos de gerenciamento e manejo pesqueiro (ZHOU et al., 2009b).

As espécies *Centropomus parallelus* e *C. undecimalis* foram consideradas muito vulneráveis nos estuários do Canal de Santa Cruz e Barra de Sirinhaém. Isso se deve à elevada captura de indivíduos juvenis; 97,3% para *C. parallelus* no Canal de Santa Cruz, e 93,8% e 92,2% para *C. undecimalis*, no Canal de Santa Cruz e em Barra de Sirinhaém, respectivamente. Por servirem de berçário para muitas espécies (Elliott et al., 2007), as pescarias realizadas em zonas estuarinas tendem a capturar uma maior quantidade de indivíduos com tamanho abaixo do comprimento de primeira maturação sexual. Muitos indivíduos jovens destas espécies ocorrem nas áreas estudadas, principalmente no Canal de Santa Cruz, o qual é caracterizado por possuir um ambiente amplo e com uma grande diversidade de habitats (SILVA-JÚNIOR et al., 2016). Estruturas de capturas semelhantes foram observadas por Giglio e Freitas (2013), em uma pescaria com rede de camboa realizada no estado da Bahia, também na região nordeste do Brasil, nas quais uma grande parte dos indivíduos capturados ainda não tinha atingido a maturação sexual. Segundo os autores, o alto índice de indivíduos imaturos também se deve ao tamanho reduzido da malha da camboa (40 mm), o que acarreta uma baixa seletividade. Adicionalmente, as espécies de Centropomidae possuem um alto valor comercial, que faz com que a incidência da pesca sobre elas seja ainda maior. De acordo com Lira et al. (2017), as espécies da família são classificadas como peixes de grande importância ecológica, considerados como espécies-chave na teia trófica do ecossistema, principalmente para o estuário de Barra de Sirinhaém, aumentando assim a necessidade de um maior acompanhamento destas pescarias, para a região onde esta pesca não é regulamentada (LIRA et al., 2017).

O mesmo foi observado em relação à família Lutjanidae, para a qual se destacou a espécie *L. alexandrei*, visto que 83,1% dos indivíduos capturados se encontravam abaixo do tamanho de primeira maturação sexual. Indivíduos dessa família costumam ser encontrados entre raízes de mangue quando juvenis, fazendo uso dessas áreas como zonas de berçário (MOURA e LINDEMAN, 2007). Este fator, somado à alta incidência de juvenis, mostra que *L. alexandrei* é uma espécie que depende dessas regiões, tornando-a mais susceptível à captura. Além disso, fatores como alta exploração comercial, devido ao seu alto valor de venda na região, crescimento lento (k-estrategista), e alto valor de Z/k estimado (proxy da mortalidade por pesca), também contribuíram para que esta espécie fosse classificada como de alta vulnerabilidade à pesca, apresentando uma alta susceptibilidade e baixa produtividade. Assim como os centropomídeos, os lutjanídeos não apresentam qualquer medida de gerenciamento pesqueiro para a região.

Dentre as espécies analisadas, *A. clupeoides*, *E. argenteus* e *H. clupeola*, de rápido crescimento e maturação precoce, foram consideradas como sendo as menos vulneráveis à pesca com a rede de camboa. Espécies das famílias Engraulidae, Gerreidae e Clupeidae são diretamente associadas a áreas ricas em nutrientes, que proporcionam uma grande variedade de alimentos, características que podem ser encontradas no Canal de Santa Cruz, uma vez que este é o maior e mais produtivo estuário da costa pernambucana (LIRA et al., 2010b). A predominância de manguezais e substrato lamoso no Canal de Santa Cruz (SILVA et al., 2011) possibilita a formação de uma grande variedade de micro-habitat, fundamentais para o crescimento e desenvolvimento durante o ciclo de vida dessas espécies (GOODALL, 1986). Sendo assim, estas características podem produzir uma maior diversidade de habitats e assembleias de peixes mais ricas em espécies morfológica e funcionalmente distintas, além de ecologicamente diversas (SILVA-JÚNIOR et al., 2016), o que pode ser visto na grande variedade de espécies capturadas na presente pesquisa.

As análises de produtividade e susceptibilidade podem ser frágeis, dado o caráter semiquantitativo da análise. Por ser uma abordagem mais conservadora, algumas espécies podem receber uma alta pontuação na vulnerabilidade devido à falta de informações (FUJITA et al., 2014), sendo classificadas como de alto risco, não chegando a expressar realmente a verdadeira vulnerabilidade das mesmas, devido à escassez de dados sobre seus atributos de produtividade e susceptibilidade (OSIO et al., 2015; LUCENA-FRÉDOU et al., 2017). Estas seriam as chamadas “falso positivas”. Por outro lado, a incorporação de incertezas é útil, uma vez que a abordagem preventiva da gestão pesqueira leva em consideração essas incertezas (LUCENA-FRÉDOU et al., 2017). Ao longo dos últimos anos, as incertezas vêm sendo incorporadas de várias maneiras nas avaliações de risco ecológico (HARWOOD e STOKES, 2003; SETHI et al., 2005; ASCOUGH et al., 2008; DARBRA et al., 2008; HOBDAY et al., 2011; ZHOU et al., 2011, 2013; MCCULLY et al., 2013; YU et al., 2014). Na presente pesquisa, essas incertezas foram incorporadas através das simulações, as quais mostram ser bastante eficientes, uma vez que nos fornecem informações que permitem priorizar o monitoramento, avaliação e manejo dos estoques e espécies que estão em um alto risco causado pela atividade pesqueira.

A setorização e regulamentação das pescarias nas áreas onde são praticadas, possibilitam indicar quais espécies devem receber maiores restrições quando se trata de atos normativos para o gerenciamento das mesmas (ZHOU et al., 2009a). Pouco se conhece sobre os impactos das pescarias artesanais realizadas em áreas estuarinas, assim como sobre as medidas de regulamentação de esforço pesqueiro. O método aqui apresentado é o primeiro a avaliar a vulnerabilidade das espécies capturadas por uma pescaria realizada em ambiente estuarino de zonas tropicais. Além de se mostrar bastante apropriada, é uma ferramenta prática para o manejo pesqueiro e a gestão das pescarias de caráter multiespecífico, para as quais há uma grande limitação de dados. Dessa forma, esta pode ser empregada para determinar quais espécies estão passíveis a sofrerem um maior risco de extinção, sejam elas fauna acompanhante ou espécies alvo das pescarias. Apesar de apresentar algumas limitação, o método é bastante flexível, podendo ter a sua complexidade aumentada em função da disponibilidade de dados, e dessa forma, se encaixar de acordo com as necessidades de cada demanda (HOBDAY et al., 2011).

Atualmente, a ciência pesqueira no Brasil vem enfrentando vários problemas no que se refere à disponibilidade de dados estatísticos das pescarias. Essa falta de dados pode ser vista como uma grande ameaça aos diversos recursos pesqueiros de importância comercial, dos quais milhões de pessoas dependem para sua alimentação e renda, tudo isso diante da falta de gestão. Recentemente, a OCEANA Brasil emitiu uma carta de recomendação endossada por vários pesquisadores pesqueiros do Brasil (OCEANA, 2017), mostrando que a falta de dados pesqueiros é altamente prejudicial para a atividade, uma vez que os estoques pesqueiros estão sendo explotados sem qualquer informação sobre os mesmos. O último boletim de pesca no Brasil foi publicado no ano de 2011 (MMA, 2011), entretanto sabidamente com uma série de deficiências. Esta carência de dados é um problema bem mais abrangente e comum em vários países tropicais e em desenvolvimento (KELLEHER, 2005). Dessa forma, a PSA é um método promissor para a análise de riscos ecológicos na atividade pesqueira, principalmente nesse contexto. Este método pode ser bem eficiente para o manejo pesqueiro, quando usado em conjunto com um programa de monitoramento já em desenvolvimento, uma vez que pode quantificar o impacto das pescarias sobre uma vasta gama de espécies, mesmo aquelas para as quais a ausência de dados é nítida, servindo com um mecanismo de filtro, identificando espécies que correm maior risco e merecem ser monitoradas de maneira mais constante (ZHOU e GRIFFITHS, 2008).

Diante do exposto, as espécies de Centropomidae e *Lutjanus alexandrei*, classificadas como altamente vulneráveis, necessitam de uma maior atenção. Estas são espécies de alto valor econômico nas regiões onde são capturadas (GIGLIO e FREITAS, 2013) e, dado o forte impacto causado pela pesca, precisam de ações de conservação visando especificamente a sua preservação nos ambientes estuarinos, devendo ser classificadas como espécies prioritárias para o manejo. Considerando todos os aspectos discutidos neste trabalho, a elaboração das estratégias de manejo deve sempre ser baseadas na estrutura de todo o ecossistema, não sendo aconselhável levar em consideração apenas algumas espécies. Para todas as áreas estuarinas aqui estudadas, sugere-se a implementação de mecanismos com o intuito de diminuir o impacto da pesca sobre os indivíduos que ainda não atingiram sua maturidade sexual – assim como para aquelas espécies que vem sofrendo um forte impacto causado pela atividade pesqueira – como: a criação de áreas de exclusão da pesca; estabelecimento de períodos de defeso durante o período reprodutivo das espécies e a determinação de tamanho mínimo de malha da rede e/ou tamanho mínimo de captura

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5- CONSIDERAÇÕES FINAIS

Os padrões de biodiversidade funcional, descritos neste trabalho, nos mostram a existência de um padrão entre os estuários de Pernambuco. O Canal de Santa Cruz (Itapissuma) foi considerado a área com maior número de espécies e indivíduos, porém essas espécies são mais próximas do ponto de vista taxonômico. O fato de Itapissuma possuir a maior zona estuarina do estado pode estar ligado às diferenças nas assembleias de peixes, uma vez que tal fator exerce uma grande influência sobre a diversidade, haja visto uma maior disponibilidade de nichos para as espécies que ali se encontram. Apesar dessa grande riqueza geral de espécies, Itapissuma, Suape, Sirinhaém e Rio Formoso são estuários dominados por poucas espécies de peixes, com aquelas pertencentes a família Gobiidae destacando-se em Itapissuma, Mugilidae nos quatro estuários e Gerreidae principalmente em Rio Formoso, apesar de também ser encontrado nas demais áreas.

Informações sobre os padrões de riqueza e diversidade, quando complementados com a diversidade funcional, fornecem uma avaliação mais completa do estado em que se encontra um determinado ambiente, gerando dados como a redundância funcional e complementariedade das espécies entre comunidades. Os estuários de Suape, Sirinhaém e Rio Formoso apresentaram uma baixa diversidade funcional e riqueza de espécies, o que pode estar associado às características desses ambientes, uma vez que são geograficamente próximos, além da existência de impactos antrópicos. Isso mostra a existência de um possível filtro ecológico em relação a Itapissuma, uma vez que este estuário possui uma morfologia bastante irregular, com sua largura variando entre 400 e 1500 m, profundidade de 2 a 5 metros na maré baixa, além da grande descarga de matéria orgânica proveniente de vários rios que nele desaguam. Esses fatores, somados a uma maior área de mangue e a predominância de substrato lamoso, proporcionam uma maior formação de micro-habitats, que são fundamentais para a produção de assembleias mais ricas e morfologicamente diversificada, com espécies morfologicamente distintas. Dessa forma, espécies que apresentam um conjunto de características próprias e estão associadas a determinado tipo de nicho, são consideradas como sendo mais vulneráveis a mudanças antropogênicas, uma vez que o impacto ou perda do habitat podem afetá-las de maneira direta – caso das espécies encontradas em Itapissuma. Por outro lado, espécies mais generalistas podem ser menos sensíveis, já que ocupam nichos mais comuns – caso das espécies abundantes em todas as demais áreas. Isso leva à necessidade de um maior monitoramento desses ambientes, uma vez que alterações deste tipo podem causar alterações na função que as espécies desempenham dentro do ecossistema.

Várias espécies de peixes ocorrem nas áreas estuarinas durante parte do seu ciclo de vida, principalmente durante os seus primeiros anos (de vida). Este é notavelmente o caso dos estuários aqui abordados, o que pode ser confirmado através da grande incidência de organismos juvenis em todas as áreas aqui amostradas. A presença desses organismos é um fator chave para o estabelecimento de áreas de proteção, principalmente quando essa área se caracteriza como uma zona de berçário para várias espécies. Isso mostra a grande importância de levar em consideração a conectividade entre os ambientes na elaboração das estratégias de manejo pesqueiro.

Este é o caso das espécies pertencentes às famílias Centropomidae e Lutjanidae, as quais, foram classificadas como espécies de alto risco na análise de produtividade e susceptibilidade (PSA), para os estuários de Itapissuma e Sirinhaém. Isso se deve à elevada captura de indivíduos juvenis. Adicionalmente a este fato, as espécies pertencentes a estas famílias apresentam crescimento lento e maturação tardia, fatores fundamentais para se conhecer a capacidade de recuperação de determinado organismo.

Diante do exposto, as espécies de Centropomidae, classificadas como altamente vulneráveis, necessitam de uma maior atenção. Estas são espécies de alto valor econômico nas regiões onde são capturadas e, dado o forte impacto causado pela pesca, precisam de ações de conservação visando especificamente a sua preservação nos ambientes estuarinos, devendo ser classificadas como espécies prioritárias para o manejo.

O método aqui aplicado pode ser facilmente utilizado em pescarias com caráter multiespecífico, principalmente naquelas nas quais dados de capturas, para aplicação de modelos tradicionais de avaliação, são deficientes ou até mesmo inexistentes. Dessa forma, este tipo de abordagem pode auxiliar na tomada de decisões referentes a estratégias para planos de gerenciamento e manejo pesqueiro, uma vez que parâmetros de história de vida e dados ecológicos podem ser facilmente utilizados como entrada de dados, flexibilizando mais ainda a sua utilização.

Considerando todos os aspectos discutidos neste trabalho, a elaboração das estratégias de manejo, devem sempre ser baseadas na estrutura de todo o ecossistema, não sendo aconselhável levar em consideração apenas algumas espécies. Para todas as áreas estuarinas aqui estudadas, sugere-se a implementação de mecanismos com o intuito de diminuir o impacto da pesca sobre as populações como: a criação de áreas de exclusão da pesca; estabelecimento de períodos de defeso durante o período reprodutivo das espécies; determinação de tamanho mínimo de malha da rede, além de se sugerir um melhor monitoramento para as regiões que já estão inseridas em zonas de proteção ambiental, como é o caso de Barra de Sirinhaém e Rio Formoso. Outro fator que deve ser considerado é o desenvolvimento de atividades que envolvam os pescadores, principalmente durante os períodos de atividades como a época de defeso e quando ocorrer o estabelecimento de áreas de exclusão de pesca.

Dessa forma, a análise de diversidade clássica, junto com a diversidade funcional e a PSA, são métodos bastante promissores para a análise de riscos ecológicos na atividade pesqueira, principalmente no contexto da falta de informações. Estes métodos podem ser bem eficientes para o manejo pesqueiro, quando usados em conjunto com um programa de monitoramento já em desenvolvimento, uma vez que podem quantificar o impacto das pescarias sobre uma vasta gama de espécies, mesmo aquelas para as quais a ausência de dados é nítida. Além de fornecer uma melhor visão de como os impactos antrópicos estão atuando sobre a biodiversidade e seu funcionamento, análises como a PSA servem com um mecanismo de filtro, uma vez que identificam espécies que correm maior risco e merecem ser monitoradas de maneira mais constante. Os métodos aqui sugeridos podem ser considerados como de particular interesse para a incorporação de estratégias de Manejo de Pesca Baseado em Ecossistema (EBFM), uma vez que estas estratégias visam manter a saúde dos ecossistemas marinhos, bem como tornar sustentáveis as pescarias neles realizadas, de maneira que as espécies não exploradas são consideradas como parte essencial do ecossistema.

Anexos

ANEXO 1

Ocean & Coastal Management 143 (2017) 175–183

Contents lists available at ScienceDirect

Ocean & Coastal Management

journal homepage: www.elsevier.com/locate/ocecoaman

Fish assemblages in tropical estuaries of northeast Brazil: A multi-component diversity approach

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ARTICLE INFO

Article history:
Received 23 December 2015
Received in revised form
21 July 2016
Accepted 4 August 2016
Available online 22 August 2016

Keywords:
Estuaries
Fishes
Taxonomic diversity
Monitoring
Diversity indices
Double principal coordinate analysis
Ecosystem-based fisheries management

ABSTRACT

Biodiversity in estuarine ecosystems suffers from the impact of environmental changes and human activities. This mainly involves changes in temperature, salinity, pollution, habitat degradation or loss and fishing activities. The diversity of species communities is traditionally assessed on the basis of their species richness and composition. However, there is growing interest in taking into account complementary components dealing with species differences (e.g. taxonomic relatedness). In spite of their social, ecological and economic importance, the diversity of tropical estuarine fish assemblages has rarely been monitored by means of a multi-component approach under different human pressure and environmental conditions. We analysed the diversity of exploited fish communities (both target and non-target species) sampled during scientific surveys within four estuarine complexes in the state of Pernambuco, Brazil: Itapissuma, Suape, Sirinhaém, and Rio Formoso. A total of 122 species were collected within 34 samples. Overall, diversity indices and species models fitting dominance-evenness profiles mainly revealed differences between assemblages from Itapissuma, being the largest estuary with wide areas of mangrove, and the other estuaries. While assemblages from Itapissuma generally encompassed more species and individuals than the other estuaries, species were more closely related from a taxonomic point of view. In addition, a Double Principal Coordinate Analysis (DPCoA) established a typology of assemblages, useful for management purposes, and linked to particular fish families: it highlighted differences between Itapissuma, Suape, Sirinhaém and Rio Formoso. This method combines matrices of species abundances and differences (here taxonomic distances according to the Linnean classification). It was particularly accurate with a first factorial plane explaining 73% of the total inertia, while only 17% was achieved by a traditional Principal Component Analysis (PCA). Overall, this study provides an assessment of the state of fish assemblage diversity in Pernambuco estuaries where contrasted human and environmental conditions occur. It underscores the accuracy of using a multi-component diversity approach, with a multivariate analysis that is not yet widely used, for monitoring the diversity of estuaries for ecosystem-based fisheries management purposes.

1. Introduction

Estuaries, interfaces between land and sea, are the site of a range of hydrological (e.g. river discharge), oceanographic and anthropogenic processes. These complex processes create habitats with intense thermal and salinity gradients and variable nutrient and pollutant concentrations (Wolanski, 2007). They notably drive fish patterns of distribution in space and time, and shape the diversity of assemblages (Elliott et al., 2007; Potter et al., 2015). Estuaries are ecologically essential as feeding and breeding grounds, providing appropriate habitats for different stages of the fish life cycle. In addition, they are a migratory pathway for both anadromous and catadromous species (Elliott et al., 2007; Potter et al., 2015).

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http://dx.doi.org/10.1016/j.ocecoaman.2016.08.004

Furthermore, the high socio-economic value of their products, especially as a source of income and food, is widely recognized (Glaser and Diele, 2004; Isaac et al., 2009). However, global environmental change and direct human pressures impact the water quality and biodiversity of estuarine ecosystems (Garcia et al., 2003; Halpern et al., 2008; Borja et al., 2010; Li and de Jonge, 2015). Notably, major changes in fish diversity from the effects of mangrove removal, shrimp farming and agricultural development have been recently documented for estuaries worldwide in terms of assemblage composition, decrease in number of species, feeding groups and larger fishes (Blaber, 2008, 2009, 2013; Weerts and Cyrus, 2002; Singkran and Sudara, 2005; Oribhabor and Ogbeibu, 2010; Viana et al., 2010, 2012).

Aquatic resources are of fundamental importance for human food and for the development of Brazil, which has one of the longest coastlines in the world (9th longest worldwide) (Burke et al., 2001). Recently, Pinheiro et al. (2015) stated that the Brazilian aquatic biodiversity is in peril, notably with regard to fishes, in many cases due to habitat degradation. However, it is widely acknowledged that the success of many fisheries resources is dependent on the quantity and quality of the habitat (Lindall and Thayer, 1982; Auster et al., 1996). The northeast is one of the most densely populated coastal regions in Brazil. In Pernambuco state, the degradation of coastal ecosystems is most severe around the main urban centre, Recife, mainly due to domestic pollution, industrial activity and habitat degradation and loss. These coastal ecosystems are also threatened by fishing (Elfes et al., 2014) because of the impact on the habitat (Turner et al., 1999) and/or fishes at the top of the trophic networks (e.g. trophic downgrading, Estes et al., 2011). Along the Pernambuco coast are located 34 fishing communities, with ca. 12 000 fishermen (Lessa et al., 2006). To supply food and satisfy economic needs, aquatic resources are intensively exploited using multiple techniques and multispecies artisanal fisheries (Frédou et al., 2006, 2009a, b). The majority of stocks are either fully or over-exploited, and there is little room for expansion into new fisheries (MMA, 2006). Despite all these threats, the degree of dependency on and use of estuarine ecosystems, even in urban areas, remains very high (MMA, 2002).

The diversity of the threats and their intensification in the Pernambuco estuaries (Lira et al., 2010a, b) highlight the need to monitor fish biodiversity for management and sustainable use purposes. While several studies dedicated to the diversity of fish assemblages have been carried out in this area (Paiva et al., 2009; Paiva and Araújo, 2010), there is still a need to assess and quantify their state in the main estuaries subjected to contrasting human pressure and environmental conditions. For this purpose, the diversity of fish assemblages can be monitored more exhaustively by a multi-component diversity approach (e.g. Wilsey et al., 2005; Mérigot et al., 2007; Gaertner et al., 2010; Lefcheck et al., 2014). This approach consists in assessing different components, or facets, such as species number and evenness, and also components that explicitly take into account the degree of difference among species. This reflects the taxonomic relatedness between species, on the basis of which diversity indices can be more sensitive to environmental and/or human drivers (Clarke and Warwick, 1998). Monitoring a single diversity component using one or few diversity indices leads to an incomplete description of the diversity of assemblages, and in turn important responses in the face of environmental and/or human drivers could be missed (e.g. Stuart-Smith et al., 2013; D'agata et al., 2014).

In this context, the aim of this work is to investigate the fish diversity assemblages on the basis of a multi-component approach, comparing tropical estuaries subjected to contrasting environmental conditions and human pressure. Firstly, complementary diversity indices were quantified for each estuary. Secondly, the typology of fish assemblages among these estuaries was characterized using a recent multivariate analysis, the Double Principal Coordinates Analysis (DPCoA), that explicitly takes into account species differences (here, taxonomic relatedness). Overall, this study provides an assessment of the state of assemblages among the studied tropical estuaries of northeast Brazil, and demonstrates that a multi-component approach of this kind can be useful for ecosystem-based fishery management in estuarine ecosystems.

2. Materials and methods

2.1. Study area

Four estuaries located along the coast of Pernambuco State were studied: Itapissuma, Suape, Sirinhaém and Rio Formoso (Fig. 1). The estuarine complex of Itapissuma is located to the north of and closest to the Recife and Olinda urban centres, and is the most strongly influenced by landscape alterations, industrial discharges, shrimp farming and high fishing pressure (CPRH, 2003; Lira et al., 2010a; Gondim, 2015). The other three estuaries are located south of Recife: the Suape estuary has suffered the high impact of deforestation and land reclamation due to the construction of a large industrial port complex (chemical, shipping and logistics companies), with predictable effects on both the biodiversity and fisheries productivity (Paiva and Araújo, 2010). Sirinhaém and Rio Formoso are located in the southernmost area, and although located within an environmental protected area (APA Guadalupe), they are also impacted by agricultural pollution (mainly sugar cane), shrimp farming and fishing (Lira and Fonseca, 1980). Rio Formoso is also a popular tourism destination (Lira et al., 2010b).

2.2. Sampling design

Fish were sampled from the end of 2012–2014 (see Table S1 in Supplementary Material for sampling dates). A total of 34 samples were obtained using block net and beach seine (seine) gear: 17 in Itapissuma, 9 in Sirinhaém, 4 in Rio Formoso and 4 in Suape. These types of fishing gear were selected as, together, they can cover most fish habitats present in the studied estuaries: shallow coastal areas and sand/mud banks (beach seine), and flooded mangrove and channels (block net). The number of samples per estuary varied according to the characteristics of each type of gear, the influence on tidal variation, and the gear setting time. Beach seine nets measured 80–180 m long, 5 m high with a mesh size of 20 mm. The operation lasted no more than 20 min and was repeated several times. Block nets were set along the mangrove forest and channels and measured 70–90 m long, 2.5 m high with a mesh size of 50 mm. At low tide, the net was anchored to the bottom. At slack high-water, the net was deployed and attached to stakes and pulled taut 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).

2.3. Statistical analyses

Firstly, we computed a species accumulation curve to assess whether the fish community in the study area was exhaustively sampled, with the gear used (Gotelli and Colwell, 2001). We used a randomized method without replacement. It enables calculation of a mean number of species for a given number of samples, with a 95% confidence interval, as well as obtaining the total number of species caught for the total number of samples considered. We also computed the Chao2 index (Chao, 1987), which extrapolates, from the occurrence of rare species, the total expected number of species in the area, for the given sampling gear. In addition, because the

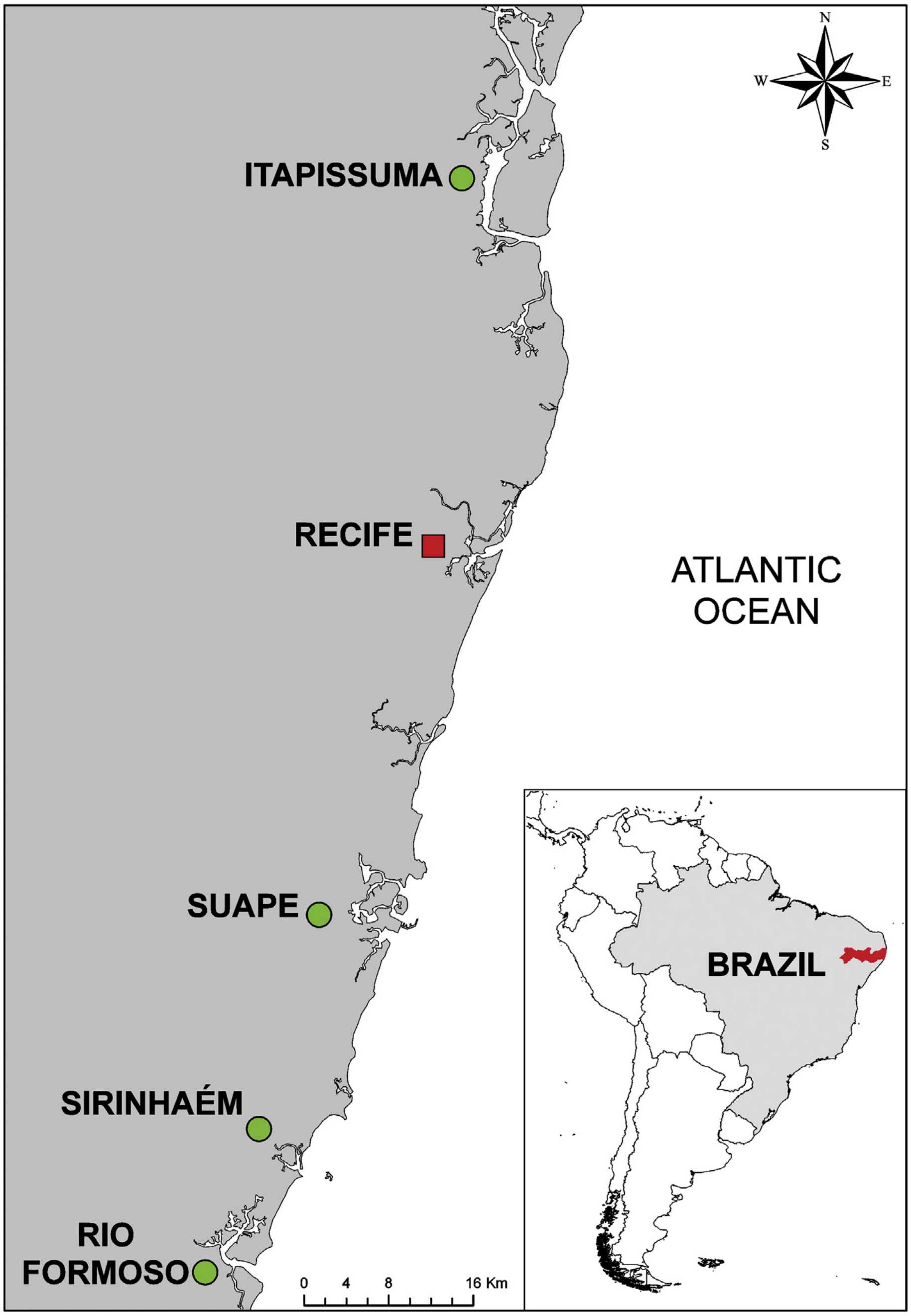

Fig. 1. The study area, Pernambuco, Brazil.

number of samples differed among estuaries (see above), we computed the total rarified number of species for each estuary for a common number of four samples (i.e. minimum sampling size), following the same randomized procedure as described above. To investigate the overall dominance/evenness pattern among the four estuaries, raw rank-abundance curves and species abundance models were also computed (Fig. S1 in Supplementary Materials).

Secondly, for each assemblage (i.e. each of the 34 samples), we computed diversity indices related to three major components of species diversity (species richness, evenness and species taxonomy; see Table 1 for indices properties that justify *a priori* their complementary use, further details in Mérigot et al., 2007). Then, we studied the correlations among these indices to identify a set providing complementary patterns/responses for the studied estuaries (Mérigot et al., 2007; Gaertner et al., 2010, 2013). This latter step was performed using a principal component analysis (PCA) based on Spearman rank correlation coefficient (see Mérigot et al. (2007) for further details). A correlation coefficient of approximately <0.5 meant that indices were complementary.

Subsequently, each of the complementary indices identified above were compared between the four estuaries with a Kruskal-Wallis test (i.e. a non-parametric ANOVA), due to the relatively low and uneven number of samples. When the null hypothesis of this test was rejected (i.e. homogeneity in distributions of rank values between estuaries), the non-parametric multi-comparison test of Siegel and Castellan (1988) was computed to assess the pairwise differences between estuaries (i.e. post-Hoc test).

Finally, a multivariate analysis named Double Principal

Table 1
Species diversity components and indices studied. x_i ($i = 1, \ldots, S$) denotes the abundance of the ith species, N ($= \Sigma_i x_i$) is the total number of individuals in the sample, p_i ($= x_i/N$) is the proportion of all individuals belonging to species i, $N_{\max}$ is the number of individuals of the most abundant species. d_{ij} is the distance given to the path length linking species i to the first common node with species j in the hierarchical taxonomic classification.

Component	Descriptor name	Formula	Expected properties	Reference
Number of species	Species richness	S = Number of species in a sample	Species richness per sample	
	Margalef	$D_{\mathrm{mg}} = \frac{(S-1)}{\ln(N)}$	Adjusted species richness by N	Margalef (1958)
Evenness	Heip	$E_{\mathrm{Heip}} = \frac{\exp(H\prime)-1}{S-1}$	Sensitive to rare species	Heip (1974)
	Simpson evenness	$E_{1-D} = \frac{1-(\sum_{i=1}^{S} p_i^2)}{1-1/S}$	Sensitive to dominant species	Smith and Wilson (1996)
	Berger Parker	$\frac{1}{d} = \frac{N}{N_{\max}}$	Sensitive to dominant species	Berger and Parker (1970)
Number of species + evenness	Shannon-Wiener	$H\prime = -\sum_{i=1}^{S} p_i \log p_i$	Sensitive to rare species	Shannon and Weaver (1949)
	Simpson diversity	$1 - D = 1 - (\sum_{i=1}^{S} p_i^2)$	Sensitive to dominant species	Simpson (1949)
Species taxonomy	Quadratic entropy	$Q = \sum_{i=1}^{S-1} \sum_{j=i+1}^{S} d_{ij} p_i p_j$	Extension of $1-D$ including taxonomic relatedness	Rao (1982)

Coordinate Analysis (DPCoA) (Pavoine et al., 2004) was implemented to provide a typology of fish assemblages (i.e. particular assemblages identified from the species, their taxonomy and number of individuals sampled). Compared to traditional multivariate methods such as the Principal Component Analysis (PCA), the DPCoA has the advantage of explicitly taking into account the species difference. It is computed from both a species abundance matrix and a species distance matrix (here based on taxonomic relatedness between species, e.g. Clarke and Warwick, 1998). The DPCoA provides a multidimensional space assembling the species and the samples to establish a typology of assemblages. We identified the contribution of the taxonomic family to assemblages among studied estuaries on the two first factorial axes (i.e. those which explained the main variability). Then, the optimum number of family groups that differentiate assemblages on these axes was defined by means of the Gap statistic (Tibshirani et al., 2001).

All the statistical analyses and diversity indices mentioned above were performed using R software (R development Core R Core Team, 2015).

3. Results

3.1. *Patterns of fish assemblage indices*

A total number of 122 fish species were identified in the 34 samples of the study area. The species accumulation curve did not stabilise towards asymptotic values (Fig. 2), as is usually the case in ecological studies, due to the large number of rare species present in natural communities. However, the slope at the end of the curve suggested that the number of additional samples necessary to collect the species more exhaustively with the fishing gear used should only be slightly higher. It is congruent with the Chao2 index which estimated the total number of species in the studied area sampled with these types of gear at 139.29 ± 8.58 (standard deviation) species. For the same number of samples per estuary (here four samples), the total rarified number of species revealed that Itapissuma was the richest, with 58.88 ± 8 species, while the other estuaries encompassed fewer species, with 36.56 ± 4.9, 35 and 32 species for Sirinhaém, Suape and Rio Formoso, respectively.

The identity and the proportion of the most dominant species differed across the four estuaries (Table 2). *Diapterus auratus* was the most dominant at Rio Formoso (60.4%), followed by *Gobionellus stomatus* in Itapissuma (50%), and *Sphoeroides testudineus* at Sirinhaém and Suape (29% and 23.6%, respectively (Table 2). The cumulative percentage of the 10 most abundant species in the study area varied according to the estuary: the highest value was recorded at Rio Formoso (91.7%) and the lowest at Sirinhaém (78.7%, Table 2). Furthermore, the whole dominance/evenness profile differed between estuaries according to the raw rank-abundance curves and species abundance models (Fig. S1 in Supplementary Materials).

The correlation analysis revealed that abundance N, species richness S and Quadratic entropy Q were complementary to describe patterns of the studied assemblages, providing proxies of the five others indices (see Materials and Methods section). Thus, only these three indices were considered for further analyses. N, S and Q revealed that estuaries had different levels of diversity (Fig. 3, Kruskal-Wallis tests, 12.19 < Chi2<18.33, d.f. = 3, p < 0.01), especially between Itapissuma and Sirinhaém (multiple comparisons tests, p < 0.05). The mean number of individuals and species per sample were higher at Itapissuma (i.e. N=2494.59 ± 1467.31 individuals and S=29.82 ± 9.62 species, respectively). In contrast, the mean taxonomic diversity at Itapissuma ($Q = 2.14 \pm 0.47$) was lower compared to Sirinhaém and Suape (i.e. 2.72 ± 0.42 and 2.73 ± 0.19, respectively, see also quartiles distributions in Fig. 3). Q corresponds to the mean taxonomic distances between two randomly chosen individuals in a sample. Thus, while Itapussima had a higher mean number of species per sample, it encompassed more closely related individuals than the other two estuaries.

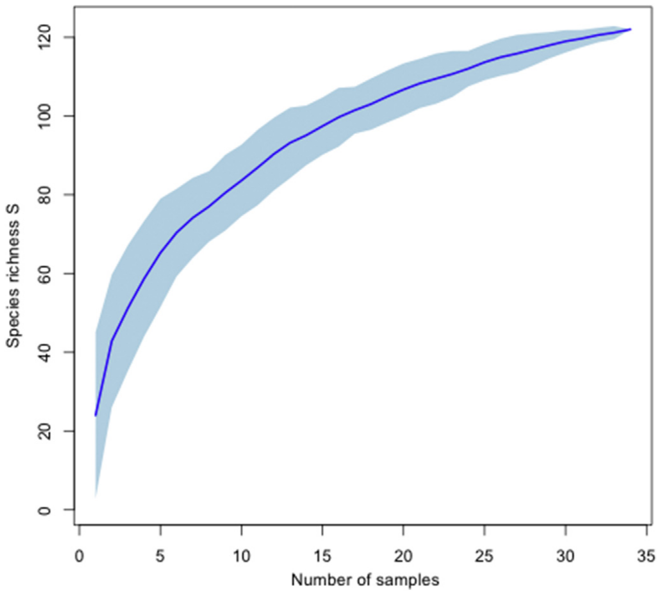

Fig. 2. Species accumulation curve of the studied area, computed by a random method without replacement. Mean species richness value ± 95% confidence interval.

Table 2
Abundance, proportion and cumulative proportion for the ten most dominant species in the four studied estuaries: Itapissuma ITA, Sirinhaém SIR, Suape SUA, Rio Formoso RIO.

	Rank	Abundance	Proportion (%)	Cumulative prop. (%)
ITA				
Gobionellus stomatus	1	21207	50	50
Eucinostomus argenteus	2	2158	5.1	55.1
Ctenogobius stigmaticus	3	2064	4.9	60
Mugil curvidens	4	1966	4.6	64.6
Gobionellus oceanicus	5	1921	4.5	69.1
Sphoeroides testudineus	6	1773	4.2	73.3
Eucinostomus gula	7	1673	3.9	77.3
Cetengraulis edentulus	8	1597	3.8	81
Archosargus rhomboidalis	9	1259	3	84
Achirus lineatus	10	981	2.3	86.3
SIR				
Sphoeroides testudineus	1	237	29	13.8
Sciades herzbergii	2	222	31.1	26.7
Diapterus auratus	3	145	15.3	35.2
Achirus lineatus	4	132	15.2	42.9
Centropomus undecimalis	5	132	12.9	50.6
Centropomus parallelus	6	119	12.1	57.5
Diapterus rhombeus	7	116	12.9	64.3
Bairdiella ronchus	8	87	9.3	69.3
Mugil curvidens	9	83	8.9	74.2
Aspistor luniscutis	10	78	12.9	78.7
SUA				
Sphoeroides testudineus	1	256	23.6	23.6
Diapterus auratus	2	195	18	41.6
Centropomus parallelus	3	113	10.4	52
Lutjanus alexandrei	4	101	9.3	61.3
Centropomus mexicanus	5	65	6	67.3
Mugil curema	6	46	4.2	71.5
Eugerres brasilianus	7	45	4.1	75.7
Mugil curvidens	8	37	3.4	79.1
Caranx latus	9	36	3.3	82.4
Centropomus undecimalis	10	36	3.3	85.7
RIO				
Diapterus auratus	1	1835	60.4	60.4
Sphoeroides testudineus	2	277	9.1	69.5
Eugerres brasilianus	3	147	4.8	74.4
Diapterus rhombeus	4	146	4.8	79.2
Mugil curema	5	104	3.4	82.6
Caranx latus	6	66	2.2	84.8
Centropomus undecimalis	7	64	2.1	86.9
Centropomus parallelus	8	60	2	88.9
Centropomus mexicanus	9	44	1.4	90.3
Centropomus pectinatus	10	41	1.4	91.7

3.2. *Typology of fish assemblages*

The first factorial plane of the DPCoA was particularly accurate for studying the typology of assemblages, as it explained 73.20% of the total variability (axis 1: 54.19% and axis 2: 19.01%), while a PCA (i.e. without taking into account taxonomic relatedness) explained only 17.39%. Sampled assemblages were different between estuaries, especially between Itapissuma and the other three, as highlighted by axis 1, explaining most of the variability (see barycenter locations in Fig. 4a; supported by a Kruskal Wallis test performed on the samples coordinates of axis 1, Chi2 = 25.84, d.f. = 3, $p < 0.001$, followed by a non-parametric multiple comparisons test). To a lesser extent on axis 2, assemblages from Rio Formoso were differentiated from the other ones (Chi2 = 9.90, d.f. = 3, $p < 0.05$).

Species families among assemblages of estuaries were not strictly the same, even if some were shared. More specifically, five groups of species families were identified on the first factorial plane (Fig. 4b, Gap statistic computed from 1000 bootstraps). On the one hand, Mugilidae and Gerreidae were associated with Suape, Sirinhaém and Rio Formoso assemblages (Fig. 4b, families with the highest contributions on the 1st and 2nd axes, see the left part of the figure). On the other hand, some species of Gobiidae were only sampled at Itapussima (Fig. 4b, contributions of the right part of the 1st axis).

4. Discussion

4.1. *Patterns of fish diversity in estuaries*

Our results highlighted that fish assemblages differ among the four estuaries studied in Pernambuco state. Dominance-evenness profiles varied between Itapissuma and Suape compared to those of Rio Formoso and Sirinhaém (Table 2 and Fig. S1 in Supplementary Materials). In addition, Itapissuma had higher mean number of species and individuals per sample than the other estuaries (Fig. 3), but individuals were more closely related from a taxonomic point of view, especially compared to Sirinhaém and Suape (Fig. 3). Different typologies of species assemblages were highlighted by the DPCoA (Fig. 4), taking into account both their abundances and species taxonomic relatedness. Differences between i) Itapissuma ii) Rio Formoso and iii) Sirinhaém, Suape were indeed related to particular species families (Fig. 4).

Estuaries within a given region are likely to encompass similar fish assemblages (Whitfield, 1999; Harrison and Whitfield, 2006). However, the four estuaries studied here are of different

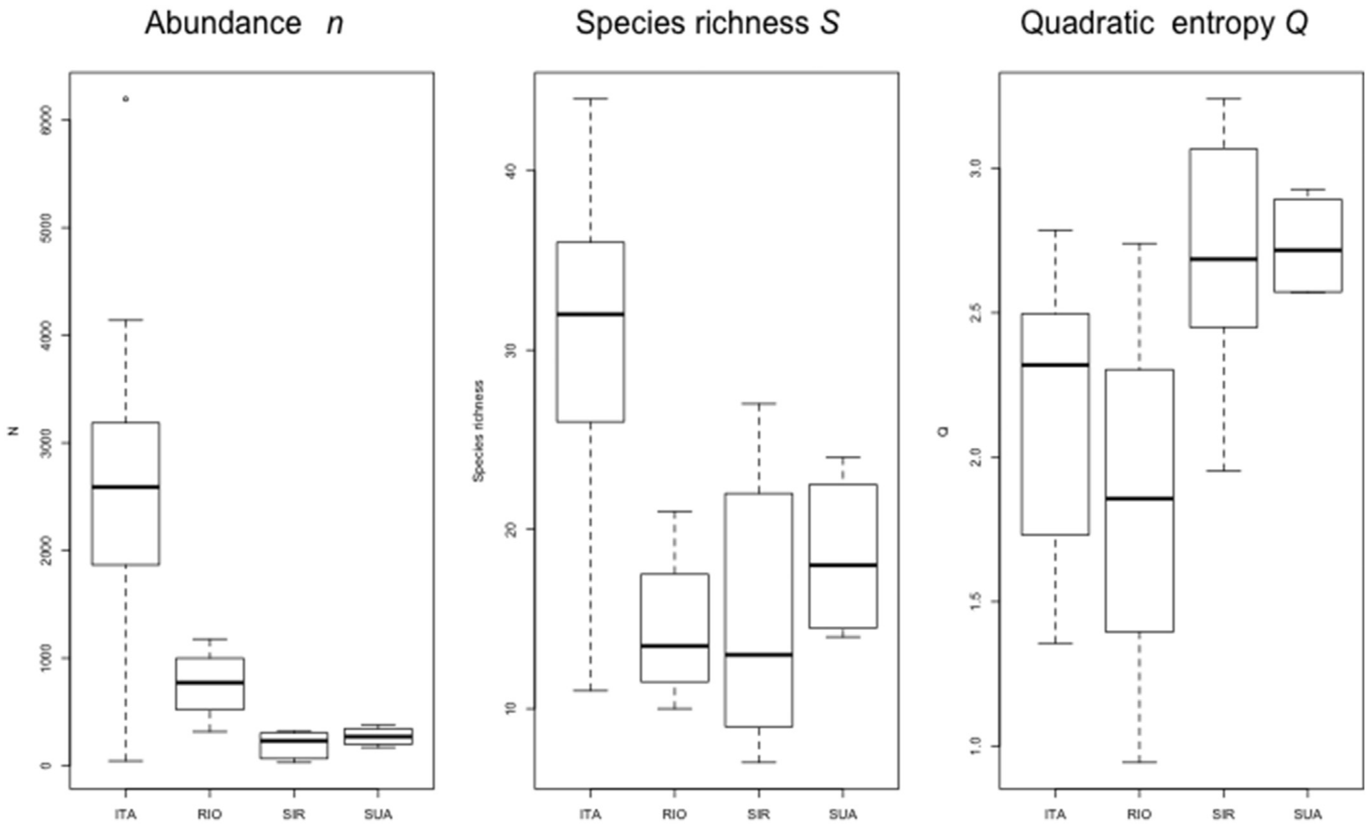

Fig. 3. Boxplots of species richness *S*, abundance *n* and quadratic entropy *Q* (see Table 1) according to the estuary: Itapissuma ITA, Sirinhaém SIR, Suape SUA, Rio Formoso RIO.

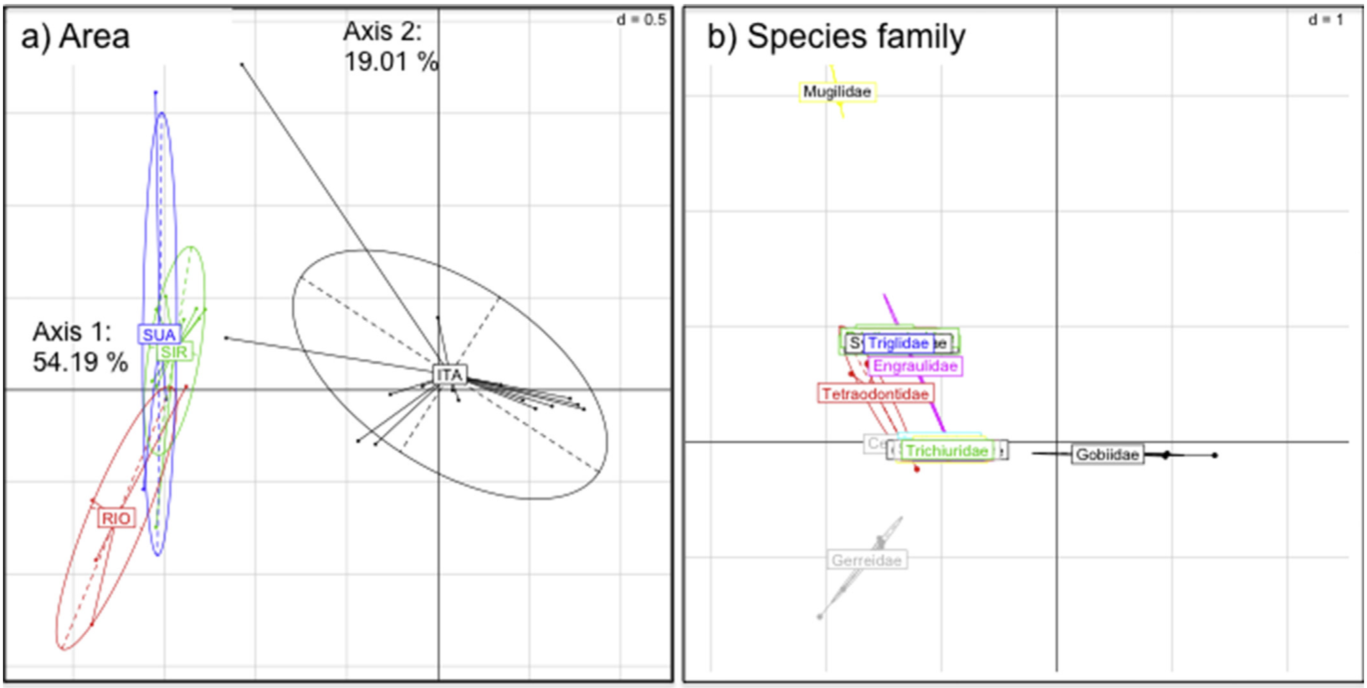

Fig. 4. Double Principal Coordinate Analysis (DPCoA), first factorial plane explaining 73.20% of the total inertia. Projections of a) samples according to the estuary: Itapissuma ITA, Sirinhaém SIR, Suape SUA, Rio Formoso RIO; b) species family.

morphology and size, and are subject to varying degrees of human pressures and abiotic conditions, which in turn lead to particular fish assemblages. Firstly, the overall results highlighting that the diversity of assemblages from Itapissuma was mainly different from that of the other estuaries corroborates the fact that this estuary, composed of different rivers, is considered as the largest estuarine ecosystem in Pernambuco state (Silva et al., 2011). In contrast, the others estuaries are of more similar size and are geographically closer (south of Recife, Fig. 1). The size of the estuary influences fish assemblages (e.g. Selleslagh et al., 2009; Nicolas et al., 2010; Cardoso et al., 2011; Pasquaud et al., 2015), notably by increasing the availability of species niches.

Secondly, Itapissuma is a homogeneous muddy estuary, in contrast to the other estuaries. Assemblages differed especially due to the relatively high abundance of some species of Gobiidae, only sampled at Itapissuma (note that the other estuaries featured different species of this family) (Table 2, Figs. 3 and 4). Gobiidae represent a large part of larval fish assemblages in most tropical estuaries (Blaber, 1997), with long larval durations influencing dispersal and colonization (Shen and Tzeng, 2008). They play an important role in estuarine ecosystems (Bruno and O'Connor, 2005), notably as omnivores in the trophic structure of Brazilian tropical estuaries (Joyeux et al., 2004) and worldwide (Whitfield, 2005; Sanvicente-Anorve et al., 2003; Polgar and Bartolino, 2010). The fish larvae of the Santa Cruz Channel (Itapissuma) was investigated by Ekau et al. (2001) and Gobiidae were among the dominant group. Thus, these results highlight the potential importance of several species of this family in the diversity of Itapissuma assemblages, which should contribute to their trophic structure in a significant way, notably due to their dominance (Table 2).

In addition, while Mugilidae were sampled in all estuaries, they were more abundant at Itapissuma (Table 2, Figs. 3 and 4). This is congruent with the fact that this family, encompassing iliophagous species, is well represented in tropical estuaries (Blaber, 1997). These areas are known to be detritus traps, with an abundance of micro −algae and −fauna, especially in mangrove areas and near mudflats (Hutchings and Saenger, 1987). Likewise, Gerreidae were sampled in all estuaries, but more predominantly at Rio Formoso (Table 2, Figs. 3 and 4). Paiva et al. (2009) also found Gerreidae among the dominant families in Rio Formoso. This family is also characteristic of tropical estuaries (Blaber, 2008). They have a particular morphology and feeding pattern of eating slower moving or small invertebrates, but they can adapt to feed on other resources such as bivalve siphon tips (see Blaber, 2008 and references therein). The fact that Gerreidae were more abundant at Rio Formoso than in the other areas, could suggest a particular resource availability in this estuary that is part of a multiple use state environmental protection area, and is also located in proximity to protected reef areas (Ferreira et al., 2006), which could enhance connectivity (Ramos et al., 2016).

Finally, assemblages among the studied estuaries were dominated by few species (Table 2, Fig. S1 in Supplementary Materials), as observed in other areas worldwide (Cabral et al., 2001; Akin et al., 2005; Maes et al., 2005; Elliott et al., 2007). Most of these species are partly or totally dependent on mangroves, the dominant habitat in tropical estuaries (dependence was relatively predominant within Itapissuma compared to the other estuaries), used for all or part of their life cycle (Blaber, 2008). In addition, the fact that juveniles of marine species were also sampled during our surveys is particularly interesting with regard to the connectivity, production and management of sea fisheries (Blaber, 2009). It notably underlines the need to avoid mangrove habitat loss, which is also usually followed by an increase in fishing pressure with the use of more efficient gear.

Furthermore, the use of the estuary at different stages in the species life cycle also underlines the importance of performing temporal surveys. Throughout this study, seasonal variations could not be assessed yet because of the insufficient number of samples in some estuaries, but further surveys would be useful to investigate this. Different aspects of fish assemblages, such as diversity, species composition, ecological and feeding guilds distribution, can have contrasted patterns in relation to seasons, depending on the use of the estuaries, such as, for instance, permanent occupancy, temporal nursery or migration (Gordo and Cabral, 2001; Pombo et al., 2007; Selleslagh and Amara, 2008; Cardoso et al., 2011). Such temporal assessment should thus offer a more in-depth extension of this study that constitutes a starting point by characterizing the state of fish assemblages in estuaries subjected to contrasting human pressure and environmental conditions. It provides a first step towards monitoring assemblages as a basis for subsequently assessing the potential future human impact and environmental changes as an aid to fisheries management.

4.2. *Implications for fisheries management*

In order to implement the most widely accepted fisheries approaches, such as the code of conduct of responsible fisheries (Pitcher et al., 2009), Brazil is expected to incorporate ecosystem-based fishery management (EBFM), for which the non-exploited species are an essential part of the ecosystem (Pikitch et al., 2004). The multi-component approach we used to quantify fish assemblages diversity is thus directly part of the EBFM (Mérigot et al., 2007; Gaertner et al., 2010, 2013). In addition, the DPCoA was more powerful than a traditional PCA to identify assemblages among estuaries (73% and 17% of total variation explained by the first factorial plane, respectively). It underscores the advantage of including species taxonomic differences in this multivariate analysis to take into account more realistically of the species ecology. However, the DPCoA has to date been poorly used for studying fish assemblages (see Pavoine et al., 2009; based on phylogenetic data). Our results suggest that this method could be of particular interest for EBFM.

Many fish species inhabit or occur in estuaries during juvenile phase, with estuary-coastal ontogenetic migrations after the first few years of life (Sheaves et al., 2015). This is notably the case in the four investigated estuaries, as underlined by the presence of juveniles sampled during surveys (see above, and Aschenbrenner and Ferreira (2015), more specifically for Lutjanid species in the study area). This is a key component of nursery habitat value (Sheaves et al., 2015) that enhances the importance of considering habitat and ecosystem connectivity in management strategies. The tropical estuaries in the present study are part of ecosystem mosaics with seagrasses, mangroves and costal reefs (Ferreira et al., 2016) that together are supposed to maintain species life cycles and biodiversity. Mosaics of Marine Protected Areas (MPAs) have increased potential to favor EBFM in fisheries management and to extend its range of application (Seixas and Veira, 2015). However, MPA areas have only been implemented on the southern part of the study area (Rio Formoso and Sirinhaem complexes), and measures taking into account EBFM in fisheries management processes are generally lacking in the studied region. Community-based management options, such as extractive reserves, where both sustainable fisheries and conservation objectives are targeted, although facing challenges Santos and Schiavetti, 2014), remain promising approaches to reduce conflicts and maintain biodiversity.

Acknowledgments

We thank three anonymous reviewers for their constructive comments on an earlier version of the manuscript. We thank all students and researchers who participated in field collections. We are also grateful to the Conselho Nacional de Desenvolvimento Científico e Tecnológico (CNPq) for providing a research grant to F Lucena Frédou, T. Frédou and B. Ferreira. This work was carried out under the National Institute on Science and Technology in Tropical Marine Environments - INCT-AmbTropic (CNPq process 565054/2010-4). We thank Michael Paul for correcting the English of the paper.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.ocecoaman.2016.08.004.

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ANEXO 2

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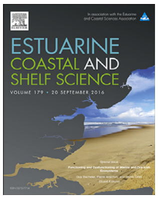
Estuarine, Coastal and Shelf Science xxx (2016) 1–8

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss



Functional diversity of fish in tropical estuaries: A traits-based approach of communities in Pernambuco, Brazil

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ARTICLE INFO

Article history:
Received 28 December 2015
Received in revised form
20 August 2016
Accepted 27 August 2016
Available online xxx

Keywords:
Monitoring
Diversity indices
Double principal coordinate analysis
Estuarine ecosystems
Species function

ABSTRACT

Environmental changes and human activities may have strong impacts on biodiversity and ecosystem functioning. While biodiversity is traditionally based on species richness and composition, there is a growing concern to take into account functional diversity to assess and manage species communities. In spite of their economic importance, functional diversity quantified by a traits-based approach is still poorly documented in tropical estuaries. In this study, the functional diversity of fishes was investigated within four estuaries in Pernambuco state, northeast of Brazil. These areas are subject to different levels of human impact (e.g. mangrove deforestation, shrimp farming, fishing etc.) and environmental conditions. Fishes were collected during 34 scientific surveys. A total of 122 species were identified and 12 functional traits were quantified describing two main functions: food acquisition and locomotion. Fish abundance and functional dissimilarities data were combined into a multivariate analysis, the Double Principal Coordinate Analysis, to identify the functional typology of fish assemblages according to the estuary. Results showed that Itapissuma, the largest estuary with a wider mangrove forest area, differs from the other three estuaries, showing higher mean values per samples of species richness *S* and quadratic entropy *Q*. Similarly, it presented a different functional typology (the first two axes of the DPCoA account for 68.7% of total inertia, while those of a traditional PCA based solely on species abundances provided only 17.4%). Conversely, Suape, Sirinhaém, and to a lower extent Rio Formoso, showed more similarity in their diversity. This result was attributed to their predominantly marine influenced hydrological features, and similar levels of species abundances and in morphological traits. Overall, this study, combining diversity indices and a recent multivariate analysis to access species contribution to functional typology, allows to deepen diversity assessment by providing additional information regarding the functional pattern of fish assemblages.

1. Introduction

Estuaries are highly productive and important providers of ecosystem services, included among the most biologically valuable areas of the world (Costanza et al., 1997). However, human activities, including physical and chemical transformations, habitat destruction and changes in biodiversity impacted these environments in recent years (Borja et al., 2010; Halpern et al., 2008, Halpern et al., 2007). Thus, the assessment of the biodiversity in estuarine regions through functional diversity is crucial, which relates to the species role in the ecosystem and potentially to its resistance and/or resilience (Paiva and Araújo, 2009; Villéger et al., 2010; Wolanski and Donald, 2011; Mouchet et al., 2013; Vilar et al., 2013).

Different indexes describe and quantify the diversity of a particular community, such as number of species, evenness and/or more complex variations between species like taxonomy, phylogeny and function (Devictor et al., 2010; Granger et al., 2015; Pavoine

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http://dx.doi.org/10.1016/j.ecss.2016.08.030

Please cite this article in press as: Silva-Júnior, C.A.B., et al., Functional diversity of fish in tropical estuaries: A traits-based approach of communities in Pernambuco, Brazil, Estuarine, Coastal and Shelf Science (2016), http://dx.doi.org/10.1016/j.ecss.2016.08.030

and Bonsall, 2011; Stuart-Smith et al., 2013). Recent studies showed that traditional indexes are limited as they consider all species equivalent (Magurran, 2004; Mouchet et al., 2010; Villéger et al., 2010). The incorporation of functional features of species have generated different patterns and responses, sometimes more sensitive in detecting changes in communities (Bender et al., 2013; Buisson et al., 2013; D'agata et al., 2014; Martins et al., 2012). A functional traits-based approach may be valuable, even for applied studies focused on conservation and restoration decisions (Cadotte et al., 2011). In this context, functional diversity of communities assumes mechanistic links with niche and functional aspects of species, and thus provides a pragmatic assessment of niche mismatch/overlap between species (Cadotte et al., 2011). Functional diversity can be defined as the value and range of the functional traits of an organism present in a community (Petchey and Gaston, 2006). Functional traits are defined as “morpho-physio-phenological traits which impact fitness indirectly via their effects on growth, reproduction and survival, the three components of individual performance” (Violle et al., 2007).

Despite the growing interest for functional diversity of fishes (Resetarits and Chalcraft, 2007; Halpern and Floeter, 2008; Albouy et al., 2011; Granger et al., 2015), few studies considered a trait-based approach for fish communities in tropical estuaries (Mouchet et al., 2013; Villéger et al., 2010, 2012). In addition, the general lack of standardized fisheries databases prevents the assessment of the effects of natural and/or anthropogenic factors on the fish diversity in these ecosystems. This is notably the case for estuaries in Pernambuco state, northeastern region of Brazil, with a coastline extension of 187 km encompassing 34 fishing communities distributed along 15 coastal municipalities (FIDEM, 1987). There are 17 estuarine zones, characterized by the strong presence of mangroves (Paiva and Araújo, 2010), many of which are impacted mainly by deforestation, erosion, pollutants from the sugar industry and sewage, as well as shrimp farming (Souza and Sampaio, 2001; CPRH, 2003a, 2003b). Such factors may affect aquatic communities with perturbations of the biological cycle of marine and estuarine species, loss of biodiversity and reduction in fisheries productivity (Paiva and Araújo, 2010). However, there is no study providing a traits-based approach on the functional diversity of estuarine fish in these environments.

In this context, this study assesses the functional diversity among estuaries with different levels of impacts and environmental features, using a fish functional traits-based approach. It highlights the usefulness of a recent multivariate analysis (Double Principal Coordinate Analysis - DPCoA) (Pavoine et al., 2004) to assess the species contribution in functional typology and deepen the diversity assessment of estuarine fishes. The analysis was carried out in two steps: firstly, diversity indices of species richness and the mean functional differences between individuals were quantified per sample for each estuary. Secondly, functional typology of fish assemblages among these estuaries was characterized by this multivariate analysis and compared to a traditional Principal Component Analysis to provide a baseline diagnostic for future monitoring in an ecosystem-based fishery management.

2. Material and methods

2.1. Study site

Four estuaries located along the coast of Pernambuco State were studied: Itapissuma, Suape, Sirinhaém and Rio Formoso. The estuarine complex of Itapissuma is located to the north and closest to Recife and Olinda urban centres, and the most influenced by landscape modifications, industrial discharges, shrimp farming and high fishing pressure (CPRH, 2003a; Lira et al., 2010b; Gondim, 2015). The other three estuaries are located south of Recife: the Suape estuary has recently suffered the high impact of deforestation and land reclamation due to the construction of a large industrial port complex, with expected effects on both biodiversity and fisheries productivity (Paiva and Araújo, 2010). Sirinhaém and Rio Formoso are located in the southernmost area, and although located within an environmental protected area (APA Guadalupe), they are also impacted by agricultural pollution (mainly sugar cane), shrimp farming and fishing (Lira and Fonseca, 1980). Rio Formoso is also a popular tourism destination (Lira et al., 2010b).

2.2. Data collection

Thirty-four samples were collected at four estuaries from the Pernambuco coast (Fig. 1): Itapissuma, Suape, Sirinhaém and Rio Formoso (17, 4, 9 and 4 samples, respectively) from October 2012 to December 2014 (see dates of each sampling in the Table 2 in supplementary material online). Two complementary fishing gears were used: block net and beach seine. These fishing gears were selected as, together, they cover most fish habitats: shallow coastal areas and sand/mud banks (beach seine) and flooded mangrove and channels (block net) (Mérigot et al., 2016). Due to their geomorphology and floor, in Sirinhaém and Itapissuma both fishing gear were used and, in Suape and Rio Formoso, only the block net was utilized. The number of samples per estuary varied due to the characteristics of each gear, the dependence of tidal variation, and the sampling time for a given gear. Block nets were set along the

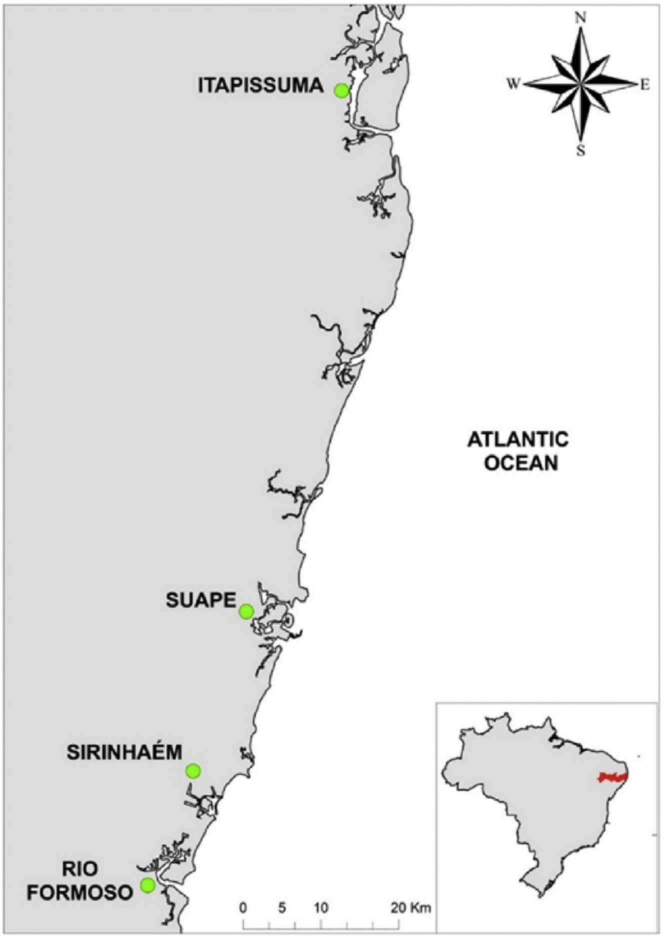

Fig. 1. Estuaries sampled along the Pernambuco Coast - Brazil.

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mangrove forest and channels measured from 70 to 90 m long and 15 m high and used along the mangrove forest. Blocking initiated at the end of the high tide and continued throughout the entire ebb tide cycle (6 h). The beach seine ranged from 80 to 180 m long and 5 m high. The seining lasted no more than 20 min and it was repeated several times.

2.3. *Functional characterization of species*

Two key functions, food acquisition and locomotion, that relate to twelve ecomorphological traits, assimilated to functional traits, were considered (Villéger et al., 2010). The functional traits, based on morphoanatomical measurements, of 122 species were calculated. A maximum of 10 individuals by species was randomly selected for measurement. When the 10 individuals were not available, pictures from the internet were supplied ("Fishbase website"; Froese and Pauly, 2016). The traits were computed as ratios (Albouy et al., 2011; Mason et al., 2007; Villéger et al., 2010) of each individual (Fig. 2; Table 1), assuming that intraspecific variations were smaller than interspecific variations (Dumay et al., 2004; Villéger et al., 2012). Body width (Bw), mouth width (Mw) and mouth depth (Md) were measured with an electronic calliper (precision of 0.1 mm). The other measures were performed with digital photography (Sony DSC-W830, 20.1 Mega Pixels) and the free software Image J (Schneider et al., 2012), with 0.1 mm of accuracy.

Each functional trait was standardized so that the mean was equal to 0 and standard deviation 1. Functional pairwise-distances between species were computed on standardized traits using the Euclidean distance. Only adult specimens, dominants in the samples, were used for measurements, to avoid bias associated to ontogenetic variations, since such differences could affect some species. Relationships between the functional traits and species obtained with a Principal Component Analysis are provided as Figs. 1 and 2 in the supplementary material online.

Table 1
Functional traits of fishes. See codes of morphometric measurements in Fig. 2.

	Functional traits	Formula
Food acquisition	Oral gape surface	$\frac{Mw\times Md}{Bw\times Bd}$
	Oral gape shape	$\frac{Md}{Mw}$
	Oral gape position	$\frac{Mo}{Hd}$
	Eye size	$\frac{Ed}{Hd}$
Locomotion	Eye position	$\frac{Eh}{Hd}$
	Body transversal shape	$\frac{Bd}{Bw}$
	Pectoral fin position	$\frac{PFi}{PFb}$
	Aspect ratio of the pectoral fin	$\frac{PFl^2}{PFs}$
	Caudal peduncle throttling	$\frac{CFd}{CPd}$
	Aspect ratio of caudal fin	$\frac{CFd^2}{CFs}$
	Fin surface ratio	$\frac{2\times PFs}{CFs}$
	Fin surface to body size ratio	$\frac{(2\times PFs)+CFs}{{}^{\pi}/_{4}\times(Bw\times Bd)}$

2.4. *Statistical analyses*

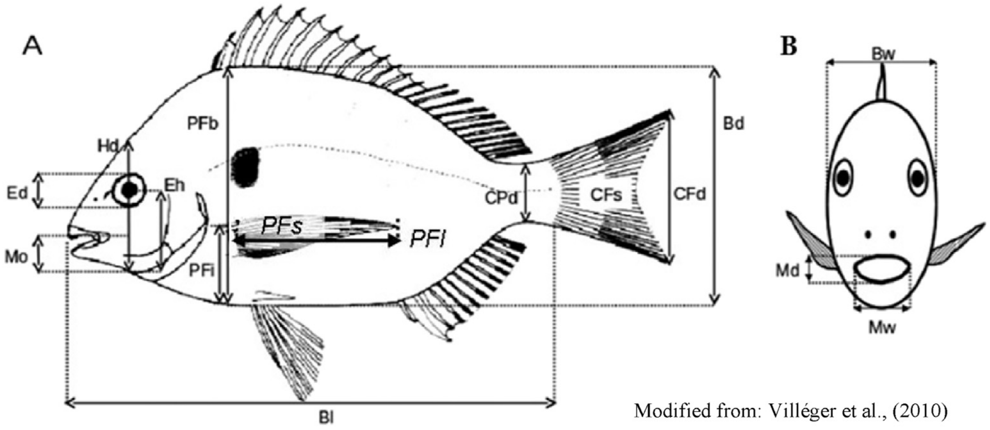
The diversity indices computed were species richness (S) and functional diversity through Rao's quadratic entropy (Q) (Botta-Dukát, 2005; Pavoine, 2012; Rao, 2010) which is the mean functional distance between two randomly chosen individuals in an assemblage. It incorporates both the relative abundance of species and the functional pairwise distances between species:

$$Q=\sum_{J=1}^{S-1}\sum_{j=j+1}^{S}d_{ij}p_ip_j$$

where: p_i is the proportion of all individuals belonging to the species i and j, and d_{ij} is the functional distance between species i and j.

The distributions and level of S and Q between the estuaries were analysed by mean of a Kruskal-Wallis test due to low sampling size and as normality assumptions were not met. When the null hypothesis of homogeneity in distributions between estuaries was rejected, the non-parametric multi-comparison post-hoc test proposed by Siegel and Castellan (1988) assessed the pairwise differences between estuaries.

To provide a functional typology of fish assemblages a multivariate analysis named Double Principal Coordinate Analysis (DPCoA) was used. This method was developed to compare several samples containing species that differ according to their taxonomic, phylogenetic or functional features (Pavoine et al., 2004). Following the description of the method provided by Pavoine et al. (2004, 2013), a key step of this approach is the definition of a common Euclidean space that embeds both species and samples. To obtain this common space, a principal coordinate analysis (PCoA) is first



Modified from: Villéger et al., (2010)

Fig. 2. Morphological measures (A): Bl body standard length, Bd body depth, CPd caudal peduncle minimal depth, CFd caudal fin depth, CFs caudal fin surface, PFi distance between the insertion of the pectoral fin to the bottom of the body, PFb body depth at the level of the pectoral fin insertion, PFl pectoral fin length, PFs pectoral fin surface, Hd head depth along the vertical axis of the eye, Ed eye diameter, Eh distance between the center of the eye to the bottom of the head, Mo distance from the top of the mouth to the bottom of the head along the head depth axis; and with an electronic caliper (B): Bw body width, Md mouth depth, Mw mouth width. Figure modified from Villéger et al. (2010).

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applied to the species distances matrix (Euclidean) according to their functional traits, where each species is weighed by its global proportion over all samples. This PCoA generates a cloud of points in a geometric (Euclidean) space of orthogonal axes (i.e. principal axes), where each point represents a species. Each axis explains a certain part of the total variance (inertia) of the data, in a decreasing order from axis 1. Thus, the first axes of the space optimize the representation of the dissimilarities among species in few dimensions. Then, the samples are positioned in this space at the centroids of the species they contain (centroids are defined in terms of means of species' coordinates on the principal axes). Finally, species that mainly characterized samples by their higher proportions can be deduced according to their position on each axis.

This analysis gives additional information regarding the overall diversity in the study area (i.e. γ-diversity) decomposed in between-assemblages (i.e. β-diversity) and within-assemblages (i.e. α-diversity), where these diversities can directly be quantified by the quadratic entropy Q (Pavoine et al., 2004). A previous application of this method on fish dealt with phylogenetic (Pavoine et al., 2009) and taxonomic distances (Mérigot et al., 2016), and this is the first use on functional diversity of fishes, though it has been applied on other groups (e.g. birds, Pavoine et al., 2004). Differences between the four estuaries in the typology of DPCoA (centroid locations drawn from samples belonging to each estuary) have been assessed by a Kruskal Wallis test on sample coordinates of the axes. For comparison purposes, a Principal Component Analysis (PCA) based only on species abundance was also performed (abundance data were log transformed prior to the analysis due to the relative high values of some species).

All the statistical analyses and diversity indices mentioned above were performed using R version 3.2.1 (R Core Team, 2015). The packages used were “ade4” (Dray and Dufour, 2007), “vegan” (Oksanen et al., 2015) and “FD” (Laliberté et al., 2014).

3. Results

A total of 48,754 individuals were sampled, belonging to 122 species, 80 genera and 36 families. Regarding the capture in biomass, it was 459.8 kg in Itapissuma, followed by Sirinhaém with 177.8 kg, Rio Formoso with 125.8 kg and Suape with 108.5 kg. The 12 morphometric measurements were registered for 663 individuals.

Regarding species richness S, Itapissuma showed the highest values in mean per sample compared to the other estuaries ($Chi^2 = 13.08$, d.f. = 3, $p < 0.05$) (see also quartiles distributions on Fig. 3A). The mean values of S per sample were 29.82 ± 9.62 species in Itapissuma, 18.5 ± 5 species in Suape, 16.2 ± 7.9 species in Sirinhaém and 14.5 ± 4.6 species in Rio Formoso. Interestingly, the quadratic entropy Q showed a quite similar overall pattern to S, but with lower variability within each estuary (Fig. 3B). The mean functional distances between two randomly chosen individuals in a sample in Itapissuma was the highest with $Q = 0.052 \pm 0.02$, whereas lower values were obtained in Sirinhaém (i.e. 0.004 ± 0.002, see also quartiles distributions in Fig. 3B) ($Chi^2 = 18.33$, d.f. = 3, $p < 0.01$). This highlights that, while samples from Itapissuma showed a higher average richness, they encompassed more functional distant individuals than the three other estuaries.

The first factorial plane of the DPCoA analysis, based on both species abundances and functional distances, explained 68.70% of the total inertia (axis 1: 55.0%, axis 2: 13.7%), whereas those of a traditional PCA (based only on species abundances) explained only 17.4%. The analysis revealed a difference in functional typology between Itapissuma assemblages and the other three estuaries on the first axis (see barycentre locations on Fig. 4); supported by a Kruskal-Wallis test performed on the samples coordinates of axis 1, $Chi^2 = 26.04$, d.f. = 3, $p < 0.001$, followed by the non-parametric multiple comparison test proposed by Siegel and Castellan (1988). In contrast, axis 2 differentiated the Rio Formoso assemblages from the three other estuaries ($Chi^2 = 10.65$, d.f. = 3, $p < 0.05$, Fig. 4).

The typology was related to the abundance and function of particular species (Fig. 5). The first axis discriminated Sirinhaém, Suape and Rio Formoso on the positive side, from Itapissuma located at the center of the axis (Fig. 4). The positions of the species related to the positions of estuaries and their habitats (Fig. 5). The list of species and their abundance are given in Supplementary material. *Gobionellus stomatus* (gob.sto) had negative coordinates on the first principal axis, and is therefore related to Itapissuma's habitat due to a relatively higher abundance and being only sampled there (Fig. 5, axis 1). Conversely, *Diapterus auratus* (dia.-aur) had positive coordinates on the first and second axes, and thus could be associated with the Rio Formoso's estuary where it had the highest abundance (Fig. 5, axes 1 and 2). Finally, *Mugil curema* (mug.cur) contributed to the positive part of the axis 1 and the negative part of axis 2. Whereas sampled in Itapissuma, it also contributed to the Sirinhaém and Suape's functional typology where it appeared in most of the samples.

4. Discussion

One means of ensuring accurate representation of the species assemblage is through effective sampling design (Kwak and Peterson, 2007). The heterogeneity of the estuaries makes standardization of sampling strategies and gears difficult. According to Mourãoo et al. (2014), although differences in the fish composition between estuaries may arise due to differential effort (variability in sampling intervals, years and gears), clear trends are detected in biodiversity studies, as for in our analysis. In our study, geomorphology was a key factor in the choice of the fishing gears, which were adapted to most of the fish habitats in the area. The beach seine was used in shallow coastal areas and sand/mud banks whereas the block net was used in all estuaries within flooded mangrove and channels. Thereby, a multi-gear approach was used in order to provide the most complete assessment of fish assemblage (Kwak and Peterson, 2007).

Assessing both species richness and functional diversity can reveal the degree of redundancy and complementarity of species among community that may result from human impacts and environmental conditions (Villéger et al., 2010, 2012). In this study, the estuaries of Sirinhaém, Suape and Rio Formoso had the lowest levels of functional diversity Q and species richness S per sample (Fig. 3). It suggests that assemblages in these areas encompassed predominant functional closely related individuals compared to Itapissuma. This similarity may occur due to some similar features of these estuaries which are geographically close (Paiva and Araújo, 2010) and might result from environmental filtering.

An important structuring factor on the functional composition of assemblages could be the environmental geomorphology. Suape, Sirinhaém and Rio Formoso have a similar structure with a large inner region and a narrow outermost area (Silva et al., 2011), as well as have a high incidence of mangroves (CPRH, 2003c). These characteristics are quite different from those of Itapissuma, which present a very irregular morphology (da Silva et al., 2011), a width ranging from 400 to 1500 m, depths of 2–5 m at low tide and is affected by an important discharge of organic matter from different rivers (CPRH, 2003a). Furthermore, muddy substrate and mangrove vegetation predominate in the Itapissuma estuary (da Silva et al., 2011), enabling the formation of microhabitats that are fundamental for juveniles to settle, complete their life cycle and feed on

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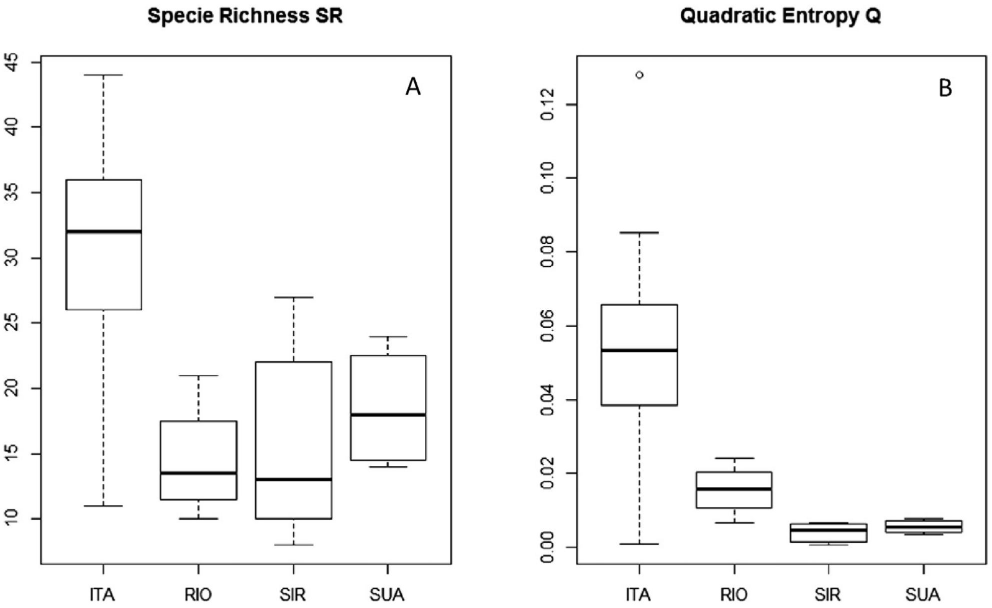

Fig. 3. Boxplots of specie richness *S* and functional diversity *Q* indices, according to the estuary: Itapissuma ITA, Rio Formoso RIO, Sirinhaém SIR, Suape SUA.

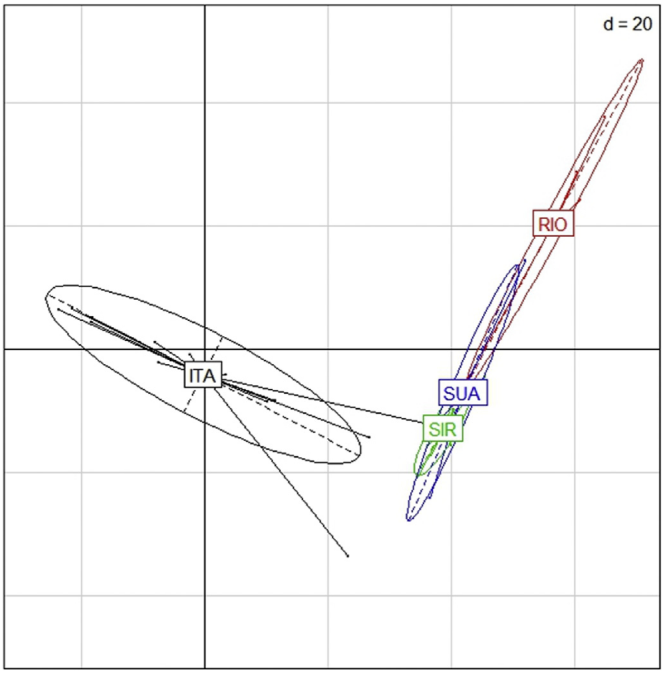

Fig. 4. Projections of samples from the Double Principal Coordinate Analysis (DPCoA), first factorial plane explaining 68.7% of the total inertia (axis 1: 55.0%, axis 2: 13.7%) for the different estuaries: Itapissuma ITA, Sirinhaém SIR, Suape SUA, Rio Formoso RIO. Coloured lines link samples to centroid of the estuary they belong. An ellipse encompasses 67% of samples for a given estuary.

specific prey available (Goodall, 1986). Overall, these characteristics may produce a more diverse habitat and lead ichthyofauna assemblages to be richer with species morphologically distinct from the other three estuaries.

The DPCoA analysis highlighted three species *Gobionellus stomatus, Diapterus auratus* and *Mugil curema* which are closely related to the estuaries Itapissuma, Rio Formoso and Sirinhaém/Suape, respectively. Itapissuma showed higher levels of functional diversity Q and species richness (Fig. 3), and differed in functional

Fig. 5. Projections of species from the Double Principal Coordinate Analysis (DPCoA), first factorial plane explaining 68.7% of the total inertia (axis 1: 55.0%, axis 2: 13.7%). Species' codes are provided in Table 1 in the supplementary material.

typology (Fig. 4). These trends were notably associated to the high level of abundance of *Gobionellus stomatus* (gob.sto) in this estuary when no specimen was sampled elsewhere. The Gobiidae family are directly associated with nutrients-rich areas, i.e. with high availability of food, and in muddy estuaries (Menezes and Figueiredo, 1985), characteristics found in Itapissuma, the largest and most productive estuary of the Pernambuco coast (Lira et al., 2010a). In addition, the Gobiidae family occupies an important place in the trophic chain of coastal environments, due to their role at the initial links in the food chain (Bemvenutti, 1987; Andreata et al., 1997; El-Deir, 2005).

Please cite this article in press as: Silva-Júnior, C.A.B., et al., Functional diversity of fish in tropical estuaries: A traits-based approach of communities in Pernambuco, Brazil, Estuarine, Coastal and Shelf Science (2016), http://dx.doi.org/10.1016/j.ecss.2016.08.030

Diapterus auratus (dia.aur) is an abundant species and contributes to the functional typology of the estuary of Rio Formoso. This species can be classified as a marine, coastal and estuarine fish, living near the bottom where they spawn and feed (Menezes and Figueiredo, 1980; Chaves and Otto, 1998; Nelson, 2006). Moreover, as a marine migrant, it is tolerant to variations in salinity and can be found in estuaries under strong marine influence (Figueiredo and Menezes, 1980; Paiva et al., 2008; Vilar et al., 2011). Rio Formoso is an estuary near the coast and thus is under an important influence of marine habitats nearby (reefs and bays).

Species associated to particular niches by having a specific combination of traits are named specialist species (e.g. Villéger et al., 2010). They are supposed to be more sensitive to environmental changes because an impact or loss of these niches can affect them, while generalist ones may be less sensitive by occupying the most common niches (Jiguet et al., 2007; Villéger et al., 2010). Thus, the difference in functional features observed between assemblages of Itapissuma compared to those of Sirinhaém, Suape and Rio Formoso may also result from past and current habitat/niche degradation, impacts of pollutants from the sugar industry and sewage, as well as from shrimp farming, that occurred for a longer time and higher levels (CPRH, 2003a, 2003b; Lira and Fonseca, 1980). The other areas are also subject to some common impacts (agriculture run-off, deforestation, shrimp farming) and have singular conditions, such as the port development in Suape and presence of protected areas in Rio Formoso (APA Guadalupe (Lira et al., 2010b)) that may affect biodiversity and fish production (Paiva and Araújo, 2010). These factors may determine different scenarios in the future for these areas, as well as new environmental policies, as the recent changes in the Brazilian Forest code and foreseen impacts to mangrove areas (Soares-Filho et al., 2014). Such impacts can influence water transparency, chemical characteristics (i.e. pH), type of organic matter and availability of food sources, which in turn affect estuarine fish communities (Camargo and Isaac, 2003). Furthermore, the sensitivity of specialist species to environmental changes suggests that particular caution of potential human impacts should be considered.

Overall, combining diversity indices and functional typology through a multivariate analysis, the DPCoA (Pavoine et al., 2004, 2013), allowed to deepen diversity assessment in functional patterns of assemblages for estuarine fishes. In estuarine ecosystems, a common approach is to consider fishes by ecological and feeding guilds (Elliott et al., 2007). It provides through functional groups a rough, synthetic and qualitative classifications of species (e.g. marine migrant; piscivorous etc.), that have been applied successfully worldwide (Bremner et al., 2003). While more time consuming, our results suggest that a quantitative traits-based approach can detect different patterns. This approach has allowed to highlight interesting responses of fish communities in a tropical estuary faced to perturbations (Villéger et al., 2010, 2012). In order to assess their complementary, it should thus be accurate in forthcoming studies to directly compare the results of DPCoAs computed from the two kind of information (functional distances between species could be computed on qualitative guilds based the Gower coefficient (Gower, 1971)).

The assessment of functional diversity of fish can be accurate in the context of environmental and anthropogenic perturbations (Villéger et al., 2010; D'agata et al., 2014). Considering that the Brazilian coast encompasses a wide range of contrasted conditions, such analyses could provide an interesting framework for comparison between the estuarine ecosystems along the coast. This will also be accurate at a global scale, for which species richness patterns and its predictors have been recently investigated (Pasquaud et al., 2015; Vasconcelos et al., 2015), and that should be extended to functional diversity.

Acknowledgments

This study was partially funded by INCT (National Institute of Science and Technology, CNPq nº 610013/2011-4) in Tropical Marine Environments - INCT-AmbTropic, and by CAPES (Coordination for the Improvement of Higher Education Personnel) for providing scholarship to the first author. We thank the members of Bioimpact Lab and Coastal Reefs group, which were essential for the proper conduct of this research.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.ecss.2016.08.030.

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Please cite this article in press as: Silva-Júnior, C.A.B., et al., Functional diversity of fish in tropical estuaries: A traits-based approach of communities in Pernambuco, Brazil, Estuarine, Coastal and Shelf Science (2016), http://dx.doi.org/10.1016/j.ecss.2016.08.030

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