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Programa de Pós-Graduação em Diversidade Biológica e Conservação nos Trópicos

FELIPE PEDROSA DE AZEVEDO BARROS

RECIFES ARTIFICIAIS X RECIFES NATURAIS: SIMILARIDADE NA COBERTURA BENTÔNICA EM FUNÇÃO DE UM GRADIENTE DE DISTÂNCIAS EM LARGA ESCALA

MACEIÓ - ALAGOAS
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Dissertação apresentada ao Programa de Pós-Graduação em Diversidade Biológica e Conservação nos Trópicos, Instituto de Ciências Biológicas e da Saúde, Universidade Federal de Alagoas, como requisito para obtenção do título de Mestre em CIÊNCIAS BIOLÓGICAS, área de concentração em Conservação da Biodiversidade Tropical.

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RESUMO

Os recifes artificiais (RAs) são estruturas feitas pelo homem, que tem como objetivo imitar algumas das características dos recifes naturais, sendo assim utilizado como uma ferramenta de conservação e recuperação de ambientes degradados. Este estudo teve como objetivo testar as seguintes hipóteses: (I) comunidades bentônicas de recifes artificiais com mais de 100 anos de introdução são similares aos costões rochosos do entorno e (II) o grau de similaridade é influenciado pela distância entre o costão e o recife artificial. As coletas foram realizadas em duas áreas da cidade de Salvador (Face Protegida e Exposta), nas quais foram amostradas imagens (fotoquadrado) do percentual de cobertura bentônica de 1 naufrágio (RA) e 3 costões rochosos (CRs), distando 0 km, 1 km e 2km do RA. Em todos os locais de coleta foram aferidos a profundidade, a hidrodinâmica e o índice de rugosidade. Os RAs se caracterizaram por apresentar um menor número de componentes e dominância de até 50% de *Turf.* Os CRs foram dominados por *Zoanthos* sp., *Protopalythoa variabilis*, *Echinometra locunter*, *Turf, Ulva lactuca* e algas calcárias. Na face Exposta, a cobertura bentônica do RA apresentou elevada dissimilaridade de todos os CRs, e na face Protegida, a cobertura bentônica no RA foi similar apenas ao local 2 km. Através das análises multivariadas, foi verificado que não existe correlação significativa entre os parâmetros ambientais medidos e a cobertura bentônica. As duas hipóteses investigadas foram rejeitadas e os padrões encontrados parecem estar mais correlacionados com interações biológicas e possíveis impactos humanos.

Palavras-chave: Naufrágios. Costão Rochoso. Invertebrados bentônicos.

ABSTRACT

Artificial reefs (ARs) are man-made structures, which aims to emulate some characteristics of the natural reefs, which have been suggested as a tool for conservation and rehabilitation of the degraded environments. This study aimed to test the following hypotheses: (i) benthic communities from artificial reefs with more than 100 years of introduction are similar to natural rocky reefs and (ii) that these similarities are influenced by the distance among them. Samples were collected in areas subjected to different wave regimes (Exposed and Sheltered), in which, benthic percent coverage were sampled through photoquadrats at 1 shipwreck (AR) and 3 Natural Reefs (NRs), 0 km, 1 km and 2 km far from the RA. In all sampling sites depth, hydrodynamics and the index of rugosity were measured. The ARs were characterized by a lower number of components and high dominance, where Turf represented 50% of the coverage. The NRs were dominated by *Zoanthos* sp., *Protopalythoa variabilis*, *Echinometra locunter,* Turf, *Ulva lactuca* and calcareous algae. In the Exposed area, the AR benthic coverage showed high dissimilarity among all NRs, and to the Sheltered area the AR was similar only to the 2 km site. Through the multivariate analysis, it was found that there is no significant correlation between the measured environmental parameters and benthic coverage. Both hypotheses raised were rejected and patterns seem to be related more to biological interactions and possible to human impacts.

Key-word: Shipwrecks. Rocky reef. Benthic invertebrates.

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1 APRESENTAÇÃO

Esta dissertação é resultado de um estudo pioneiro no brasil que buscou compreender as interações e similaridades entre recifes artificiais e costões rochosos naturais, em função de um gradiente de distâncias em escala de quilômetros. Esta dissertação está composta por duas seções. A primeira é referente à revisão de literatura sobre tópicos mais relevantes e seus desdobramentos para a compreensão do trabalho subsequente. A segunda seção foi redigida em formato de artigo científico, submetido ao periódico **Journal of Sea Research**, sob o título: *Artificial vs. natural reefs: benthic cover similarities related to a large-scale distance gradient*.

2 REVISÃO DA LITERATURA

2.1 Bioincrustação

A bioincrustação marinha é definida como um processo natural resultante da colonização ou do crescimento de micro-organismos, tais como bactérias e micro-algas, e de macro-organismos, como macroalgas e macroinvertebrados sésseis, sobre uma superfície artificial submersa introduzida pelo homem (ex: plataformas, pilares, cascos de navios, boias) (Seaman, 2000; Svane & Petersen, 2001). De forma geral, a bioincrustação consiste em uma grande concentração de biomassa em substratos consolidados submersos (Railkin, 2004).

O desenvolvimento das comunidades incrustantes se inicia com a adsorção de macromoléculas orgânicas no momento em que um objeto é submerso, sendo logo em seguida colonizado por bactérias (Wahl, 1989; Bode et al., 2006). A partir do segundo dia de imersão em diante, tipicamente, diatomáceas e protozoários fixam-se, formando um biofilme rico em recursos alimentares, que após uma semana servirão como atrativo para a colonização de larvas e esporos de algas (Whal, 1989; Glasby & Connell, 2001). Durante a etapa inicial de colonização, os micro-organismos (ou micro-incrustação) dominam o substrato consolidado. Posteriormente, uma comunidade de organismos macroscópicos incrustantes (ou macro-incrustação) desenvolve-se sobre a micro-incrustação (Dobretsov, Abed, & Voolstra, 2013; Sahu et al., 2011). Ao final desse processo, essa comunidade bioincrustante é composta principalmente por cracas, briozoários, ascídias, esponjas e macroalgas (Jackson & Miller, 2009).

O estabelecimento dos organismos bentônicos é influenciado por diversos fatores, tais como os fatores químicos (pH e salinidade), físicos (pressão e hidrodinamismo) e biológicos, sendo os biológicos representados relações com outros organismos que competem por espaço e alimento, por exemplo (Wahl, 1989). Além destes, o tipo do substrato (material), a rugosidade, a complexidade estrutural e orientação (vertical ou horizontal), também são fatores capazes de alterar a estrutura das comunidades bentônicas (Wahl, 1989; Guichard & Bourget, 1998; Seaman, 2000; Sherman, 2002; Oigman-pszczol & Figueiredo, 2004; Perkol-Finkel et al., 2006). Segundo Anderson & Underwood (2006), tais fatores podem determinar modificações nos padrões de colonização e, consequentemente, na sucessão e estabelecimento dessas comunidades.

De maneira geral, entende-se que quanto mais complexo, dinâmico e heterogêneo um substrato for, mais abundante, rico e diverso será a sua comunidade biológica, seja ela bentônica ou pelágica (Guichard & Bourget, 1998; Gratwicke & Speight, 2005; Wilson et al., 2006; Walker et al., 2007; Hunter & Sayer, 2009; Van Gaever et al., 2010).

A zona costeira e a plataforma continental fornecem grande quantidade de substratos consolidados naturais e artificiais para a fauna incrustante (Coutinho, 2002). Cerca de 98% das espécies de animais marinhos vivem no ambiente bentônico, sendo que pelo menos 127.000 destes vivem em substratos consolidados e somente 30.000 em sedimentos inconsolidados ou móveis (Da Gama et al., 2009).

Por substrato consolidado disponível para colonização, entende-se não somente estruturas físicas como rochas e recifes, mas também, outros organismos presentes na coluna d'água, planctônicos ou nectônicos, ou o próprio bentos que também podem atuar como substrato de colonização e fixação para comunidades bentônicas incrustantes. Desta forma, as comunidades incrustantes são altamente diversificadas, uma vez que podem ser formadas por uma ou mais camadas ou estratos compostos por diversos organismos (Da Gama et al., 2009).

2.2 Substratos consolidados

Dentre os ecossistemas marinhos costeiros bentônicos, os recifes de corais são considerados como um dos mais importantes do planeta por apresentarem alta riqueza de espécies de importância ecológica e econômica, grande concentração de biomassa e alta produtividade biológica (Wilson et al., 2006; Riegl et al., 2009; Graham et al., 2014). Segundo Jacobi & Langevin (1996), tais características fazem com que estes ambientes sejam amplamente estudados a fim de resolver questões ecológicas básicas, como competição, predação e sucessão. Em geral, esses estudos buscam encontrar padrões que sirvam como modelo para as demais comunidades.

Semelhante às características ecológicas dos recifes de corais, os costões rochosos são substrato consolidado, formados por rochas (sedimentares ou magmáticas, por exemplo), sendo considerados muito relevantes nos locais onde ocorrem, devido a sua alta riqueza de espécies e grande biomassa, servindo como locais de alimentação, crescimento e reprodução de um grande número de espécies (Coutinho & Zalmon, 2009).

No mundo inteiro, os costões rochosos são tipicamente caracterizados por um forte zonação espacial. A disposição dos organismos é definida em faixas dispostas horizontalmente no costão, onde cada espécie é mais abundante dentro de uma determinada zona, a qual proporciona condições favoráveis à sua sobrevivência (Coutinho & Zalmon, 2009; Moreno & Rocha, 2012). As espécies que ocorrem em cada zona podem variar em função da latitude, dos níveis de maré e dos fatores abióticos.

No Brasil, os costões rochosos verdadeiros estão presentes na costa brasileira entre o Rio Grande do Sul até Pernambuco, mas quase que exclusivamente nas regiões Sudeste e Sul. Apenas a região Sudeste foi amplamente estudada (especialmente o litoral do Rio de Janeiro e de São Paulo), no entanto a maioria dos trabalhos é de caráter descritivo procurando caracterizar as comunidades e sua distribuição espacial (Moreno & Rocha, 2012).

Alguns estudos descritivos também focaram problemas, como a importância das características do ambiente no processo de recolonização (Tortolero-Langarica et al., 2014), bem como estudos mais aplicados comparando a sucessão de comunidades bentônicas da região entremarés em duas áreas com diferentes graus de poluição (Breves-Ramos et al., 2005).

A grande diversidade de organismos e o fácil acesso a esses ecossistemas, tornaram os costões rochosos e os recifes de corais áreas altamente vulneráveis aos impactos antrópicos, tais como o turismo (pisoteio e mergulho recreativo) e a exploração desordenada, pesca, poluição (Pastorok & Bilyard, 1985; Wilkinson, 2000; Meenakumari et al., 2008). Estudos mostram que a comunidade bentônica é altamente sensível a alterações no ambiente, havendo perda da diversidade e abundância nos locais sujeitos a estes diferentes impactos (Hong, 1983; Fierro et al., 2015).

Devido a isso, diferentes tipos de estudos vêm sendo desenvolvidos no sentido de minimizar estes problemas (Grieve et al., 2014; Del-Pilar-Ruso et al., 2015). Dentre estes, está a utilização de substratos artificiais como ferramenta de compensação e recuperação dos ambientes naturais degradados (Culter, 1997; Clark & Edwards, 1999; Mousavi et al., 2015).

2.3 Recifes artificiais

A uso de qualquer tipo de substrato artificial, seja ele de material natural (como rochas e madeira) ou de origem humana (estruturas de ferro, concreto ou borracha), e que apresente a capacidade de imitar algumas das características físicas, biológicas e/ou socioeconômicas de um recife natural (RN), é definido como recife artificial (RA) (Seaman, 2000; Baine, 2001; Boaventura et al., 2006).

De maneira geral um RA deve ser desenhado, construído e introduzido no meio aquático, de forma planejada (Broughton, 2012). Entretanto, grandes estruturas afundadas de forma acidental, que não foram desenhadas com esta finalidade, podem também ser classificadas como RAs secundários (ex: embarcações, plataformas de petróleo e aviões) (Seaman, 2000; Broughton, 2012).

Ao longo das últimas décadas, os RAs têm sido utilizados com diversas finalidades econômicas e ecológicas, como a criação de novas áreas recreativas para o mergulho (e, consequentemente, diminuição da pressão nos RNs), exclusão da pesca em algumas áreas (principalmente pesca de arrasto), controle da erosão da praia, conservação da biodiversidade, e para testar teorias ecológicas (Seaman e Jensen, 2000; Baine, 2001; Spieler et al., 2001; Boaventura et al., 2006). No entanto, na maioria dos casos, a introdução de RAs em ambientes marinhos e estuarinos tem como objetivo principal incrementar os recursos pesqueiros (Seaman, 2000).

Segundo Love & Schroeder (2004), um RA, dependendo do seu tamanho, pode aumentar em até 3 vezes a biomassa local de peixes. Devido a esse incremento, os RAs são amplamente utilizados em mais de 30 países, principalmente como ferramenta de recuperação de estoques pesqueiros em declínio (Jensen, 2002; Seaman, 2002).

Apesar dos registros históricos datarem de centenas de anos desde o início do uso dos “RAs” como ferramenta para o incremento da produtividade pesqueira (Christian et al., 1998), foi apenas a partir da década de 70 que houve um grande avanço na área para a construção e pesquisa de grandes RAs por parte do governo Japonês, com investimentos superiores a US$ 200 milhões (Simard, 1996). Posteriormente, países da costa oeste Europeia também começaram a utilizar RAs com finalidade de incrementar a pesca e gerenciar conflitos costeiros (Alencar et al., 2003).

Após os primeiros resultados do Japão quanto aos benefícios do uso de RAs, os EUA começaram a desenvolver grandes projetos e, na década de 90, já existiam mais de 2.000 sistemas de RAs ativos (Nesse, 2002). Estudos realizados próximo aos parques de exploração de petróleo dos EUA mostraram que os estoques pesqueiros de grandes peixes pelágicos aumentaram em média 5 vezes após a implantação da plataforma, e nas áreas onde foram removidas, os estoques declinaram (Schroeder & Love, 2004). Até a última década, existiam mais de 6.000 plataformas ativas pelo mundo, e outras centenas desativadas estão funcionando como RAs secundários (Schroeder & Love, 2004).

No Brasil, os estudos com RAs tiveram início a partir da década de 90 (Alencar et al., 2003), principalmente com pesquisas voltadas para os processos de bioincrustação (Silva, 2001), produtividade biológica (Brandini & Silva, 2000), formas e materiais (Silva et al., 2003; Zalmon & Gomes, 2003). Entre os principais estudos, Krohling et al. (2006) descreve o papel funcional da macrofauna na bioincrustação e a sua relação com as comunidades icticas; Krohling et al. (2008) encontrou que mesmo sobre a alta influência da sedimentação e da turbidez (causadas pela precipitação pluviométrica e a vazão de rios) muitas espécies bioincrustantes conseguem recrutar indivíduos jovens; Fagundes Netto & Zalmon (2011) mostram que tanto os recifes artificiais quanto os recifes naturais exercem efeitos tanto de atração quanto de produção sobre as populações de peixes, mas isso depende de cada espécie e de como os recifes serão manejados; Zalmon et al. (2011) descreve que um recife artifical pode alterar a comunidade infaunal ao seu redor, mas que essa alteração possui pouco alcance, sendo rapidamente dissipada pela ação das correntes no fundo marinho.

Apesar da comprovada eficiência dos RAs em relação ao aumento das taxas de captura de peixes e da biomassa da comunidade bentônica, alguns autores consideram que essas estruturas também podem causar efeitos ecológicos e econômicos negativos (Seaman, 2000; Seaman & Jensen, 2000). Atualmente o foco maior sobre essas questões é a discussão sobre a capacidade dos RAs em atuar como “produtores” ou “atratores” de peixes. Tais discussões levam a um impasse, uma vez que, dependendo da espécie alvo avaliada, o RA pode atuar como “produtor” (ex: peixes bentônicos e organismos sésseis), ou como atratores de espécies comerciais de grande porte, devido a disponibilidade de novas áreas de alimentação e abrigo (Schroeder & Love, 2004). Por exemplo, Pickering & Whitmarsh (1997) verificaram que algumas espécies de peixes e lagostas, são atraídas para RAs logo após a sua instalação, dando a impressão de aumento no estoque pesqueiro, mas após alguns meses os estoques artificiais declinam e as espécies tendem a voltar para o recife natural.

Outros impactos negativos associados aos RAs, devido a menor importância econômica, são em geral menos estudados, como é o caso da mudança na direção das correntes (Boaventura et al., 2006), do hidrodinamismo local (Guichard & Bourget, 1998), alteração da granulometria (sedimentação) (Danovaro, 2002) e da fauna local (Zalmon et al., 2012), e o “roubo de larvas” oriundas de RNs, (uma vez que essas larvas se fixam em um RA, a disponibilidade destas para o ambiente natural pode diminuir) (Stephens, 2002).

Sendo assim, é preciso desenvolver estudos que tenham como objetivo estudar a real capacidade que um RA tem em “imitar” um RN, consequentemente, é preciso compreender a fundo as interações entre as comunidades dos RAs e dos RNs (Carr & Hixon, 1994; Perkol-Finkel et al., 2006;Thanner et al., 2006; Perkol-Finkel & Benayahu, 2007; Perkol-Finkel & Benayahu, 2009;).

Estudos comparativos entre a comunidade bentônica de RAs e RNs vêm investigando diversos fatores que possam influenciar as características de ambas as comunidades. Por exemplo, Hunter & Sayer (2009), verificou em seu estudo que heterogeneidade e a orientação do substrato podem alterar a diversidade e a abundância da comunidade epibêntica. Perkol-Finkel et al. (2005) investigou a influência do tempo (em larga escala) na estrutura da comunidade bentônica, já outros investigaram os estágios iniciais de desenvolvimento dos RAs (Stephens, 2002; Atilla et al., 2003; Walker et al., 2007).

Os resultados são conflitantes, enquanto alguns autores encontram grande similaridade entre RAs e RNs, sob certas condições (Aseltine-Neilson, Bernstein, Palmer-Zwahlen, Riege, & Smith, 1999; Perkol-Finkel et al., 2006; Shimrit Perkol-Finkel & Benayahu, 2009; Thanner et al., 2006), outros, de forma oposta, encontram baixo grau de similaridade entre as comunidades desses recifes (Burt et al., 2009; Coelho et al., 2012).

Outro fator que pode ser considerado como fundamental para compreender o quanto um RA pode interferir no ambiente natural bentônico, é a distância entre estes. Dos poucos trabalhos realizados que relatam a influência de um RA no ambiente natural bentônico, a maioria está relacionada à comunidade de substrato inconsolidado. Em geral os resultados destes trabalhos são divergentes, uma vez que alguns autores encontram alterações no ambiente em escalas de dezenas de metros (Wilding & Sayer, 2002; Wilding, 2006), outros afirmam que não existem impactos (Barros et al., 2001, 2004). Segundo Zalmon et al. (2011), os RAs são capazes de alterar a estrutura da comunidade bentônica de seu entorno, mas essas alterações são de baixa magnitude sendo dissipadas após poucos metros de distância ou são sobrepostos pela influência das correntes ou outros fatores ambientais (ex: sazonalidade ou aumento de matéria orgânica oriunda de rios).

Segundo Perkol-finkel (2006), quando um RA é colocado próximo a um recife natural adjacente, onde ambos possuem características estruturais semelhantes, as estruturas das comunidades bentônicas se tornarão muito semelhantes após algumas décadas. No entanto, quando a complexidade de dois ou mais substratos diferem entre si, independente se artificial ou natural, as comunidades diferirão mesmo depois de mais de um século (Perkol-Finkel et al., 2006; Walker et al., 2007; Hunter & Sayer, 2009).

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3 ARTIFICIAL *VS.* NATURAL REEFS: BENTHIC COVER SIMILARITIES RELATED TO A LARGE-SCALE DISTANCE GRADIENT

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ABSTRACT

Artificial reefs (ARs) are man-made structures, which aims to emulate some characteristics of the natural reefs, which have been suggested as a tool for conservation and rehabilitation of the degraded environments. This study aimed to test the following hypotheses: (i) benthic communities from artificial reefs with more than 100 years of introduction are similar to natural rocky reefs and (ii) that these similarities are influenced by the distance among them. Samples were collected in areas subjected to different wave regimes (Exposed and Sheltered), in which, benthic percent coverage were sampled through photoquadrats at 1 shipwreck (RA) and 3 Natural Reefs (NRs), 0 km, 1 km and 2 km far from the RA. In all sampling sites depth, hydrodynamics and the index of rugosity were measured. The ARs were characterized by a lower number of components and high dominance, where Turf represented 50% of the coverage. The NRs were dominated by *Zoanthos* sp., *Protopalythoa variabilis*, *Echinometra locunter, Turf*, *Ulva lactuca* and calcareous algae. In the Exposed area, the AR benthic coverage showed high dissimilarity among all NRs, and to the Sheltered area, the AR was similar only to the 2 km site. Both hypotheses raised were rejected and patterns seem to be related more to biological interactions and human impacts.

Key-word: Shipwreck. Rocky reef. Benthic invertebrates.

3.1 Introduction

Different factors, such as: depth, hydrodynamics, habitat complexity and substrate orientation, and also ecological relationships such as: competition, predation/herbivory and recruitment, influence benthic communities strucuture in hard bottom environments (Baynes & Szmant, 1989; Wahl, 1989; Svane & Petersen, 2001; Bullard et al., 2004; Oigman-pszczol & Figueiredo, 2004; Boaventura et al., 2006). The benthic communities on these natural environments, such as rocky reefs, provide food resources to secondary consumers and habitat for other benthic invertebrates leading to an increased habitat complexity and hence providing shelter for many other organisms, including fish (Burt et al., 2009). Due to these features natural reefs are considered one of the most productive environments in marine ecosystems (Hackradt et al., 2011).

An artificial reef can function in a very similar way to these natural environments, showing similar features, and hence, leading to such productivity, notably in areas where natural environments are degraded (Seaman, 2000). Artificial reefs are man-made structures placed intentionally or accidentally on a substrate, which are able to mimic some of the features of natural reefs (Seaman, 2000; Svane & Petersen, 2001). These structures can be projected for this purpose, and build in different shapes and materials (Lukens 1997) and also can be introduced in aquatic environments intentionally but for other purposes, such as oil and gas platforms, shipwrecks and airplanes (Seaman, 2000). However, the most common example of the accidental introduction of a structure into the aquatic environment, are shipwrecks (Perkol-Finkel et al., 2005).

When submerged, an artificial reef is colonized by different organisms, mainly by the larvae and spores from the dispersion of the nearest natural reefs (Fitzhardinge, 1989; Baynes & Szmant, 1989; Svane & Petersen, 2001). After the establishment of this rich and diverse benthic fauna, the artificial reefs not only receive the larvae but also start to function as exporters of larvae to the natural environment (Bohnsack & Sutherland, 1985). At the end of this process these artificial reefs communities reshape the local marine ecosystem with the provision of new habitats, by increasing the production of biomass (*e.g.*: benthic community) (Seaman & Sprague, 1991).

Due to these features, such structures have been suggested as a tool in the recovery of the diversity and local biomass of the degraded natural reefs (Bohnsack & Sutherland, 1985; Baine, 2001; Seaman, 2002; Walker & Schlacher, 2014). A several studies have investigated the design and arrangement of materials used in construction, orientation (vertical or horizontal) and appropriate locations for the placement, in order to improve the efficiency of artificial reefs (Bohnsack & Sutherland, 1985; Culter, 1997; Lukens, 1997; Baine, 2001).

Despite the environmental and socioeconomic benefits attributed to artificial reefs (e.g.: biomass increment and reduced both the fishing pressure and the tourism on natural reefs) (Seaman, 2000; Seaman, 2002; Leeworthy et al., 2006), the benefits of these structures are questioned by some others authors that discuss the negative effects after the introduction of these artificial reefs, for example, "stealing” of larvae and fish from the natural environments (Stephens, 2002), changes in sediments’ grain size and currents direction and hence, changes in abundance and composition of the fauna (Ambrose & Anderson, 1990; Boaventura et al., 2006; Zalmon et al., 2012; Zalmon et al., 2014). According to Wilding & Sayer (2002), the spatial range of environmental changes caused by an artificial reef can be relatively small (a few meters) or can extend for several hundred meters (Ambrose & Anderson, 1990; Davis, VanBlaricom, & Dayton, 1982).

Regardless interactions between artificial and natural reefs to be positive or negative, it is a known fact that they exist and although several studies had already studied these, the majority has investigated the interactions in fish communities (Santos et al., 2010; Fagundes-Netto et al., 2011; Simon et al., 2011; Simon et al., 2013) or the influence of artificial reefs in the fauna of the surrounding soft sediments (Ambrose & Anderson, 1990; Barros et al., 2001; Danovaro & Fraschetti, 2002; Zalmon et al., 2012).

Despite there is some decades since the studies related to artificial reef started, growing awareness for the importance of understanding the interactions between the benthic communities of artificial and natural structures arise just after the beginning of the XXI century (Walker & Schlacher, 2014). Some of these studies, such as the Perkol-Finkel et al. (2005) and Perkol-Finkel et al. (2006) make clear in their findings that younger artificial reefs (up to 20 years) tend to have a different community from natural reefs, but over time, these communities tend to be similar in terms of diversity and abundance. Also, according to these authors, questions about the interactions between artificial reefs already well established (over 60 years) and natural reefs need to be answered. Artificial structures with a large time of colonization are hard to find and it represents an obstacle to carried out this kind of study (Perkol-Finkel et al., 2006).

Few works have evaluated the influence of an artificial reef on natural surrounding environments, such as the study of Barros et al. (2001) and Zalmon et al. (2014), which evaluated the impact of an artificial reef on infaunal communities of the surrounding soft sediments. Although numerous studies have already addressed the ecology of rocky reefs (Thompson et al., 2002; Barros et al., 2012; Moreno & Rocha, 2012), few have investigated the influence of artificial reefs in these environments, such as the Simon et al. (2011 and 2013), who studied changes in fish communities among shipwrecks and rocky reefs.

In this context, this study aimed to test the following hypotheses:

(i) benthic communities from artificial reefs with more than 100 years of introduction are similar to natural rocky reefs and (ii) that these similarities are influenced by the distance among them.

3.2 Materials and methods

3.2.1 Study Site

This study was carried out in the city of Salvador, Northeast of Brazil. This city is situated on a small peninsula, separated by the *Todos os Santos* Bay and the open waters of the Atlantic Ocean. The bay (centered among latitude 12° 50' S and longitude 038° 38' W) is the second largest bay in the country and has the greatest coral reef biodiversity of the South Atlantic (Cruz et al. 2015) (Figure 1). The climate is tropical and humid, and a mean value of 27 °C for water temperature.

The rocky reefs extend along the west coast of the Salvador, in the inner area of the bay, providing habitat to the establishment of several benthic species (Hatje et al., 2009). This area of the bay is sheltered of the direct wave action, however, its entrance is choked allowing a strong water circulation and a tidal range inside the bay, 30% higher on average than the oceanic tide.

Outside of the bay, in the east coast of Salvador, reefs are exposed to wave action (Figure 1). Historically, this region is also known by the presence of numerous ancient shipwrecks, dating from the XIX and XX centuries, and theirs rich associated fauna (Cruz et al., 2009).

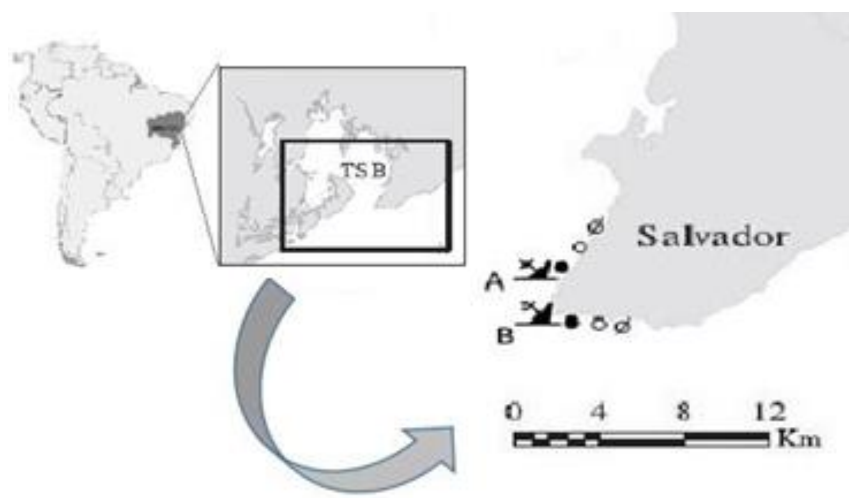

Figure 1 - Schematic drawing of the study area showing sampling sites (A = Artificial Reef Maraldi, B = Artificial Reef Cap Frio, rocky reefs ● = 0 km, ○ = 1 km and ϕ = 2 km far from the Artificial Reef. TSB = *Todos os Santos* Bay (Modified from Cruz et al., 2015).

This study was carried out on natural rocky reefs (NRs) and on two shipwrecks, Maraldi and Cap Frio, considered here as artificial reefs (ARs), in both the Exposed and the Sheltered areas of the *Todos os Santos* Bay.

The AR Maraldi is an iron vessel, sunk in 1875 (140 years ago) and is located in an average depth of 6 m at the Sheltered area of the bay (13˚ 00 ' 54 "South, 038˚ 32 '08" West). The AR Cap Frio is a steel freighter, sunk in 1907 (106 years ago), located at the Exposed area of the bay (13˚ 00' 46" S, 038˚ 32' 05" W), in an average depth of 13 m (Carvalho 2015). Both RAs are distant about 170 m from the nearest NRs.

3.2.2 Sampling Strategy

3.2.2.1 Benthic Coverage

Data sampling was carried out in january/2015 in eight locations, as follows: one shipwreck and three NRs in the Exposed area and one shipwreck and three NRs in the Sheltered area. In both cases, the 3 NRs were chosen at a gradient of distances from the shipwreck, as follows: a 170 m (represented by 0 km), 1 km and 2 km far from the shipwrecks (Figure 1).

To characterize the benthic coverage, we used the non-destructive method of photoquadrats (50 x 50 cm), through SCUBA and free diving. At each site 12 transects of 5 m long were randomly placed on the bottom, where three equidistant images were recorded in each transect, resulting in 288 samples. All transects were arranged parallel to the beach line and as horizontal as possible in order to keep the same depth for all samples at the same transect.

3.2.2.2 Environmental Parameters

In all sampling sites environmental factors such as: depth, index of rugosity, visibility and hydrodynamic, were measured. To evaluate the intensity of the hydrodynamics at each site, a scale of wave exposition proposed by Krajewski & Floeter (2011) was used in which water dynamic is classified on an arbitrary scale from 1, the smallest wave action, to 9 the highest. This classification was performed by three experienced divers. According to Nunes et al. (2013), the methodology proposed by Krajewski and Floeter (2011) reflects the same results as more complex methodologies such as the plaster balls (Jokiel & Morrissey, 1993). We also used results of Nunes et al (2013), who presented values of this scale for the same NRs of this study.

The Rugosity was measured at each sample, prior to photoquadrat, using the link-chain method (Luckhurst & Luckhurst, 1978) as a rugosity index.

3.2.3 Processing Samples

The benthic percent coverage of each sample was measured classifying different components on the images (the term component here refers to any item registered in photoquadrats) in biological and non-biological categories. The biological categories include: a) invertebrates' taxa, which were identified to the lowest taxonomic level as possible and b) algae, which were identified to species level. When it was not possible to identify the algae, it was grouped into: *Turf,* Macroalgae, Articulated Calcareous Algae and Non Articulated Calcareous Algae.

The non-biological category is composed by the components: "Sedimentation", unconsolidated sediments deposited on hard substrata, "Hole", without any benthic coverage, "Rock", represented by sedimentary rocks and "Sedimentation+*Turf* ", consisting of unconsolidated sediments deposited on the *Turf* algae complex (here considered as non-biological category because of the prevailance of sediments).

The images were analyzed by PhotoQuad program (Trygonis & Sini, 2012) to quantify the percentage of coverage of the different components, using the methods of 'freehand regions' and 'multi-scale image segmentation regions', where the percentage of coverage is calculated from the outline of the design of each component present in the image. The efficiency of these methods was tested by the authors, who observed more precise results than the traditional method of points (Trygonis & Sini, 2012).

3.2.4 Statistical Analyses

To verify the degree of similarity in benthic coverage between ARs and NRs at different distances, at both Protected and Exposed areas univariate and multivariate analyzes were applied.

Univariate indices such as richness of taxa (S) (based only on data from the invertebrate taxa) and the components percent coverage (%) were calculated. Only living individuals were quantified and classified (Benayahu & Perkol-Finkel, 2004).

Multivariate analyzes such as non-Metric Multidimensional Scaling (nMDS) (Clarke & Warwick, 1994) and Analyzes of Similarity (ANOSIM one-way) (Anderson, 2001) were applied to benthic percent coverage registered for artificial reefs and natural rocky reefs at both areas, using distance as a predictive factor. These analyzes were also applied to the data of the two areas together.

The Percentage of Similarity (SIMPER) analysis (Clarke, 1993) was applied to that same data in order to identify the components that most contributed to the patterns evidenced by the nMDS.

These analyzes were applied to a similarity matrix, constructed using the Bray-Curtis similarity index without transformation.

In order to verify the influence of the environmental gradients on the benthic coverage the BIO-ENV analysis (Clarke & Ainsworth, 1993), using Spearman correlation index (□), was applied to the distance matrices. For biological matrix the Bray-Curtis similarity index was used and for environmental matrix Euclidean Distance was applied.

A Canonical Correspondence Analysis - CCA (Braak & Verdonschot, 1995) was applied to these same data in order to investigate relationships of the spatial patterns of the samples of ARs and NRs with environmental gradients. The permutation test of Monte Carlo was applied to test the significance of the correlations pointed by the CCA. (p <0.05).

All the analyzes was applied through the statistical packages Primer 6.0, Statistica 12 and Past 3.0.

3.3 Results

3.3.1 Environmental Parameters

The highest values of depth were recorded to the ARs, 8 and 4 m, at the Exposed and Sheltered areas, respectively. The depth of the NRs ranged from 1 to 2.5 m on both areas (Figure 2).

The hydrodynamics of the ARs were classified with values of 8 and 4, at the Exposed and Sheltered areas, respectively. Among the NRs, the 0 km received the highest value to this classification on the Exposed area. In the Sheltered area the highest value achieved was 3 for 1 km (Figure 3).

The mean values for the index of rugosity ranged from 1.6 to 2.8, at the AR of the Exposed area and 1km NR at the Sheltered area, respectively (Figure 3). The visibility of the areas ranged from 5 to 9 m on both areas, the highest values of visibility were recorded to the 0 km (9 m) Exposed area and AR (8,5 m) and 1 km (8 m) Sheltered area (Figure 2).

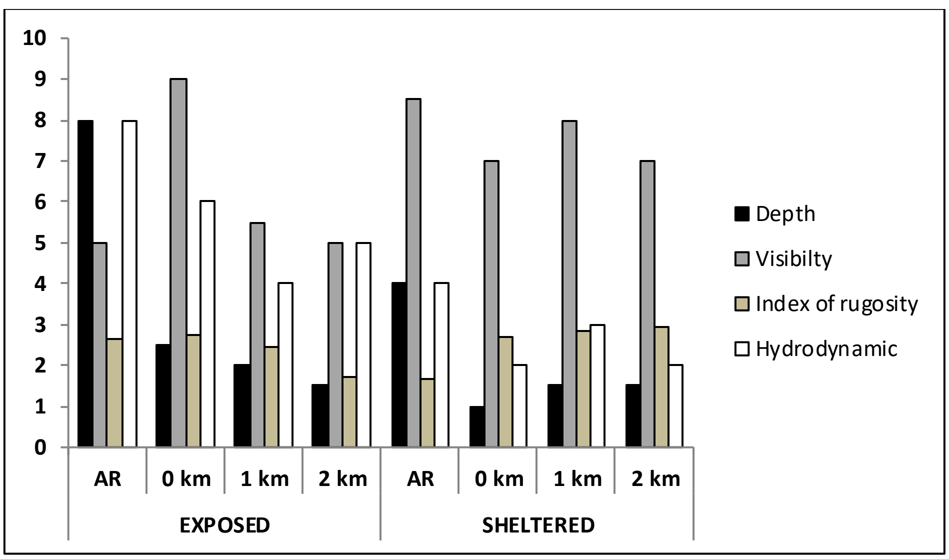

Figure 2 - Values of depth (m), hydrodynamics and index of rugosity to the Artificial Reefs and Natural Reefs at 0, 1 and 2 km distances in the Exposed and Sheltered areas. AR = Artificial Reef (Erro bars represent standard deviation).

3.3.2 *Benthic coverage*

At all sampling sites, the 4 components of non-biological category were registered (Stone, Sedimentation, Sedimentation+*Turf,* and Hole). Among the biological categories were recorded: 1 species of algae, *Ulva lactuca*, and 4 algae groups (*Turf*, Articulated coralline algae, Calcareous algae, and Macroalgae) and

61 taxas of invertebrates, where 30 belong to the phylum Porifera, 16 to class Ascidiacea, 7 to the phylum Cnidaria, 4 to the phylum Echinodermata, 1 to the phylum Bryozoa, 1 to subclass Crustacea, 1 to the phylum Mollusca and 1 to class Polychaeta.

Regarding the percent coverage of these components was observed that 55.4% of the samples corresponded to algae, 37.2% to invertebrate taxa and 7.4% to the components of the non-biological category.

3.3.3 Exposed Area

A total of 46 invertebrate taxa were identified, which 20 were from the phylum Porifera, 15 Ascidiacea, 5 Cnidaria, 3 Echinodermata, 1 Crustacea, 1 Mollusca and 1 Bryozoa. All of the 4 groups of algae and one species, *U. lactuca* were recorded. All components of non-biological category were recorded.

To the AR, 23 taxa of invertebrates (mean 5.11 ± 2.17) were recorded, which 5 were exclusive, 24 taxa in 0km (average of 7.23 ± 1.77), with 12 exclusive taxa, 14 taxa within 1 km (average of 5.21 ± 1.11), with 1 exclusive, and 10 taxa in 2 km (average of 4.93 ± 1.18), also with 1 exclusive. For algae, all groups were registered at all sites, with the exception of the articulated coralline algae that were not recorded at the AR.

The most abundant component at this area was *Turf* with 28%, followed by *Zoanthus* sp. (15%), Calcareous algae (11%), *Ulva lactuca* (7%), *Protopalythoa variabilis* (7%), *Echinometra locunter* (7%), *Eudistoma saldanhai* (5%), Macroalgae (4%) Sedimentation+*Turf* (3%) and *Palythoa caribaeorum* (3%).

Zoanthus sp., *U. lactuca* and *E. locunter*, were not registered at AR, which were dominated by *Turf* and *E*. *saldanhai* (22.7% and 49.01%, respectively). The former was registered exclusively at the AR.

At the 0 km site the dominant components were *Zoanthus* sp. (27.5%), Calcareous algae (23.7%), *Turf* (15.5%) and *E. locunter* (15.1%). At the 1 km the dominant were *Turf* (27.3%), *P. variabilis* (19.9%) and *Zoanthus* sp. (13.8%). In 2 km the highest values of percent coverage were registered for *U. lactuca* (28%), *Turf* (20%), *Zoanthus* sp. (19.9%) and Calcareous algae (14.4%) (Figure 3).

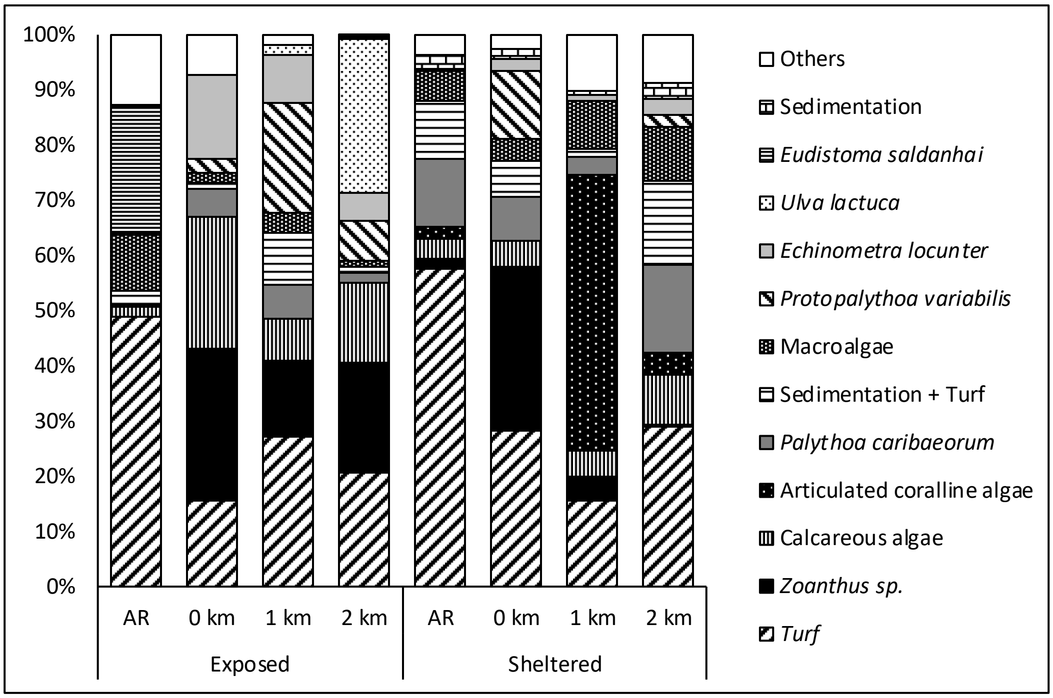

Figure 3 – Relative abundance (%) of the the cover components (> 5%) recorded at the Artificial Reefs and NRs at 0, 1 and 2 km distances in Exposedand Sheltered areas. AR = Artificial Reef.

The nMDS analysis applied to the percent coverage data of each sample of each site, shows a trend toward separation among the AR and NRs samples, regardless of distances (Figure 4A,B).

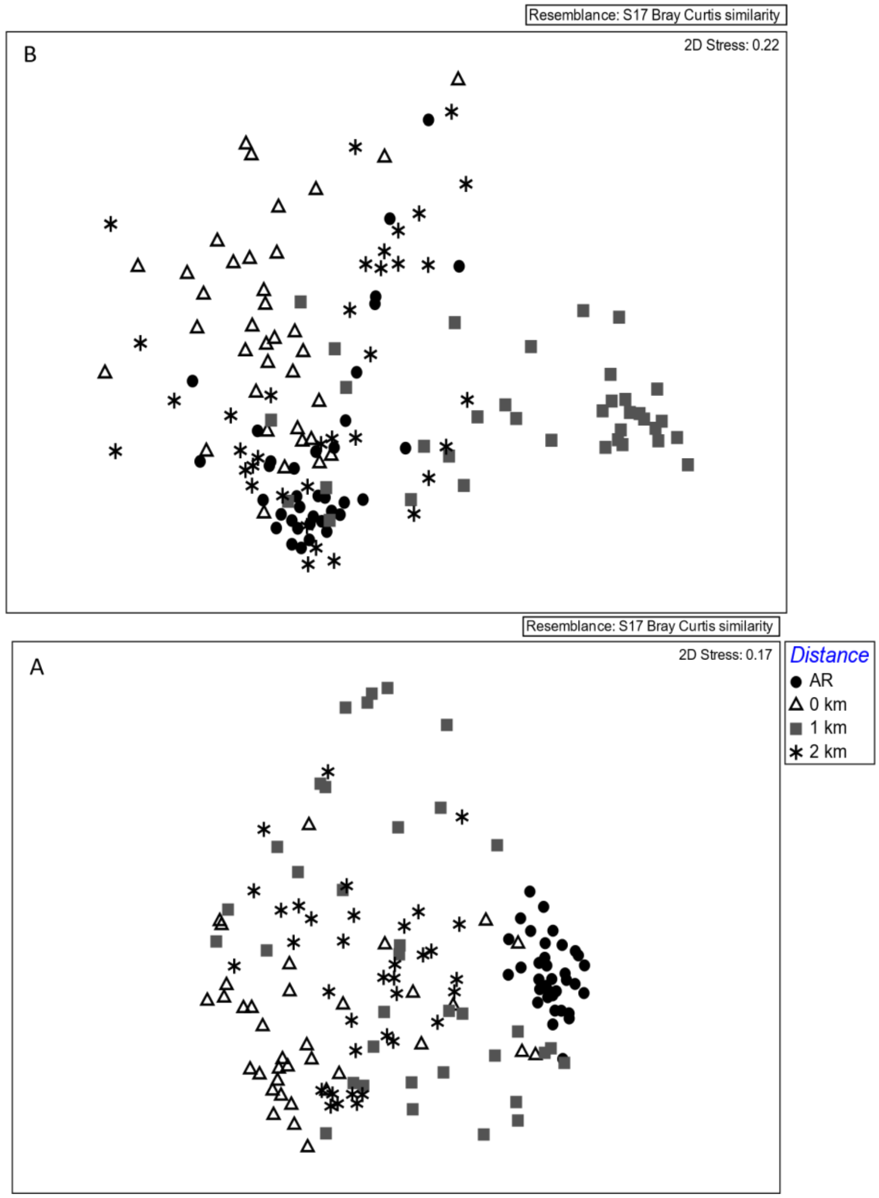

Figure 4 - Non-metric Multidimensional Scaling Analysis (nMDS) applied to the data of benthic percent coverage of the Artificial Reefs (AR) and Natural Reefs at 0, 1 and 2 km distances in Exposed (A) and Protected (B) area.

Through the analysis of similarity applied to these same data, was observed a higher values of R when comparing samples from AR and all of the distances (R> 0.48), and smaller values of R when comparing samples among NRs at different distances (R <0.34) (Table 1).

Table 1 - Results of the analysis similarity (ANOSIM one-way) applied to the data of benthic percent coverage of the Artificial Reefs (AR) and Natural Reefs (NRs) at 0, 1 and 2 km distances in Exposed and Sheltered areas.

R Global=0.49 p=0.01	R Value	p
Pairwise tests – Exposed area		
AR vs 0 km	0.8	0.01
AR vs 1 km	0.48	0.01
AR vs 2 km	0.84	0.01
0 km vs 1 km	0.19	0.01
0 km vs 2 km	0.34	0.01
1 km vs 2 km	0.24	0.01
Pairwise tests – Sheltered area		
AR vs 0 km	0.4	0.01
AR vs 1 km	0.59	0.01
AR vs 2 km	0.16	0.01
0 km vs 1 km	0.6	0.01
0 km vs 2 km	0.33	0.01
1 km vs 2 km	0.49	0.01

3.3.4 Sheltered Area

An amount of 43 invertebrate taxa were identified, where 25 were Porifera, 7 Cnidaria, 5 Ascidiacea, 3 Echinodermata, 1 Polychaeta, 1 Crustacea and 1 Bivalvia. For algae, all 4 groups were recorded. Among non-biological components all components were recorded, except Rocha.

The number of invertebrate taxa in sites varied as follows: 18 taxa at AR (mean 3.91 ± 1.59), 18 at 0 km (average 5.91 ± 1.2) 21 at 1 km (average of 4.33 ± 1.32) and 29 at 2 km (mean 5.9 ± 1.54). Of these, 8 were exclusive to AR, 4 to 0 km, 4 to 1 km and 10 to 2 km. The 4 algae groups were recorded in all sites.

The most abundant component was *Turf* with 32%, followed by Articulated coralline algae (13%), *P. caribaeorum* (10%), *Zoanthus* sp. (9%) Sedimentation+Turf (8%) Macroalgae (7%), Calcareous algae (5%), *P. variabilis*

(3%) and Sedimentation (2%). *Turf* was the dominant component at AR and at 2 km (57.5 and 28.8%, respectively), followed by *P. caribaeorum* (12.4 and 16.6% respectively) and Sedimentation+*Turf* (10.3 and 15 2%, respectively). Articulated coralline algae dominated on the 1 km, representing 49.9% of the total coverage and Turf was the second most abundant component with 15.5%. At the 0 km the dominant taxa were *Turf* (28.1%) and *P. variabilis* (12.1%), followed for *Zoanthus* sp. (10.3%) (Figure 4).

The nMDS analysis showed a trend toward separation among samples from different sites, with the exception of 2 km which overlapped samples of the AR. (Figure 5B).

Through the analysis of similarity applied to these same data, was observed a high similarity between AR and 2 km (**R** = 0.16). The higher **R** values were registered to comparisons among 1 km and the other sites (**R** < 0.49) (Table 1).

3.3.5 Artificial Reef vs Natural Rocky Reefs

Through results of the nMDS analysis applied to the percent coverage data at different reefs of the two areas, it is possible to observe the formation of a group of samples from ARs, indicating a high similarity between ARs and hence, a high dissimilarity with NRs, regardless of the distance (Figure 5).

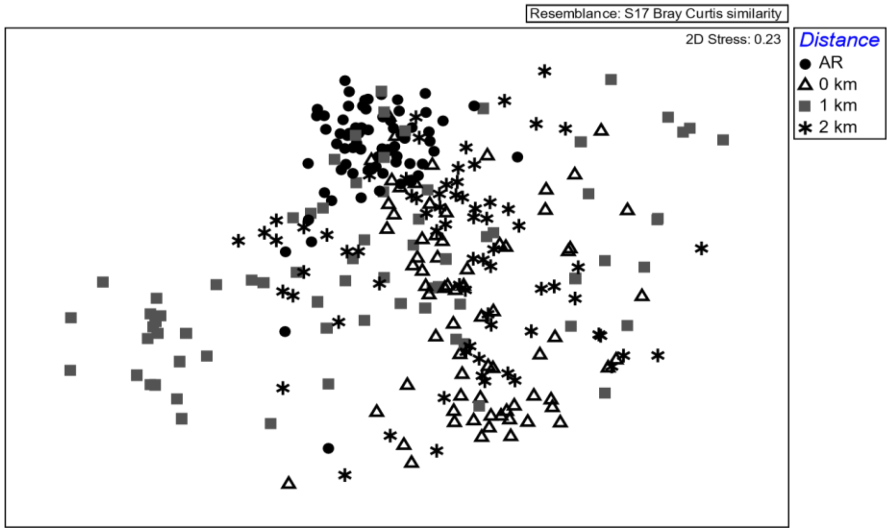

Figure 5 - Non-metric Multidimensional Scaling Analysis (nMDS) applied to all data of the benthic percent coverage of the Artificial Reefs (AR) and Natural Reefs at 0, 1 and 2 km distances in Exposed (A) and Protected (B) area.

The one-way ANOSIM analysis results indicate a high dissimilarity among samples from the ARs and NRs where **R** values for the pairwise comparisons are greater than 0.33. On the other hand, comparing NRs, the pairwise **R** values were considered low, all values lower than 0.019, indicating high similarity between these sites (Table 2).

Table 2 - Results of the similarity analysis (ANOSIM) applied to all data of the benthic percentage of coverage of the Artificial Reefs (RA) and Natural Reefs (NRs) at 0, 1 and 2 km distances in Exposedand Sheltered areas.

Global: R=0.78 p=0.01	R Value	p
Pairwise tests - ARs vs NRs		
AR vs 0 km	0.52	0.01
AR vs 1 km	0.33	0.01
AR vs 2 km	0.37	0.01
0 km vs 1 km	0.019	0.01
0 km vs 2 km	0.016	0.01
1 km vs 2 km	0.014	0.01

The result of SIMPER analysis showed a value of 73.91% of dissimilarity between ARs and NRs where 6 taxa of invertebrates, 4 components of algae and one non-biological component contributed with 90% of the dissimilarities found (Table 03). Of these, four components were the main contributors at the ARs and 7 at the NRs.

Table 3 – Percentage of Similarity (SIMPER) test results applied to the data of benthic percentage of coverage of the Artificial Reefs and Natural Reefs at distances 0, 1 and 2 km in Exposed and Sheltered areas.

Components	Average Cover		Dissimilarity Contribution
	Artificial Reef Group	Natural Reef Group	
Turf	53,29	22,62	24,84
Zoanthus sp.	0,86	15,95	10,86
Palythoa caribaeorum	6,52	6,74	7,97
Eudistoma saldanhai	11,39	0,00	7,73
Calcareous algae	2,66	10,79	6,83
Sedimentation + Turf	6,40	5,76	6,57
Articulated coralline algae	1,00	8,96	6,54
Macroalgae	7,90	4,87	6,28
Protopalythoa variabilis	0,26	7,30	5,06
Echinometra locunter	0,05	5,85	3,96
Ulva lactuca	0,00	5,01	3,40

3.3.6 Percent Coverage vs Environmental Parameters

The BIO-ENV analysis result provides Spearman correlation values lower than 0.082 between distance matrices of the both benthic percent coverage and environmental parameters. The variable that showed the greatest value of correlation was the depth ($\rho = 0.084$), followed by hydrodynamics ($\rho = 0.040$), index of rugosity ($\rho = 0.017$) and visibility ($\rho = 0.017$).

The result of CCA analysis (Figure 7) indicates that the axis 1 explains 53.82% of the data variation and the axis 2, 33.11%. Observing the ordination of samples in the graphic is possible to note that the axis 1 separates ARs samples from NRs samples. The vectors indicate that at this axis depth and hydrodynamic increase towards the AR Exposed area, where the taxon *E. saldanhai* was dominant. Taking into account the axis 2, the site 1 km tends to be separated from other NRs due to the increase in the index of rugosity. Despite these trends, the Monte Carlo permutation test, resulted in not significant correlations (*p*=0.39 to axis 1 and *p*=0.18 to axis 2).

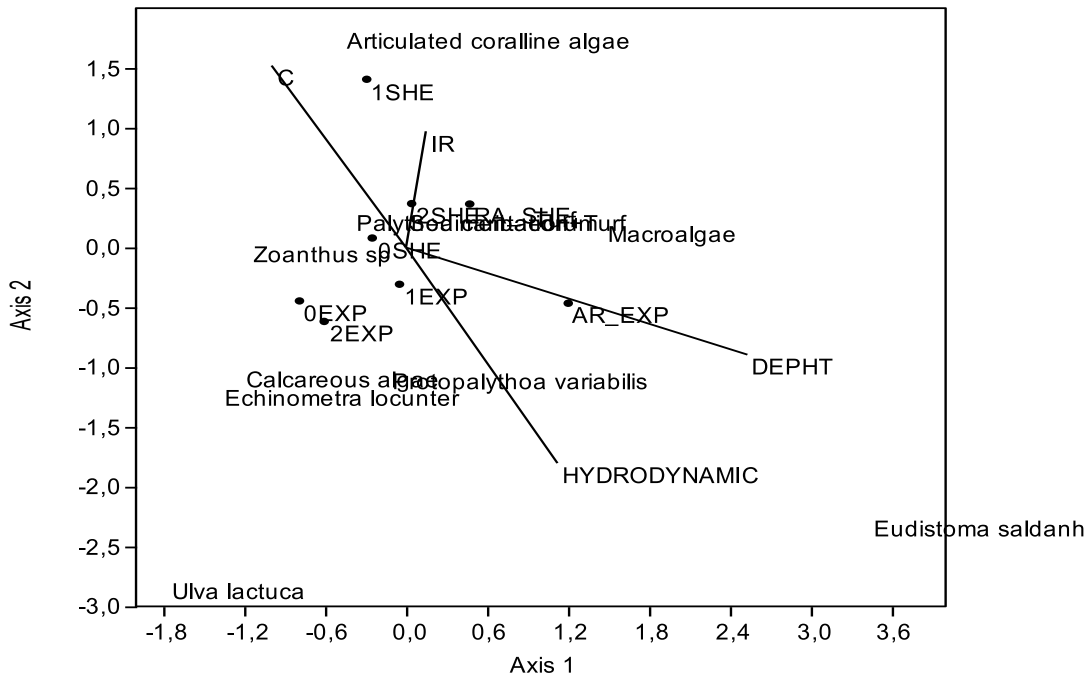

Figure 7 - Canonical Correspondence Analysis (CCA) applied to all data of benthic percent coverage and environmental parameters of the Artificial Reefs (AR) and Natural Reefs at distances 0, 1 and 2 km in the Exposed (EXP) and Sheltered (SHE) areas (IR=index of rugosity).

3.4 Discussion

The benthic community at the artificial reefs studied here, in the sheltered area as well as in the exposed one, presented high dissimilarities with the natural rocky reefs. These dissimilarities are regarded to univariate community descriptors such as: composition, number of taxa and percent coverage. Multivariate analysis also evidenced dissimilarities among samples from ARs and NRs. Although the NRs own certain degree of within samples dissimilarities, the overlap of communities were high in these environment.

These results are mainly due to lower values of number of taxa and high dominance for both ARs, where *Turf* component reach 50% of total benthic coverage. On the other hand, although *Turf* had occurred in relevant numbers at the NRs, other components of benthic coverage such as calcareous algae and the zoantharians *Zoanthus* sp. and *Protopalythoa variabilis* were the dominants.

Sublittoral rocky reefs are commonly dominated by different species of algae, cnidarians and ascideans in higher or lesser extent, alternating dominance (Moreno & Rocha, 2012). The studied rocky reefs are quite different from other rocky reefs systems from Brazil, in either tropical or subtropical areas, in the sense that they are shallower and narrower (Mendes et al., 2015). Different from our results, Mendes et al. (2015), sampling this same rocky reefs in 2008, registered high dominance of *Turf* and suggested that it is related to low depth and high sunlight incidence, which allowed this component to spread out. In general, shallower waters, with greater light intensity and moderate physical disturbance, favors macroalgae dominance by higher supply of nutrients and low sedimentation (Oigman-Pszcsol et al., 2004). The herbivory is pointed by many authors as the main factor structuring rocky reefs where fish and sea urchins, in particular, graze flexible macroalgae and favor the establishment of encrusting coralline algae (Ayling, 1981, Ojeda & Deaborn, 1989 Oigman-Pszcsol et al., 2004).

Aseltine-Neilson et al. (1999), investigating the occurrence of *Turf* at ARs in shallow waters, in a mediterranean climate, concluded that after 10 to 15 years this community should be similar to adjacent natural reefs. Here, the ARs, both older than 100 years, were located in the deepest places of the studied area with high and moderate hydrodynamic and even that, they still differ from adjacent NRs and were dominated by *Turf.*

Turf shows rapid growth, high reproduction rates, recruiting earlier than larvae of invertebrates (Oigman-Pszczol et al**.,** 2004)**,** and in natural/optimal conditions**,** it seems to be limited by herbivores such as sea urchins and fishes or by competition with other incrusting organisms (i.g.: calcareous algae and cnidarians) (McCook et al., 2001; Vermeij et al., 2010). Some authors also use the high dominance of *Turf* as an indicative of reef degradation such as Azevedo et al. (2011) who related it with intense tourism activities.

Sea urchins were relative abundant in the NRs and almost a negligible component in the ARs. Added to that, several authors recorded high abundance of herbivorous fishes at these NRs (Floeter et al., 2001; Longo et al 2014; Ferreira et al., 2015). It seems that these biological interactions might be responsible for the patterns of dominance found here and, hence, the dissimilarities among ARs and NRs.

According to Perkol-finkel (2005 and 2006), at the moment that an AR is introduced adjacent to natural reefs in the marine environment, the benthic community structure tends toward similarity in a matter of decades. However, when substrates complexity differs among reefs, no matters if artificial or natural, such communities will be different even after more than a century (Perkol-Finkel et al., 2006; Walker et al., 2007; Hunter & Sayer, 2009).

In this way, our results could be also related to the reefs' substratum heterogeneity. However, the lack of correlation between distance matrices generated to percent coverage and environmental factors measured, as well as, the lack of significance registered by the Monte Carlo permutation to correlations among percent coverage and environmental gradients, lead to believe that the index of rugosity, used as a measure of complexity, and also hydrodynamics, visibility and depth, had no stronger influence on the studied communities. This conclusion is also supported observing the values of the index registered at the AR and 2 km NR in the sheltered area. Although the values registered at these locals had been the most extreme in this study, these reefs presented high degree of overlap in communities, as one can see by both the nMDS and ANOSIM results.

Although many authors suggest the habitat complexity as a factor which rules the benthic community structure in terms of organisms' abundance, composition and diversity, other environmental factors are suggested to be limiting to these organisms such as depth, surface orientation (if horizontal or vertical), and Exposed (Walker, 2007), and hence, considering the complexity as a lesser importance factor.

Through the partial analysis of exposed and sheltered data it is possible to note trends toward different results from what was found in the analysis of the data as a whole. The high overlap in communities from AR and 2 km NR at the sheltered area and high dissimilarity among the 1km NR and all of the others NRs, also in this area**,** are examples of these outlier patterns.

These findings reinforce the lack of relationships with the presence of the ARs or distances of them from the NRs, or even the lack of relationships with the environmental gradients. The 1 km NR was dominated by articulated calcareous algae followed by *Turf.* This condition might be result of human activities on natural rocky reefs, mainly pollution in this case (Airoldi et al., 2008), once these components are able to resist to unfavorable conditions, colonizing spaces left by other species less tolerant as zoantharians (Morcon & Workeling, 2000). A high concentration of solid wastes such as different types of plastic, cans, piece of clothes and household items**,** was registered at this NR (unpublished data), evidencing the human disturbance at this place.

The higher similarity between the sheltered AR and the most distant NR (2 km) and the lesser with the nearest (0 km), constitutes evidence that there is no influence or evident pattern of distances in benthic coverage structure among artificial reefs and natural rocky reefs and that other indirect factors might be driving these communities such as human impacts.

Tourism activities also can cause great damage to reef systems mainly by physical disturbance through trampling and diving, especially in urban zones (Barradas et al., 2012).

Shipwrecks are often used for recreational dive activities (Carvalho, 2015) and shallower ones are likely to be most visited by inexperienced SCUBA divers and free divers. Giglio et al. (2015) registered twice the rate of diver contacts in shipwrecks than natural reefs. These kinds of contacts could damage more fragile organisms, mainly zoantharians (Santos et al., 2015).

In a different way, some authors found relationships in benthic communities related to proximity from an artificial reef. Wilding and Sayer (2002), studying the influence of a gradient of distances from an AR on the adjacent soft bottom community verified low abundance and diversity of macroinvertebrates near the AR and an increasing in these descriptors at distances higher than 5 meters. Some works have demonstrated that although changes in benthic community could appear after an AR introduction, such changes have a low magnitude and are rapidly dissipated by currents with increasing distances (Davis et al., 1982; Zalmon et al., 2012)

The great majority of studies that investigates hard bottom communities have focused on similarities between artificial and natural reefs regardless distances (Aseltine-Neilson et al., 1999; Perkol-Finkel et al., 2006; Thanner et al., 2006; Perkol-Finkel & Benayahu, 2009; Burt et al., 2009; Burt et al., 2011; Coelho et al., 2012) and suggests that there are, in general, few similarity (Perkol-Finkel et al., 2005; Perkol-Finkel et al., 2006; Burt et al., 2009; Coelho et al. 2012).

Thanner et al (2006), in a small scale design (few meters) suggested high degree of overlap among artificial and natural reefs. Perkol-Finkel et al. (2006) in the same way also registered a certain degree of similarity but only for horizontal surfaces of the shipwreck and with high habitat heterogeneity. The few studies which used large spatial scale (km) pointed to high dissimilarity among artificial and natural reefs (Burt 2009, Coelho 2012) as we found here. These findings raise questions about the spatial scale in which this interactions or relationships could be relevant and hence, detectable.

At the present study it is clear that in a spatial scale of hundreds of meters, the degree of similarity among artificial reefs and natural rocky reefs is low, even after more than a century, and overcome by other factors, mainly biological interactions and human activities. In this way, both of our proposed hypothesis: (i) benthic communities from artificial reefs with more than 100 years of introduction are similar to natural rocky reefs and (ii) that these similarities are influenced by the distance among them, were entirely rejected.

In order to elucidate the role of artificial reefs in marine environments is suggested focus on small scale approach for complimentary future studies**.** Understand the extend of human activities damages is also of great relevance once costal reefs are over high human pressure and it seems that these activities may be considered as a structuring factor driving benthic communities.

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