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NÍDIA CRISTIANE DE MELO MARINHO

MACROFAUNA EM ÁREAS COM E SEM ESTRUTURAS DE DEFESA COSTEIRA

Recife
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NÍDIA CRISTIANE DE MELO MARINHO

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Tese apresentada ao Programa de Pós-Graduação em Biologia Animal da Universidade Federal de Pernambuco, como requisito parcial para obtenção do título de doutor em Biologia Animal.

Orientador: Prof. Dr. Paulo Jorge Parreira dos Santos

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RESUMO

As zonas costeiras, por sua complexidade natural e relevância socioeconômica, vêm sendo alvo de crescentes intervenções para conter a erosão, especialmente diante do agravamento provocado pelas mudanças climáticas e a elevação do nível do mar. No Brasil, o problema é particularmente evidente no Nordeste, com destaque para Pernambuco, onde cerca de um terço do litoral apresenta processos erosivos. Municípios como Paulista, Olinda, Recife e Jaboatão dos Guararapes têm adotado diversas obras de defesa costeira, muitas vezes sem conhecimento adequado sobre seus impactos ecológicos. Nesse contexto, este estudo avaliou a resposta da macrofauna bentônica à implantação de estruturas de defesa costeira (EDCs) no litoral pernambucano em três fases. A primeira, por meio de revisão sistemática, bibliométrica e meta-análise, mapeou o conhecimento global sobre efeitos de EDCs na macrofauna bentônica (2002-2024), revelando aumento expressivo de publicações após 2015 e que quebra-mares de concreto são os mais comuns, com impactos negativos significativos na abundância e densidade da macrofauna, além de lacunas como ausência de dados sobre materiais e variáveis abióticas. Na segunda fase, nove áreas de Pernambuco foram avaliadas, confirmando padrões globais: EDCs favorecem deposição de sedimentos mais finos e comunidades dominadas por espécies oportunistas, enquanto ambientes naturais mantêm comunidades equilibradas. Por fim, a terceira fase comparou conjuntos de quebra-mares do Janga e Casa Caiada, evidenciando que características do projeto influenciam diretamente seus efeitos. O Janga, com estruturas mais próximas da praia e menor distância entre elas, apresentou maior deposição sedimentar e homogeneização biológica, enquanto Casa Caiada demonstrou impactos inferiores e maior heterogeneidade. Conclui-se que, embora os efeitos negativos das EDCs sejam predominantes, compreender suas configurações é crucial para influenciar a resposta biológica local. Ao integrar abordagens globais e análises locais, este estudo avança o conhecimento sobre impactos ecológicos das EDCs, oferecendo subsídios para intervenções mais sustentáveis e fundamentadas na conservação ambiental.

Palavras-chave: Macrofauna, Praias arenosas, Impactos, Estruturas de Defesa Costeira, Erosão.

ABSTRACT

Coastal areas, due to their natural complexity and socioeconomic importance, have been the target of increasing interventions to curb erosion, especially given the worsening caused by climate change and rising sea levels. In Brazil, the problem is particularly evident in the Northeast, particularly in Pernambuco, where approximately one-third of the coastline is experiencing erosion. Municipalities such as Paulista, Olinda, Recife, and Jaboatão dos Guararapes have adopted various coastal defense projects, often without adequate knowledge of their ecological impacts. In this context, this study evaluated the response of benthic macrofauna to the implementation of coastal defense structures (CDSs) on the Pernambuco coast in three phases. The first, through a systematic and bibliometric review and a meta-analysis, mapped global knowledge on the effects of EDCs on benthic macrofauna (2002-2024), revealing a significant increase in publications after 2015 and that concrete breakwaters are the most common, with negative impacts on macrofauna abundance and density, in addition to gaps such as a lack of data on materials and abiotic variables. In the second phase, nine areas of Pernambuco were evaluated, confirming global patterns: EDCs favor finer sediment deposition and communities dominated by opportunistic species, while natural environments maintain balanced communities. Finally, the third phase compared sets of breakwaters in Janga and Casa Caiada, showing that project characteristics directly influence their effects. Janga, with structures closer to the beach and shorter distances between them, showed greater sediment deposition and biological homogenization, while Casa Caiada presented lower impacts and greater heterogeneity. It is concluded that, although the negative effects of EDCs are predominant, understanding their configurations is crucial to influencing local biological responses. By integrating global approaches and local analyses, this study advances knowledge about the ecological impacts of EDCs, providing support for more sustainable interventions based on environmental conservation.

Keywords: Macrofauna, Sandy beaches, Impacts, Coastal Defense Structures, Erosion.

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1 INTRODUÇÃO GERAL

As zonas costeiras**,** áreas geográficas especiais na interface entre os sistemas marinho e terrestre, destacam-se não apenas por sua extrema complexidade ambiental, mantendo equilíbrio dinâmico a curto prazo através de forças naturais (ondas, correntes, maré e vento) e a longo prazo em resposta às flutuações do nível do mar, mas também por sua fundamental importância socioeconômica, decorrente da concentração populacional, do desenvolvimento econômico intensivo e da abundância de recursos naturais que sustentam atividades vitais**.** Esta combinação única de dinamismo natural e relevância humana as consolida como um dos ambientes mais críticos e ativos da superfície terrestre (CROSSLAND *et al*., 2005; MANSO *et al.,* 2006).

Adicionalmente, é crucial ressaltar que as zonas costeiras abrigam inúmeros ecossistemas marinhos compostos por espécies únicas, reforçando sua riqueza biológica. Sua relevância multidimensional, expressa nos âmbitos econômico (recursos naturais, pesca, turismo), social (alta concentração populacional, infraestruturas críticas) e ambiental (serviços ecossistêmicos, regulação climática), aliada à fragilidade de seus processos naturais (como dinâmica sedimentar e flutuações do nível do mar), demanda urgentemente estudos aprofundados. Essas pesquisas necessitam gerar informações precisas para proteger tais regiões contra riscos costeiros, entre os quais a erosão costeira destaca-se como um dos principais desafios (FOTI *et al*., 2020)

No Brasil, a erosão costeira e o avanço do mar são mais evidentes na região Nordeste, e mais fortemente em Pernambuco (COSTA *et al*., 2010). Estima-se que 1/3 do litoral pernambucano sofre, em níveis diferenciados, com processos erosivos e recuo da linha da costa (MANSO *et al*., 2006), situação que deve se agravar como consequência da subida do nível do mar decorrente de mudanças climáticas globais (MARENGO *et al*., 2016). Em vários trechos do estado, houve forte intervenção pública com construção de diversos tipos de obras de contenção do avanço do mar (quebra-mares, engorda, enrocamento e *seawalls*) nos municípios de Paulista, Olinda, Recife e Jaboatão dos Guararapes (ARAÚJO, 2007).

O histórico de instalação de estruturas de defesa costeira em Pernambuco é antigo, mas ganhou maior intensidade a partir da década de 1970, com a construção de grandes obras nas praias de Casa Caiada e Rio Doce, em Olinda. Nas décadas seguintes, o agravamento do problema da erosão em outras regiões levou à construção de novas estruturas, como na praia do Janga, no município de Paulista (PEREIRA *et al.*, 2007; SILVA *et al.,* 2005).

Em Olinda, os registros de erosão remontam a 1909, após a expansão do Porto do Recife. A partir da década de 1950, iniciaram-se intervenções com estruturas rígidas, culminando na construção de um conjunto de sete quebra-mares ao longo das praias de Bairro Novo, Casa Caiada e Rio Doce, concluído entre as décadas de 1970 e 1980 (PEREIRA *et al.,* 2007). Apesar de mitigar parcialmente a erosão, essas obras promoveram alterações na topografia e na hidrodinâmica locais, resultando em forte deposição de sedimentos, formação de saliências junto aos quebra-mares e, em alguns pontos, grandes bancos de areia (PEREIRA *et al.,* 2006).

Em Pernambuco, é comum que os problemas de erosão sejam transferidos para áreas adjacentes, de sul para norte, à medida que novas estruturas são instaladas e o fluxo de sedimentos transportado pela deriva litorânea é interrompido ou reduzido. Assim, a praia do Janga sofreu forte impacto erosivo entre as décadas de 1980 e 1990, agravado pela transferência de erosão de Olinda, após a construção das estruturas de defesa no trecho Rio Doce-Paratibe, e pelas obras já existentes em Casa Caiada e Rio Doce (PEREIRA *et al.*, 2007).

Na década de 1990, iniciou-se a construção de nove quebra-mares paralelos à linha de costa da praia do Janga, concluídos em 2001 (SILVA *et al.,* 2015). Desde então, a paisagem local passou por mudanças significativas, com formação de pequenas baías, intensa deposição de sedimentos junto aos quebra-mares e erosão acentuada no centro dessas baías. Análises realizadas por Silva et al. (2015) indicaram que a maioria das baías ainda não havia atingido equilíbrio estático e que o processo erosivo provavelmente continuaria.

Atualmente, cerca de 20 anos após a conclusão do projeto, já é possível observar a formação de dois tômbolos na praia do Janga, resultado dos processos de erosão e deposição. Em outros quebra-mares, há acúmulo de sedimentos, formando saliências adjacentes às estruturas. Considerando a tendência observada, é provável que, ao longo do tempo, novos tômbolos se formem, fragmentando a praia contínua em pequenas praias individuais.

Embora as obras de defesa costeira, sobretudo os quebra-mares, sejam comparadas aos recifes naturais quanto à sua função, os recifes artificiais diferem dos naturais em diversos aspectos, como na composição e na complexidade do habitat (SEDANO *et al*., 2020). Essas estruturas são frequentemente associadas a efeitos negativos, como modificações na simetria da praia, interferem no dinamismo da costa e na transferência de sedimentos, modificam ou transferem o problema da erosão para outras áreas da praia não protegidas nas proximidades, introdução de estruturas duras no habitat de fundo macio, modificação do perfil praial, represamento e redução da qualidade da água e diminuição da beleza cênica da praia (PILKEY; WRIGHT, 1988; DUGAN *et al*., 2008; ANFUSO *et al.,* 2017).

Mesmo não sendo projetadas para fins ecológicos, essas obras de defesa costeira provavelmente desempenharão um papel ecológico cada vez mais relevante nos ecossistemas marinhos costeiros, uma vez que as populações humanas continuam a crescer nas áreas costeiras (SALE *et al*., 2010). Dessa forma, é de vital importância que se desenvolva uma compreensão do desenvolvimento das comunidades biológicas nas áreas protegidas por essas estruturas e como elas se diferenciam daquelas dos habitats naturais. A macrofauna de fundo mole tem sido amplamente utilizada para detectar mudanças no ambiente bentônico e pode fornecer valiosas contribuições sobre as modificações que estão acontecendo nessas áreas.

De forma geral, as associações de macrofauna são influenciadas por alterações relacionadas aos ambientes em que estão inseridos, sendo controlados por fatores como temperatura, salinidade, hidrodinâmica, tipo de substrato, também por processos bióticos (competição, predação, migração, recrutamento) e por efeitos antrópicos (eutrofização, obras de engenharia costeira), atuando esses fatores em diferentes escalas tanto espaciais, quanto temporais (BRAULT; BOURGET, 1985; BONADA *et al*., 2007).

Embora trabalhos sobre os impactos da instalação de obras de defesa costeira tenham sido desenvolvidos (WILDING, 2006; WALKER *et al*., 2008; WEHKAMP; FISCHER, 2013 a; b; MUNARI, 2013, COOMBES *et al*., 2015, SEDANO *et al*., 2020), ainda se sabe relativamente pouco sobre os efeitos da perda e modificação de habitats sobre as comunidades que vivem nos sedimentos inconsolidados das regiões mais atingidas por essas alterações. Dessa forma, o objetivo do presente trabalho é avaliar a resposta da macrofauna bentônica ao impacto causado por instalações de obras de defesa costeira, comparando as associações que habitam áreas influenciadas por obras instaladas no nosso litoral com aquelas de habitats naturais.

2 OBJETIVOS

Objetivo geral

Avaliar a resposta da macrofauna bentônica ao impacto causado pela instalação de obras de defesa costeira no litoral pernambucano.

Objetivos específicos

- Analisar a produção científica sobre efeitos das obras de defesa costeira na macrofauna bentônica;
- Descrever e caracterizar a estrutura das associações da macrofauna bentônica em áreas com a presença de estruturas artificiais de defesa costeira, áreas com a presença de recifes naturais e praias expostas, sinalizando espécies indicadoras de cada ambiente;
- Comparar e identificar diferenças na estrutura das associações da macrofauna bentônica em duas praias que sofreram fortes modificações estruturais devido à implantação de grandes conjuntos de quebra-mares.

3 ESTRUTURA DA TESE

A presente tese organiza-se sob a modalidade de artigos compilados, compreendendo os seguintes elementos: uma introdução geral, três capítulos autônomos configurados como artigos científicos, considerações finais e o referencial bibliográfico que fundamenta o conjunto da tese.

O primeiro capítulo intitulado “Impact of coastal defense structures on macrobenthos: integrated systematic and bibliometric investigation along with a meta-analysis” analisa a produção científica global (2002-2024) sobre os efeitos de estruturas de defesa costeira (EDC) na macrofauna bêntica, mediante uma abordagem integrada que combina revisão sistemática e meta-análise com foco em sedimentos inconsolidados.

O segundo capítulo intitulado “Macrobenthic communities in coastal environments: artificial defense structures, natural reefs and exposed beaches**”** teve como objetivo central comparar a estrutura das associações da macrofauna bentônica em três cenários distintos: (1) áreas com estruturas de defesa costeira, (2) áreas protegidas por recifes naturais e (3) praias expostas sem proteção, identificando espécies indicadoras e parâmetros ecológicos-chave associados a cada ambiente.

O terceiro capítulo intitulado “Design differences between two sets of breakwaters have different effects on the benthic macrofauna of sandy beaches” teve como objetivo avaliar a estrutura de comunidades de macroinvertebrados bentônicos, comparando as estruturas funcionais taxonômicas e tróficas que habitam duas praias que foram altamente modificadas após a construção de grandes conjuntos de quebra-mares.

4 CAPÍTULO 1 - Impact of coastal defense structures on macrobenthos: integrated systematic and bibliometric investigation along with a meta-analysis

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ABSTRACT

Based on scientific articles discussing the impact of coastal defense structures (CDS) on benthic macrofauna published between 2002 and 2024, this paper analyzes global scientific production through an integrative review combined with a meta-analysis focusing on CDS effects on soft sediment macrofauna. To support reading a detailed analysis was carried out to understand the growth in literature, authorship patterns, network of keywords, distribution of production between countries, main types of CDS, most used materials and future challenges for this area of research. An increase in publications and citations was observed in recent years. The most frequent type of structure was breakwaters, and the most used material was concrete. Although scientific production on the subject has advanced in recent years, there is a strong heterogeneity in the data available in the documents, in addition to the absence of some fundamental data. Through meta-analysis, it was possible to conclude that most studies (65.3%) indicated impact effects, direct or indirect, most (78.7%) of which included decreases in either richness or abundance/density of the soft sediment macrofauna in areas with the presence of CDS. Meta-analysis results suggest that CDS have a greater effect size on abundance/density than on the richness of organisms and that different types of structures cause different magnitudes of effects. Another critical point is that some studies indicate that there has been disturbance or change in the community structure in areas with CDS, but due to the high heterogeneity of the data and the fact that the cases are not treated independently, generally being grouped as a result of human activities, the understanding of the real level of impact caused by CDS is frequently compromised.

Keywords: Coastal engineering, systematic review, benthos, beaches, scientific articles

Introduction

Many of the world's coastlines face problems related to coastal erosion (Pilkey & Cooper 2014). With the current scenario of temperature increases and sea level rise, it is expected that erosion processes will soon reach uncontrollable proportions (IPCC, 2019). This problem becomes much more concerning considering that coastal regions are ideal places for human concentration and the development of various economic activities, such as industry and tourism (Barragan & Andreis, 2015). Thus, climate management strategies must be developed to protect coastal infrastructures, which often involve introducing rigid coastal defense structures (CDS) such as breakwaters, jetties, and seawalls (Nordstrom, 2014).

Defense structures are considered poor substitutes for natural structures as they differ from natural structures in several aspects, for example, the composition of colonizing organisms (Coombes *et al*., 2015), habitat complexity (Loke & Todd, 2016), influence on competitive interactions between species (Jackson *et al.,* 2008) and in facilitating the colonization of exotic species (Tanasovici *et al.,* 2020). All these characteristics strongly influence the establishment and development of benthic communities.

Benthic organisms are widely used in ecological investigations because they are sessile and easily traceable (Sedano *et al*., 2020). Much of the current knowledge on how benthic macrofauna responds to the introduction of artificial structures focuses on testing and understanding how physical attributes, such as type (Spagnolo *et al.,* 2014; Becker *et al.,* 2020), orientation (Hanlon *et al.,* 2018), texture (Zawada *et al.,* 2010) and physical location of the hard substrate (Vaz-Pinto *et al.,* 2014), affect colonization and/or community structuring. However, little is known about how their introduction and physical attributes affect the macrofauna community of adjacent soft sediments. Recent studies addressing anthropogenic impacts on sandy beaches have focused on reporting how the context (*i.e.* local morphodynamic factors) and other cumulative anthropogenic stressors impact macrofauna differently (Costa *et al.,* 2022) and point out that studies involving CDS receive little attention in the literature (Afghan *et al*., 2020).

Due to the increase in the construction of CDS, the importance of using methods to record and analyze the evolution of scientific data published in consolidated databases related to this topic is growing, for example, in systematic reviews. Unlike traditional reviews, systematic reviews adopt a replicable, scientific, and transparent process, through a defined methodology, which aims to minimize possible bias in research (Tranfield *et al.*, 2003). Combined with this type of review, it is possible to introduce bibliometric reviews. The concept of bibliometrics was first used by Pritchard (1969), who related it to studies that seek to quantify written communication processes. The main idea of bibliometrics is to analyze existing scientific production on certain topics through database research. This type of study has become popular and necessary in recent decades due to the large volume of material produced and made available daily (Araújo, 2006).

Although review studies based on bibliometric analysis have grown in recent years, information regarding this line of research, *i.e.*, the effects of CDS on marine organisms, is scarce. Organizing the information available on this topic can assist in scientific research or even in the construction of new defense structures that can be developed in the future. To respond to existing gaps, this study uses scientometric tools, which integrate qualitative analyses of information, mapping, and grouping of knowledge, as well as content analysis to identify research progress and the knowledge system on the topic.

Combined with the systematic and bibliometric review, it is also important to include a meta-analysis, as this type of analysis provides a robust view of the issue investigated, identifying the homogeneity and antagonistic effects of previously published studies (Aguinis *et al.,* 2011; Junni *et al.*, 2013). This junction offers a statistical procedure, quantifying the effect of the degree of variation of the factors that can affect the variables analyzed, and synthesizing the results of the paper, providing a higher level of reliability, which is unavailable in isolated studies (Tranfield *et al.,* 2003).

In this context, this article aims to (I) develop a systematic and bibliometric review from the perspective of increased protection of global coastlines, discussing quantitative and qualitative aspects of scientific production focusing on the effects of CDS on marine benthic macrofauna, and (II) carry out a meta-analysis on the impact of defense structures on the richness and abundance/density of marine benthic macrofauna that inhabit soft sediments calculating the effect size magnitude for these parameters, and whether the effect size varies according to the type of coastal defense structure.

Methodology

Systematic and bibliometric review and meta-analysis dataset

For the systematic and bibliometric review and meta-analysis of available literature, we used the Preferred Reporting Items for Systematic Reviews and Meta-analysis (PRISMA) guidelines (Moher *et al.,* 2015).

The literature review was performed based on articles available in the Scopus database on September 24, 2024, using the following terms in the main search field: breakwater OR jetties OR "rip raps" OR groynes OR seawall OR "coastal engineering" OR "coastal protection structures" OR "coastal structures" OR "coastal armoring" AND macrofauna OR macroinvertebrates OR macrobenthos OR benthos OR benthic. The search queried the “Article title”, “Abstract” and “Keywords” database available up to September 2024.

A total of 414 documents were found, 11 reviews were excluded, and one was duplicated, leaving the remaining 402 papers. To further refine the data, a close-up inspection was carried out manually to exclude off-topic publications, i.e., articles that did not respond to the proposed theme, such as papers carried out in freshwater environments and studies that did not use macrofauna as the object of study.

After applying manual exclusion criteria, 58 publications were selected. Data collection was organized around three main axes: (I) bibliometric information on the publications, (II) the methodological characteristics of the publications, and (III) a meta-analysis to quantify the effects of CDS on the richness and abundance/density of marine benthic macrofauna.

For the bibliometric information, the following topics were used: documents by year, citations, journals, co-occurrence of keywords, countries, co-authorship of countries/territories, authors and co-citation. For the methodological characteristics of the publications were used: defense structure type, defense structure material, structure orientation, methodology used in the document, type of substrate, and eco-engineering.

For the meta-analysis, within the bibliometric dataset, only documents developed on soft sediments (n=25) were selected, 11 of which were discarded because they did not present the necessary information about richness and abundance. An active search was also performed based on the cited references in the articles used, resulting in the selection of 2 more, totaling 16 papers.

A diagram is presented in Figure 1 demonstrating the criteria used for the bibliometric review and meta-analysis adapted to the PRISMA guidelines.

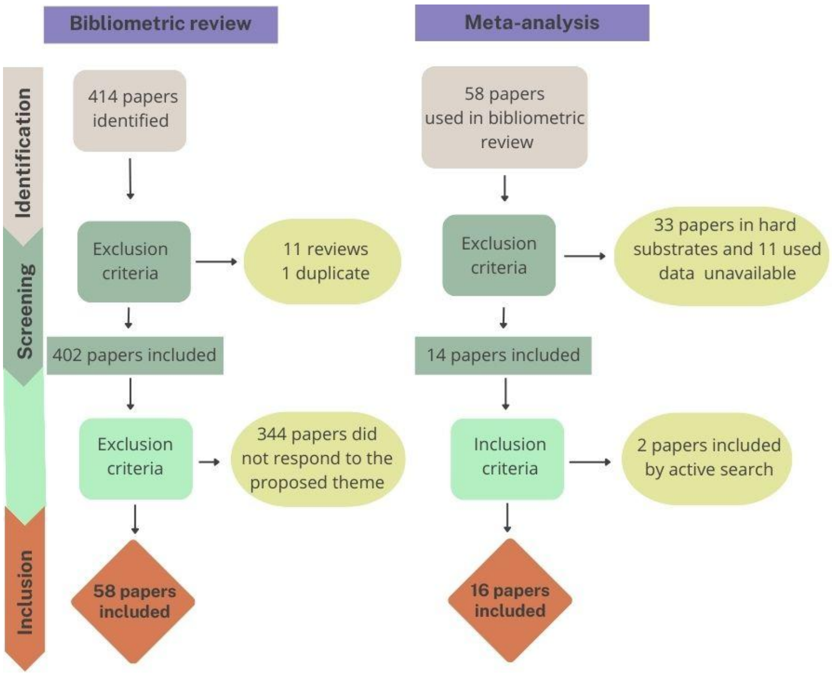

Figure 1. PRISMA analysis flowchart for bibliometric review and meta-analysis.

Analysis method

To analyze the journals in which the selected articles were published, we used CiteScore, which is a metric developed by Scopus, an Elsevier database, which establishes the impact of citations in journals.

The bibliographic data of the documents were saved as a CSV file and bibliometric analyses were performed using the VOSviewer software (https://www.vosviewer.com). VOSviewer is considered a multidimensional scaling tool (Van Eck & Waltman, 2010) that allows the creation, visualization, and exploration of maps based on bibliometric data available in online databases. The output results are displayed in groups, allowing the clear visualization of connections between the data (Pauna *et al.,* 2019). Here, this program was used to investigate the relationships between countries, authors, keywords, and documents involved.

The meta-analysis was carried out using the logarithm response ratio (effect/control) for community richness and abundance/density, *i.e.*, the biodiversity parameters discussed. The effect was calculated based on the information contained in the articles regarding the differences between areas with the presence of defense structures and control areas. On the logarithmic scale, an effect size of zero means no difference, while a negative value means that there was a decrease in the variable examined. A positive value signifies an increase, resulting from the effect of defense structures on the community.

To allow a better understanding of the magnitude of the effect of structures on benthic communities, we explore the effect sizes on both richness and abundance/density separately, as well as comparing the effect sizes of the different types of CDS using a boxplot. To verify whether there are significant differences of effect sizes between the control and affected areas for richness and abundance/density parameters, and between the different types of defense structures, except for Groyne since there was only one observation on this type of structure, a permutational analysis of variance (ANOVA) was performed considering two factors, type of structure and biological parameter (nested in type of structure). T-tests were also performed to observe whether there are significant differences between each parameter (richness and abundance/density) or each type of CDS average effect size and a theoretical no-effect. We adjusted the data to consider the effect size in module, in order to exclude the effect of the signal (positive or negative), thus avoiding positive values eliminating negative values, accepting the premise that regardless of the signal, the effect would be considered as an impact. All analyses were conducted using Primer-E 7.0 PERMANOVA+, BioEstat 5.0, and PAST4.17c Paleontological Statistics software package for education and data analysis. The significance level adopted was 5%.

Results and Discussion

Bibliometric dataset

The 58 publications included in this analysis were published between 2002 and 2024 and were distributed irregularly, showing a pattern of growth over the years (Figure 2a). The progressive increase in the number of publications is a strong indicator of the development of some topics (Wang *et al.,* 2014) and reflects the importance of developing studies that can expand our knowledge in this area.

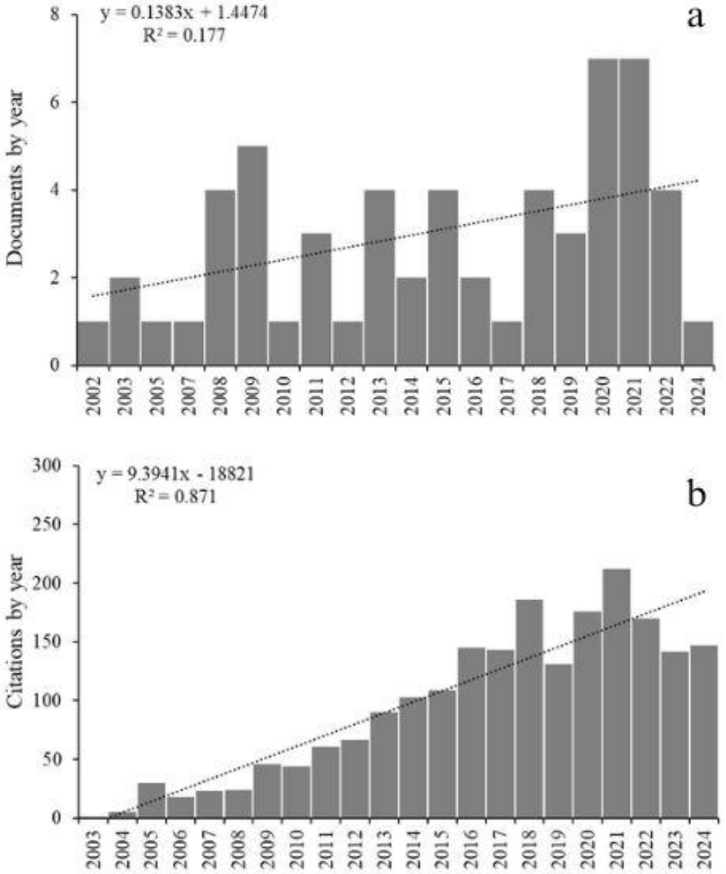

Figure 2. Number of documents (a) and citations (b) by year from 2003 to 2022.

The selected documents were distributed across 37 journals, including some high-impact journals. More than half (51.3%) of the journals had a 2023 CiteScore above 5, and 27% above 7. The weighted average of these journals' CiteScores produced a value of 10.29, which is equivalent to the 98th percentile in Animal Science and Zoology and the 97th in Aquatic Sciences, reinforcing the relevance of the topic. Regarding article citations, it is possible to observe a linear growth trend (Figure 2b).

The analysis of the co-occurrence of keywords, with a minimum number of 3 occurrences, included 217 author keywords. The most cited was "Biodiversity" which was mentioned seven times. "Adriatic Sea" and "Breakwater" were mentioned six times each, "Seawall" five times, and "Coastal structures" and "Intertidal" were mentioned four times each. Chronologically, it is possible to observe that some keywords were used more recently, such as "Ecological engineering", "Artificial structures", and "Coral reefs (Figure 3).

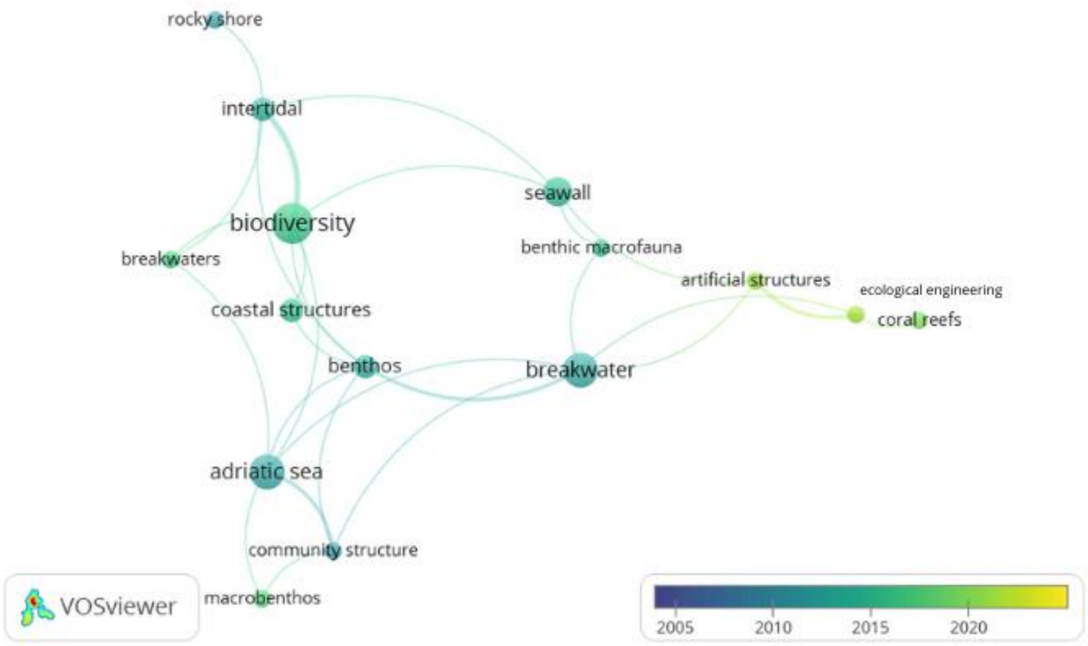

Figure 3. Co-occurrence network map of keywords publishing papers on the effect of coastal defense structures on benthic macrofauna.

Although the strength of the links between words was not very strong (total link strength=32), we observed a pattern between the terms used and the delimitation of the topic addressed. Although the terms used in the documents were similar, Afghan *et al.* (2020) observed a greater uniformity in the use of terms by authors in recent years, varying from “breakwater(s)” to “artificial defense structures” and “coastal defense structures”. When observing the choice of terms over time in the data set of the present study, we can observe a more recent insertion of some terms mentioned above, which may indicate a change in approach and vision on this topic, in addition to signaling a possible tendency towards future research.

In terms of study area, 31 countries/territories were observed, with Italy producing the greatest volume of research (13), followed by the United Kingdom, Australia, Spain and the United States with 9 documents each. Among the publications, the research carried out in the Mediterranean and the Adriatic Sea can be highlighted.

Regarding countries' contributions, in terms of the volume of published articles, European countries stood out. This high volume of research is mainly related to the development of the DELOS Project. The DELOS project (http://www.delos.unibo.it/) was developed in 7 European countries (Italy, Spain, Denmark, Sweden, Netherlands, United Kingdom, and Greece) and aimed to provide an inventory of low crest defense structures that exist in coastal areas and to analyze the hydrodynamics and stability of these structures, in addition to their effects on beach morphology. It also aimed to investigate the impacts of structures on biodiversity and the functioning of coastal assemblages through observations and field experiments at spatial and temporal scales and in relation to different environmental conditions (Hawkins *et al.,* 2010). Integrative projects like this are highly important for developing research that aims to understand problems caused by human actions, thereby enabling the development of new technologies that aim to mitigate the consequences caused by this type of impact.

When analyzing medical research, Man *et al.* (2004) indicated that differential spending on research between nations is responsible for part of the variation in the production of publications worldwide. They also highlighted that other causes can also influence this, such as the issue of English language proficiency. Since most high-impact journals are reviewed and published in English, this can be a determining factor for publishing in the best journals. The authors also noted that even relatively small changes in research funding, such as a 1% increase in research funding or a 10-point increase in a nation's certificate of English proficiency, were associated with large variations in scientific output. Another point is that according to Auranen & Nieminen (2010), the funding environments of university research vary across countries and there are significant country-specific differences among university systems, with regard to steering impulses and competition incentives, and how competitive funding systems are much more efficient in producing scientific publications.

International scientific research is described as the co-authorship of articles between researchers from at least two countries. The co-authorship country analysis included 31 countries that published documents on the theme of this review. Of these, 17 countries in the network are not connected. The largest set of connected countries consists of 16 items. The United States and the United Kingdom were the most connected with other countries (11), Italy was connected with 10, and Australia and Israel were connected with nine countries (Figure 4a).

Analyzing co-authorship between countries can clarify the type and relationship of collaborations between countries. An obvious issue when we look at the countries that contributed the most to scientific production is the country's development-status. According to Isfandyari-Moghaddam *et al.* (2023), the scientific advancement of these more developed countries can be attractive and inspiring for scholars from other countries to collaborate scientifically. Another possible point is that these countries invest more in terms of scientific equipment, laboratories and human resources than most other countries, therefore, they can be more attractive for the development of scientific collaboration.

Among the documents analyzed, we identified 207 authors. The authors with the highest number of publications were Espinosa F. with 5 documents, followed by Sedano, F. with 4 and Guerra-García, J.M., Navarro-Barranco, C., Masi, B.P., Zalmon, I.R., Abbiati, M. and Airoldi, L. with 3 documents. According to Scopus Author Metrics (citation data from the last 10 years), the authors with the highest number of publications also have the highest research significance, compared to other authors in this study, considering *h*-index, percent of documents co-authored with researchers in other countries and percentage of documents in the top 25% most cited documents worldwide.

Regarding co-citation, 5029 cited authors were found, of which 24 were mentioned at least 35 times. The most cited author was Hawkins, S.J. with 163 citations, followed by Chapman, M.G. with 155 citations and Airoldi, L. with 154 citations. The co-citation network map (Figure 4b) was divided into 3 clusters, with 271 links and high total link strength.

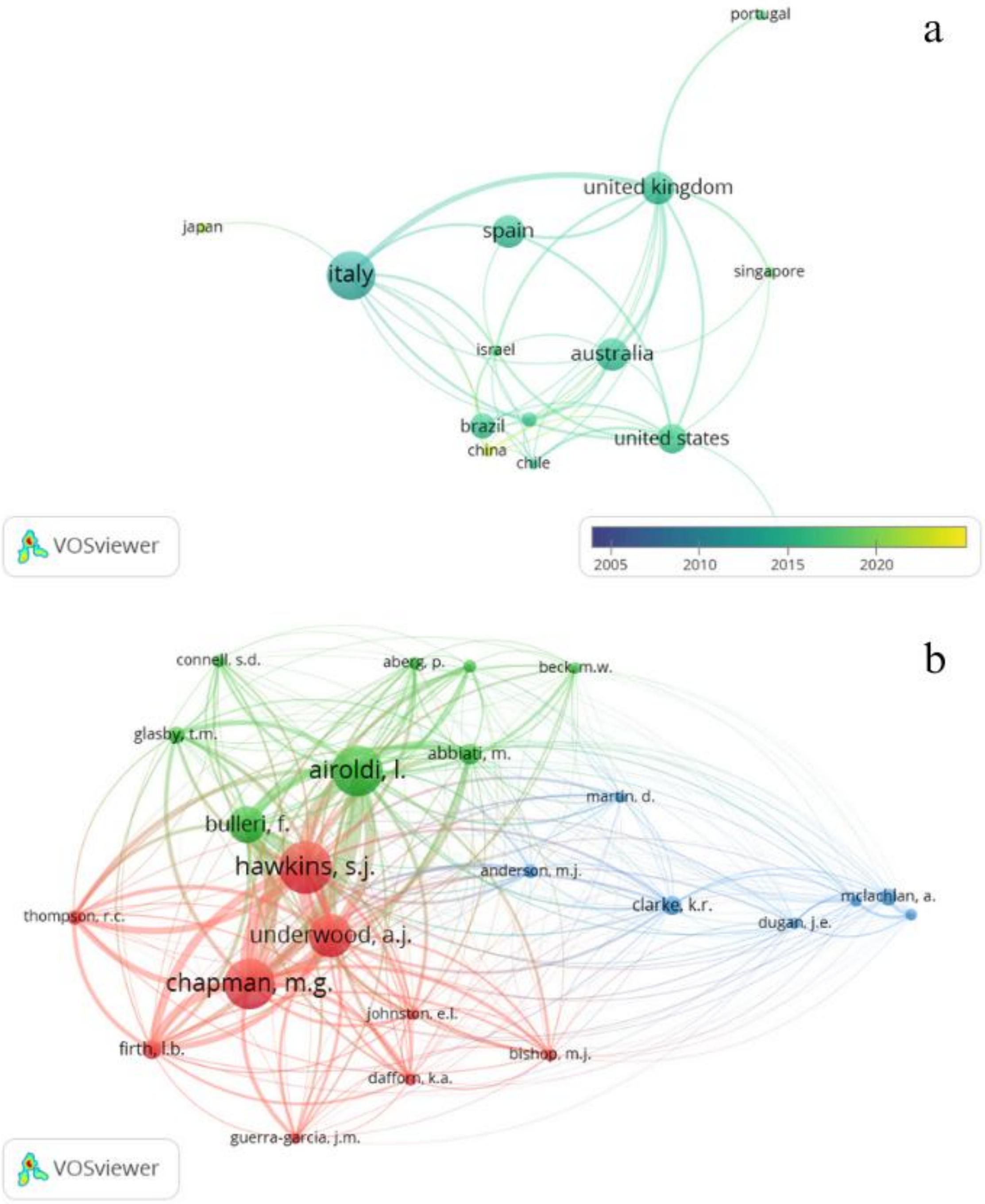

Figure 4. (a) Co-authorship network map of countries and (b) The co-citation network map of authors publishing on the effect of coastal defense structures on benthic macrofauna.

Coastal Defense Structures

Nine different types of coastal defense structures were found in analyzed publications: breakwaters, seawalls, dikes, geosynthetic tubes, cubes, acropolises/tetrapods, ripraps, jetties/groins, and piles of rubble. Most of the documents

(83%) only cited one type of defense structure, while 9 % had two types and three or more types for each. The most common type of defense structure was breakwaters, present in more than 53% of publications (Figure 5a).

In 40% of the documents, the types of materials used in CDS were not specified. This lack of information is quite alarming, as it is widely recognized that the type of material plays an important role in the initial development of macroinvertebrate associations, favoring some species and preventing others from establishing themselves (Andersson *et al.,* 2009). The most common type of material found was concrete, which appeared in 27% of the articles, in addition to contributing to 12% of studies that used concrete and natural rocks at the same time (Figure 5b). The fact that concrete is the most used material in construction is mainly due to its lower price compared to other materials. Furthermore, it is also easily manipulated, allowing the creation of structures that differ in terms of shape and size, and can guarantee the long durability of structures, as it is highly resistant (Fabi *et al.,* 2011). However, artificial structures built with concrete do not adequately represent the benthic community of consolidated substrates, which generally present a reduced richness, abundance, and diversity of organisms (Ido & Shimrit, 2015). The choice of material can also influence the ambient temperature. Aguilera *et al.* (2019) indicated that the temperature in artificial structures was, on average, 3.7º C higher than in the natural environment, influencing the abundance and richness of species. In other words, the use of other materials could be a better option for the construction of these structures, increasing the complexity of the site, favoring the settlement of organisms, and minimizing the impact caused in the area where the structure was built.

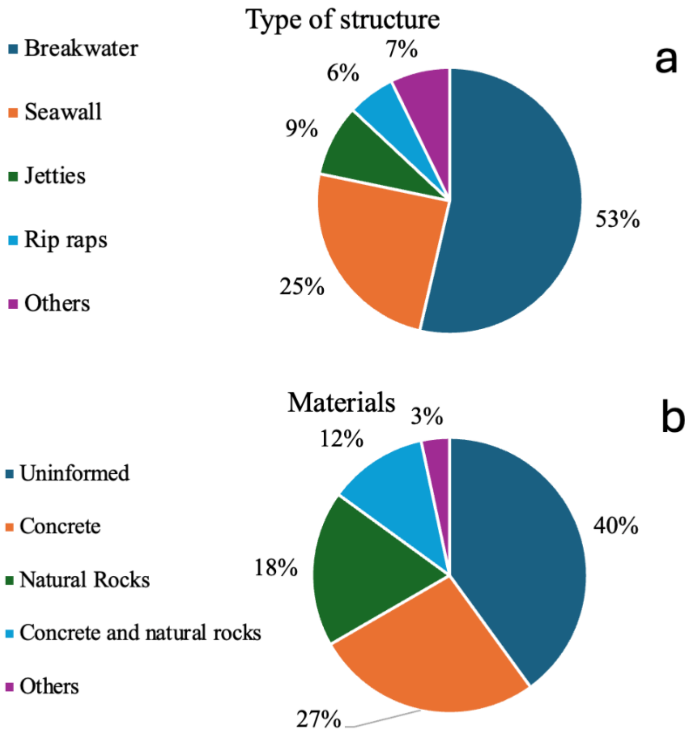

Figure 5. (a) Types of structure and (b) materials from the documents related to the effect of coastal defense structures on benthic macrofauna.

In 36.2% of the articles, the orientation of the structures was not informed. Structures parallel to the coast were indicated in 32.8% of the publications, perpendicular structures appeared in 13.8%, both types of orientation in 15.5% and vertical in only 1.7%. Although most articles did not provide information about structure or orientation, numerous studies have shown that orientation may influence the composition of epibiotic assemblages on surfaces of different types of materials (Connell & Glasby, 1999; Glasby, 2000; Siddik *et al*., 2018). Much of the understanding of the effects of orientation on epibiota is based on the experimental insertion of new artificial substrates, not on CDS themselves (Connell, 2001; Riera *et* al., 2018). These substrates are mainly made by laying panels, whose physicochemical characteristics do not correspond to the usual three-dimensional substrates found in coastal defense structures. In addition to there being less information available on the epibiota that colonizes these urban structures, the amount of time taken for species to become established is also not explored by most authors, especially those who propose to evaluate the same structure in the long term (Manoukian *et al.,* 2011).

Of the methodologies used to evaluate macrofauna in the documents, the most common was collection through observations using photography, videos, visual census and quadrants (40%). Manual collections, using samplers (corer, claw, pitfall and pump) or through scraping were also frequent (34%), while those using both methodologies (collection and photographs) only accounted for 17% of publications. The use of artificial substrates for organism colonization was the least used methodology, only representing 9% of the papers. Notably, this field of research seems to have undergone significant transformation in terms of sampling methodologies, in addition to the tools used, such as the development of new digital photography and scanning technologies, with the reduction in the costs of high-speed resolution cameras and data storage, making this approach more practical. Image-based surveys allow initial samples to be revisited and are non-invasive compared to traditional survey methods that typically involve nets or dredges (Bethoney & Stokesbury, 2018). Such methodologies that do not require organism collection have been widely used and can provide excellent results, in addition to causing the least possible damage to the already impacted environment. Considering the type of substrate sampled, most studies (59%) focused on rigid substrates (defense structures themselves with possible comparisons with natural substrates), and only 38% focused on the impacts on mobile substrates (unconsolidated sediment), signaling a lack of studies aimed at organisms that live in this type of environment. Only 3% of studies were conducted on both types of substrates. Although many studies highlight the impacts that coastal defense structures can have on marine organisms, most of them are focused on species or groups that live in the supralittoral regions of the structures (Sobocinski *et al.,* 2010; Hubbard *et al.,* 2014) or in the benthic community, colonizing the defense structure itself (Martins *et al.,* 2009; Sedano *et al.,* 2020). Studies that focus on communities of unconsolidated substrates, mainly in the sublittoral, are quite scarce, even though these areas and communities are considered the most impacted by the construction of structures (Dugan *et al.,* 2018; Bone *et al.,* 2022).

Due to the extensive impacts that CDS can have on organisms, the idea of building structures that more closely resemble natural structures and play a similar role in organism colonization has been developed, known as eco-engineering (Morris *et al.,* 2018). To minimize the ecological impacts of large constructions, it is necessary to understand how marine communities respond to such changes and the development of construction methodologies that simultaneously consider engineering requirements and costs, as well as environmental and social values (Wang *et al.,* 2014). For these reasons, the concept of eco-engineering has become increasingly relevant and there has been an increase in the number of publications using this approach.

Nevertheless, this new way of constructing CDS, focusing on the importance of ecological issues, was only observed in 16% of the publications analyzed between 2009 and 2024. This information indicates that this is a relatively new concept and that its implementation should be intensified in the coming years, due to the demand for structures that have a lesser impact on the regions in which they are implemented.

Effects of coastal defense structures: a meta-analysis

For the meta-analysis, 16 documents were used. A total of 72 sampling areas were considered, accounting for all collection sites where the studies were carried out.

The effect of defense structures was observed between -3.455 to 1.889 (Table S1). Most studies (68.9%) indicated negative effects (decrease in richness or abundance/density) and (30.1%) indicated positive effects (increase in richness or abundance/density). This analysis suggests that although some studies do not find significant differences that could confirm the impact of defense structures on macrofauna, most studies indicate a strong negative effect of these structures, whether direct or indirect. Notably, when there is a “positive” effect of the defense structure, associated with an increase in richness or abundance/density in the location, this should not be confused with a “good impact”. Using the data in module, without considering the signs, a significant difference was found between the effects on richness compared to abundance/density ($p < 0.0001$) (Figure 6a). The t-test for the effect on richness indicated significant differences between the points analyzed ($t = 7.5351$; $p < 0.0001$), where the 95% confidence interval presents average impacts ranging from 8 to 15%. The same test performed for abundance/density was also significant ($t = 7.2586$; $p < 0.0001$), with a 95% confidence interval indicating average impacts ranging from 8 to 141%. From these results, it is possible to infer that coastal defense structures have a greater average effect size on abundance/density than on richness, although not necessarily more important, since the inclusion or exclusion of a single species can strongly impact ecosystem functions in many ways. The reduction in species richness has a greater impact on ecosystem functions than the decrease in abundance, as it may reflect the loss of functional diversity. Each species plays a unique role, and their loss can cause functional collapses, cascading effects on ecological interactions, and reduce ecosystem resilience.

In addition, species-rich ecosystems offer more stable and diverse services, such as climate regulation and resource provision (Oliver *et al.,* 2015). Although less harmful than the loss of richness, the greatest effect on abundance/density demonstrated in the present paper can be explained mainly by the decrease of sensitive species or the increase in more tolerant or opportunistic organisms. These organisms often have rapid adaptation and fast reproduction, allowing them to successfully occupy the new ecological niches offered (Bumbeer & Rocha, 2016). Thus, invasive species tend to outnumber native species in terms of biomass and occupation/dispersal rate, causing intense changes in local biodiversity (Dimitriadis *et al.,* 2021; Tsirintanis *et al.,* 2023).

Observing Figure 6b, we can see that, despite the large variation, the Seawall seems to have a greater impact on the richness and abundance/density of organisms than the other defense structures. The t-test for the effect on richness and abundance/density in the breakwaters indicated significant differences between the points analyzed ($t = 4.0916$ and $p < 0.0001$), where the 95% confidence interval presents average impacts ranging from 12 to 38%. The same test performed for Seawalls was also significant ($t = 6.3396$ and $p < 0.0001$), with a confidence interval indicating average impacts ranging from 71 to 136%. For Rip raps, the test was also significant ($t = 2.702$ and $p < 0.05$), with confidence interval indicating average impacts ranging from 2% to 141%.

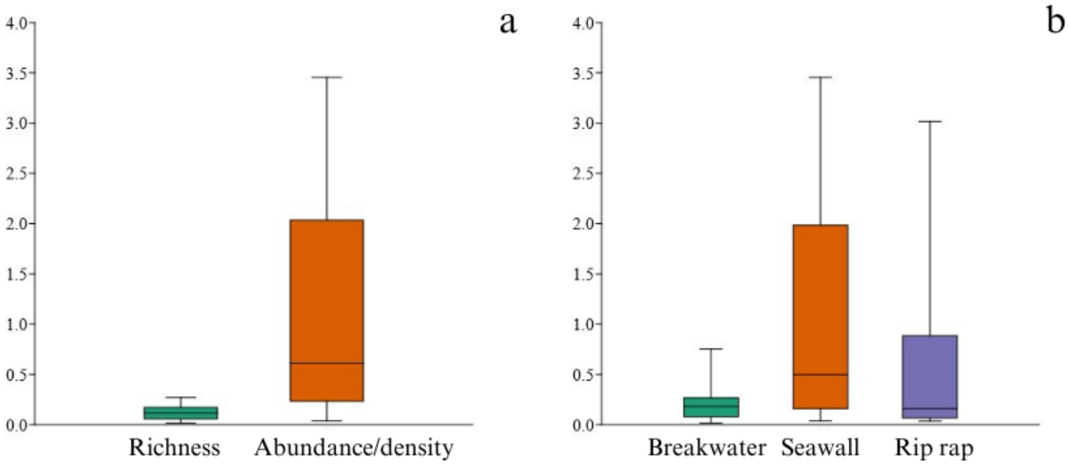

Figure 6. (a) Boxplot using log response ratio (effect/control) in module for community Richness and Abundance/density and (b) boxplot using log response ratio (effect/control) in module for Breakwater, Seawall and Rip raps from the documents related to the effect of coastal defense structures on benthic macrofauna. Groyne was not added to graph (b) because it only contains one observation.

The two-way nested permutational ANOVA showed no difference between structure types (F = 0.297; p = 0.749; unique permutations = 9952) but a significant effect for parameters nested within structures (F = 7.441; p = 0.0003; unique permutations = 9955). The a posteriori test showed that effect size differences between the parameters Richness and Abundance/density were registered for Breakwater (t = 2.965; p = 0.011; unique permutations = 3868) and for Seawall (t = 3.456; p=0.0007; unique permutations = 9819) but not for Rip raps (t = 1.957; p = 0.077; unique permutations = 494). Considering the results of the ANOVA and the assumption of conservation of variance for a theoretical mean of no effect, we calculated the 95% confidence intervals (95%CI) for Abundance/density and Richness which are presented in Figure 7. For Abundance/density, the majority (71.4%) of negative effect size values were outside the 95% CI, while few (25%) occurred for positive effect sizes. This implies that CDS impacts are expected to decrease, around three times, rather than increase the abundance of benthic animals. On the other hand, for Richness there are similar proportions of significant negative (77.8%) or positive (61.5%) effect sizes, indicating that the increase or decrease of taxa are equally expected as an impact of the CDS. Overall, 65.3% of the data indicates significant effects of CDS on the benthic community parameters with most impacts being negative (78.7%), for either Abundance/density or Richness.

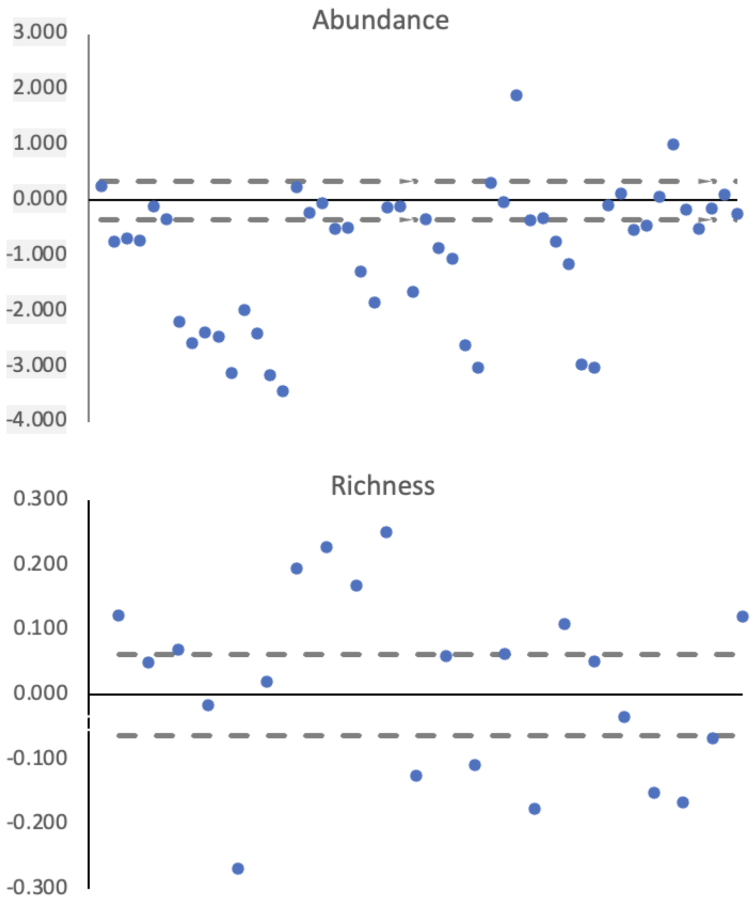

Figure 7. Effect size values for both Abundance/density and Richness. An effect size of zero means no difference, while positive and negative values result from the effect of CDS on the community. Dashed lines indicate the 95% confidence values for a theoretical mean of no effect.

Due to their structural and functional characteristics, seawalls are often associated with significant impacts on coastal environments. They are primarily solid vertical structures, usually made of concrete, although other materials can be used. One of the main points that reinforces the great effect of seawalls on macrofauna is the changes in coastal hydrodynamics caused by these structures. The seawall reflects wave energy, instead of dissipating it, which intensifies erosion at the base of the structure and in adjacent areas. This can lead to the formation of deeper channels, loss of sediment and reduction of intertidal beaches. These effects have altered the availability of habitat for coastal species (Dugan *et al.,* 2011). Seawalls can also create physical barriers that impact macrofauna living in nearby areas, hindering connectivity between shallower and deeper habitats, and potentially interrupting species recruitment and migration processes.

Additionally, due to the low structural complexity of seawalls, they are known to support lower biodiversities of encrusting organisms and mobile epibenthic species than, for example, more complex natural rocky reefs (Chapman & Underwood, 2011).

The effect sizes of breakwater were small especially compared to those of seawalls, and the effects of breakwater structures were more homogeneous. Breakwaters and rip raps have structural similarities since both consist of barriers generally constructed with rock blocks or in some cases stacked concrete. Due to the way they are constructed, they may present more complexity and tend to imitate some of the complexity of natural substrates, through the formation of microhabitats due to their irregular structure, favoring biodiversity (Gittman *et al.,* 2016).

Although many of the articles analyzed do not explicitly mention that CDS caused impacts on macrofauna, it is possible to observe a large effect of these structures on the richness and abundance/density of organisms. We were also able to observe that different types of structures cause different effects, especially concerning the average effects on richness or abundance/density that had large variations between structures.

Another very important point is that some studies indicate that there has been disturbance or change in the community structure in areas with defense structures, but due to the high heterogeneity of the data and the fact that the cases are not treated independently, generally being grouped as a result of human activities, the understanding of the real level of impact caused by these structures is frequently compromised.

Conclusion

This study carried out an analysis of published literature data focusing on the effect of CDS on benthic macrofauna. Combined with this analysis, we used a meta-analysis to calculate the effect size of defense structures on the richness and abundance/density of benthic macrofauna and the differences between the types of coastal defense structures.

An increase in the number of publications has been observed in recent years, in addition to an increase in citations. A significant proportion of articles on coastal defense structures were published in high-impact journals, demonstrating the topic's relevance. The analysis of co-authorship between countries indicated that more developed countries tend to collaborate more with other countries, while less developed countries produce fewer studies, with less collaboration.

This study highlighted many gaps in the publications, especially concerning very important information. For example, most of the articles (40%) do not mention the material used in the construction of the defense structures, just over 1/3 of the papers focused on the effects of the structures on the organisms that live on the unconsolidated substrate, and important abiotic data was lacking in most studies.

The use of the concept of eco-engineering was observed, which, although considered recent, is already present in several more current articles. Concerns regarding the impact caused by coastal defense structures point to a transition trend, however, this update in the construction and renovation of existing structures that aim to mitigate the impacts caused in these environments will take time and require a change in the mentality of coastal managers.

From the meta-analysis, we were able to conclude that 65.3% of the studies present effect sizes indicating significant impacts of defense structures. Most studies indicated negative effects, especially for the Abundance/density parameter. Although some studies indicate an increase in richness and a few in the abundance of benthic animals in areas where defense structures are present, it is important to emphasize that we should not consider this increase a “positive” impact, as this increase is often linked to an increase in opportunistic species. We observed that CDS have a greater effect on abundance/density than on the richness of organisms, although not necessarily more important, since the inclusion or exclusion of a single species can strongly impact ecosystem functions.

The set of synthesized data and information generated in this study provides an important basis for understanding the scope and limitations of previously published scientific research on this topic and should stimulate future studies of higher quality.

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Impact of coastal defense structures on macrobenthos: integrated systematic and bibliometric investigation along with a meta-analysis

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Supplementary Material

Table S1 List of documents used in the Bibliometric Review.

Reference	Title
Jaramillo et al. (2002)	Beach and faunal response to the construction of a seawall in a sandy beach of south central Chile
Chapman and Bulleri (2003)	Intertidal seawalls—new features of landscape in intertidal environments
Bacchiocchi and Airoldi (2003)	Distribution and dynamics of epibiota on hard structures for coastal protection
Moschella et al. (2005)	Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design

Bertasi et al. (2007)	Effects of an artificial protection structure on the sandy shore macrofaunal community: the special case of Lido di Dante (Northern Adriatic Sea)
Keller and Pomory (2008)	Effects of porous mesh groynes on macroinvertebrates of a sandy beach, santa rosa island, florida, U.S.A.
Walker et al. (2008)	Habitat modification in a dynamic environment: the influence of a small artificial groyne on macrofaunal assemblages of a sandy beach
Dugan et al. (2008)	Ecological effects of coastal armoring on sandy beaches
Masi and Zalmon (2008)	Zonation of intertidal benthic communities on breakwaters of different hydrodynamics in the north coast of the state of Rio de Janeiro, Brazil
Bertasi et al. (2009)	Comparing efficacy of different taxonomic resolutions and surrogates in detecting changes in soft bottom assemblages due to coastal defence structures
Martins et al. (2009)	Influence of a breakwater on nearby rocky intertidal community structure
Masi et al. (2009)	Benthic community zonation in a breakwater on the north coast of the state of Rio de Janeiro, Brazil
Burt et al. (2009)	Coral recruitment and early benthic community development on several materials used in the construction of artificial reefs and breakwaters
Fanini et al. (2009)	Effects of beach nourishment and groynes building on population and community descriptors of mobile arthropodofauna
Moreira and Troncoso (2010)	Temporal dynamics of the benthic assemblage in the muddy sediments of the harbour of Baiona (Galicia, NW Iberian Peninsula)
Burt et al. (2011)	Benthic development on large-scale engineered reefs: a comparison of communities among breakwaters of different age and natural reefs
Munari et al. (2011)	Coastal defence through low crested breakwater structures: Jumping out of the frying pan into the fire?
Shih et al. (2011)	Eco-environmental impact assessment of pre-leisure beach nourishment on the benthic invertebrate community at Anping coast

Masi and Zalmon (2012)	Intra-annual variation of intertidal benthic community in a breakwater zone on the north coast of Rio de Janeiro, Brazil
Hajisamae et al. (2013)	Can Wave Breaking Walls in Shallow Coastal Areas Serve as Habitat for Aquatic Organisms?
Wen et al. (2013)	High coral cover and subsequent high fish richness on mature breakwaters in Taiwan
Cha et al. (2013)	Comparative study on the fauna composition of intertidal invertebrates between natural and artificial substrata in the northeastern coast of Jeju Island
Munari (2013)	Benthic community and biological trait composition in respect to artificial coastal defence structures: a study case in the northern Adriatic Sea
Becchi et al. (2014)	The effect of breakwaters on the structure of marine soft-bottom assemblages: A case study from a North-Western Mediterranean basin
Kumar et. (2014)	Preliminary studies on nature of epibiota assemblage on low crested coastal protection structures
Rolet et al. (2015)	Anthropogenic impact on macrobenthic communities and consequences for shorebirds in Northern France: A complex response
Turon et al. (2015)	Corridors for aliens but not for natives: effects of marine urban sprawl at a regional scale
Burgos-Rubio et al. (2015)	The role of patellid limpets as omnivorous grazers: a new insight into intertidal ecology
Ido and Shimrit (2015)	Blue is the new green–ecological enhancement of concrete based coastal and marine infrastructure
Bauman et al. (2016)	Prickly business: abundance of sea urchins on breakwaters and coral reefs in Dubai
Oricchio et al. (2016)	Distinct community dynamics at two artificial habitats in a recreational marina
Ho et al. (2017)	Wall of orange cup coral, *Tubastraea coccinea*, at the inlet breakwaters of a nuclear power plant, southern Taiwan
Grizzle et al. (2018)	Marine molluscs in nearshore habitats of the United Arab Emirates: Decadal changes and species of public health significance
Morris et al. (2018)	Can transplanting enhance mobile marine invertebrates in ecologically engineered rock pools?
Heery and Sebens (2018)	Artificial structures as a source of elevated detrital loads for sedimentary environments

Carugati et al. (2018)	Impact of breakwater relocation on benthic biodiversity associated with seagrass meadows of northern Adriatic Sea
Cacabelos et al. (2019)	Patchiness in habitat distribution can enhance biological diversity of coastal engineering structures
Critchley and Bishop (2019)	Differences in soft-sediment infaunal communities between shorelines with and without seawalls
Sedano et al. (2019)	Comparing sessile benthos on shallow artificial versus natural hard substrates in the Eastern Mediterranean Sea
Chowdhury et al. (2020)	Do oyster breakwater reefs facilitate benthic and fish fauna in a dynamic subtropical environment?
Kim et al. (2020)	Influence of dike-induced morphologic and sedimentologic changes on the benthic ecosystem in the sheltered tidal flats, Saemangeum area, west coast of Korea
Masucci et al. (2020)	Eroding diversity away: impacts of a tetrapod breakwater on a subtropical coral reef
Sedano et al. (2020)	From sessile to vagile: understanding the importance of epifauna to assess the environmental impacts of coastal defence structures
Sedano et al. (2020)	Understanding the effects of coastal defence structures on marine biota: The role of substrate composition and roughness in structuring sessile, macro-and meiofaunal communities
Heery et al. (2020)	Human-engineered hydrodynamic regimes as a driver of cryptic microinvertebrate assemblages on urban artificial shorelines
Tanasovici et al. (2020)	Invasive coral *Tubastraea* spp. population growth in artificial habitats and its consequences to the diversity of benthic organisms
Mamo et al. (2021)	Upgrades of coastal protective infrastructure affect benthic communities
Strain et al. (2021)	A global analysis of complexity–biodiversity relationships on marine artificial structures
Masucci et al. (2021)	Impacts of coastal armouring on rubble mobile cryptofauna at shallow coral reefs in Okinawa, Japan

Stender et al. (2021)	Evaluating the feasibility and advantage of a multi-purpose submerged breakwater for harbor protection and benthic habitat enhancement at Kahului Commercial Harbor, Hawai ‘i: case study
Jaramillo et al. (2021)	Ranking the ecological effects of coastal armoring on mobile macroinvertebrates across intertidal zones on sandy beaches
Sedano et al. (2021)	Coastal armouring affects intertidal biodiversity across the Alboran Sea (Western Mediterranean Sea)
Momota and Hosokawa (2021)	Potential impacts of marine urbanization on benthic macrofaunal diversity
Tang et al. (2022)	Influence of coastal engineering on the intertidal macrobenthic community in the Dongtou Islands, China
Sun et al. (2022)	Microhabitat thermal environment controls community structure of macrobenthos on coastal infrastructures
Laurino et al. (2022)	Does coastal armoring affect biodiversity and its functional composition on sandy beaches?
Di Camillo et al. (2022)	Characterization of intertidal macrofaunal communities of two sandy beaches under different anthropogenic pressures
Kaffenberger et al. (2024)	Intertidal assemblages on groynes along sandy shores in Portugal: Exploring the effects of orientation and distance to rocky shore

Table S2 Articles used in the meta-analysis, with information on the type of structure, the parameter used, the effect and the percentage of the effect.

	Reference	Type of strucuture	Parameter	Effect	%Effect
1	Di Camillo et al 2022	Breakwater	Richness	0,120	32
2	Laurino et al 2022	Seawall	Abundance	-0,352	-56
3	Tang et al 2022	Seawall	Richness	-0,125	-25
4	Jaramillo et al 2021	Seawall	Abundance	-2,190	-99
4	Jaramillo et al 2021	Seawall	Abundance	-2,576	-100
4	Jaramillo et al 2021	Seawall	Abundance	-2,385	-100
4	Jaramillo et al 2021	Seawall	Abundance	-2,468	-100
4	Jaramillo et al 2021	Seawall	Abundance	-3,112	-100
4	Jaramillo et al 2021	Rip rap	Abundance	-2,963	-100
4	Jaramillo et al 2021	Rip rap	Abundance	-3,016	-100
4	Jaramillo et al 2021	Seawall	Abundance	-1,985	-99
4	Jaramillo et al 2021	Seawall	Abundance	-2,402	-100
4	Jaramillo et al 2021	Seawall	Abundance	-3,152	-100
4	Jaramillo et al 2021	Seawall	Abundance	-3,455	-100
4	Jaramillo et al 2021	Seawall	Abundance	0,235	72
4	Jaramillo et al 2021	Seawall	Abundance	-0,228	-41
4	Jaramillo et al 2021	Seawall	Abundance	-0,062	-13
4	Jaramillo et al 2021	Seawall	Abundance	-0,509	-69
4	Jaramillo et al 2021	Seawall	Abundance	-0,499	-68
4	Jaramillo et al 2021	Seawall	Abundance	-1,282	-95
4	Jaramillo et al 2021	Rip rap	Abundance	-0,088	-18
4	Jaramillo et al 2021	Rip rap	Abundance	0,111	29
4	Jaramillo et al 2021	Seawall	Abundance	-1,841	-99
4	Jaramillo et al 2021	Seawall	Abundance	-0,131	-26
4	Jaramillo et al 2021	Seawall	Abundance	-0,122	-25
4	Jaramillo et al 2021	Seawall	Abundance	-1,646	-98
4	Jaramillo et al 2021	Seawall	Abundance	-0,335	-54
4	Jaramillo et al 2021	Seawall	Abundance	-0,868	-86
4	Jaramillo et al 2021	Seawall	Abundance	-1,062	-91
4	Jaramillo et al 2021	Seawall	Abundance	-2,618	-100
4	Jaramillo et al 2021	Seawall	Abundance	-3,032	-100
4	Jaramillo et al 2021	Rip rap	Abundance	-0,538	-71
4	Jaramillo et al 2021	Rip rap	Abundance	-0,451	-65
4	Jaramillo et al 2021	Seawall	Abundance	0,320	109
4	Jaramillo et al 2021	Seawall	Abundance	-0,036	-8
4	Jaramillo et al 2021	Seawall	Abundance	1,889	7640
4	Jaramillo et al 2021	Seawall	Abundance	-0,359	-56
4	Jaramillo et al 2021	Seawall	Abundance	-0,333	-54
4	Jaramillo et al 2021	Rip rap	Abundance	0,060	15

4	Jaramillo et al 2021	Rip rap	Abundance	0,999	897
5	Chowdhury et al 2020	Breakwater	Abundance	0,262	83
6	Critchley e Bishop2019	Seawall	Richness	0,059	15
7	Rolet et al 2015	Seawall	Richness	-0,108	-22
7	Rolet et al 2015	Seawall	Density	-0,178	-34
8	Becchi et al 2014	Breakwater	Richness	0,122	32
8	Becchi et al 2014	Breakwater	Richness	0,050	12
8	Becchi et al 2014	Breakwater	Richness	0,069	17
8	Becchi et al 2014	Breakwater	Richness	-0,017	-4
9	Hajisamae et al 2013	Breakwater	Abundance	-0,751	-82
9	Hajisamae et al 2013	Breakwater	Richness	-0,269	-46
10	Munari et al 2011	Breakwater	Richness	0,020	5
10	Munari et al 2011	Breakwater	Richness	0,195	57
11	Lucrezi et al 2010	Seawall	Density	-0,525	-70
11	Lucrezi et al 2010	Seawall	Density	-0,158	-30
11	Lucrezi et al 2010	Seawall	Density	0,103	27
11	Lucrezi et al 2010	Seawall	Density	-0,248	-43
12	Sobocinski et al 2010	Rip rap and Bulkheads	Richness	0,051	13
12	Sobocinski et al 2010	Rip rap and Bulkheads	Richness	-0,034	-8
12	Sobocinski et al 2010	Rip rap and Bulkheads	Richness	-0,151	-29
12	Sobocinski et al 2010	Rip rap and Bulkheads	Richness	-0,166	-32
13	Walker et al 2008	Groyne	Richness	-0,067	-14
14	Dugan et al 2008	Seawall	Richness	0,062	15
14	Dugan et al 2008	Seawall	Abundance	-0,746	-82
14	Dugan et al 2008	Seawall	Richness	-0,176	-33
14	Dugan et al 2008	Seawall	Abundance	-1,155	-93
15	Bertasi et al 2007	Breakwater	Richness	0,227	69
15	Bertasi et al 2007	Breakwater	Richness	0,168	47
15	Bertasi et al 2007	Breakwater	Richness	0,250	78
15	Bertasi et al 2007	Breakwater	Abundance	-0,689	-80
15	Bertasi et al 2007	Breakwater	Abundance	-0,728	-81
15	Bertasi et al 2007	Breakwater	Abundance	-0,108	-22
16	Jaramillo et al 2002	Seawall	Richness	0,109	29

5 CAPÍTULO 2 - Macrobenthic communities in coastal environments: artificial defense structures, natural reefs and exposed beaches

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[†] in memoriam

Abstract

Artificial coastal defense structures globally reshape shorelines, yet their ecological impacts remain poorly quantified compared to natural defenses or exposed natural habitats. This study assessed benthic community structure responses to artificial (breakwaters, seawall), naturally protected (reefs), and exposed beach environments along the coast of Pernambuco (NE Brazil). Environmental variables and macrofauna were analyzed across nine areas. The areas with the presence of artificial structures (Breakwaters 1 to 3 and Seawall/Reef) retained more fine sediments than the natural areas (Reefs 1 and 2 and Exposed beaches 1 and 2) which were sandier. Areas with artificial structures favored opportunistic taxa, indicating high densities but reduced ecological quality (AMBI index: 1.6-3.9), inversely, exposed beaches and reef protected areas showed higher ecological quality (AMBI: 0.2-1.8) despite lower abundance/richness resulting from the difficulty of fixing and retaining organisms in environments with higher hydrodynamic energy. Strong negative correlation linked ORP to AMBI, validating ORP as a robust environmental predictor. Bathing suitability and the BOPA indices showed weak/nonsignificant relationships with AMBI. Indicator species analysis (IndVal) revealed breakwater-specific taxa reflecting microhabitat specialization driven by homogenizing structural features. No indicators emerged in natural areas, indicating higher community variability. Coastal defense structures simplify benthic communities toward disturbance-adapted assemblages, while natural areas maintain higher ecological variability.

Keywords: Breakwaters; Seawalls; Benthic indicators; Sandy coasts; Anthropogenic impacts.

Introduction

Sandy beaches are environments that show high variability throughout tropical and temperate regions (MCLACHLAN & BROWN, 2006). They are the most visited natural environments in the world, due to the supply of goods and services, such as fishing resources and tourism and leisure activities, and, consequently, they have a high human population density (SCHLACHER et al., 2007). Studies indicate that approximately 40% of the world's population and 60% of the population of the world's largest cities are located within 100 km of the coast (TIBBETTS, 2002).

Urban expansion in coastal areas has been identified as one of the biggest threats to coastal areas (FIRTH et al., 2013), habitat for a wide range of biota that perform essential ecosystem services for good environmental functioning (DEFEO et al., 2009). Coastlines are sensitive regions that have been suffering from rising sea levels in response to continuous global warming and climate change (CAZENAVE & COZANNET, 2014; MCLACHLAN & DEFEO, 2018; DURAND et al., 2022).

With the latest sea level estimates (IPCC AR6, 2023), it is predicted that the acceleration of sea level rise will further intensify maritime erosion and marine flooding (NICHOLLS & CAZENAVE, 2010). As a result, changes in coastal landscapes are expected to worsen, increasing the vulnerability of these areas around the globe, especially in densely populated regions (DURAND et al., 2022).

One of the more frequent measures adopted to reduce and remedy the damage caused by the advance of the sea is the installation of coastal defense structures. In many cases, several artificial containment structures, such as bagwalls, jetties, dikes, and breakwaters, have been constructed without a full understanding of the local coastal dynamics (MALLMAN & PEREIRA, 2014). Although these structures can support different communities, through the availability of consolidated habitats, allowing occupation by benthic organisms, and the provision of refuge and nutrients (BISHOP et al., 2017), it is known that these structures differ from natural substrates in terms of physical-chemical characteristics, in addition to being more susceptible to biotic invasion (FIRTH et al., 2013) and potentially promoting contamination and changes in ecological communities (DAFFORN et al., 2015). However, there is a lack of studies that directly compare the effects of these artificial structures with environments protected by natural reefs and fully exposed areas, limiting the understanding of how different forms of coastal protection influence benthic communities.

Macrobenthic organisms that inhabit coastal regions play an important role in these ecosystems, actively participating in animal biomass production and the cycling of seabed nutrients (MCLACHLAN, 1983). Their presence is closely linked to beach hydrodynamics and biotic factors, including the availability of resources for settlement (DEFEO & ALAVA, 1995; MCLACHLAN, 1996).

The soft bottom benthic community is composed of a high diversity of biological groups (e.g., mollusks, polychaetes, crustaceans) that are considered excellent biological indicators of pollution and disturbances as they present a variety of functional and indicative characteristics, such as body size and shape, mobility, position in the sediment, lifespan, sensitivity, tolerance and feeding strategy (BORJA et al., 2000; PILÓ et al., 2016; MOSBAHI et al., 2019; DONG et al., 2021). Due to their biological and ecological characteristics, these species are widely incorporated into environmental monitoring work, to estimate biotic indices that seek to assess the state of ecological quality (RIERA et al., 2013; BORJA et al., 2015) resulting in valuable information about the consequences of anthropogenic pressures.

Therefore, this study aims to compare the structure of benthic soft bottom macrofauna communities in three distinct scenarios: (1) areas with coastal defense structures, (2) areas protected by natural reefs, and (3) exposed beaches without protection. In addition, we aim to estimate biotic indices and to identify indicator species and key environmental variables associated with each environmental scenario.

Material and methods

Study area and sampling design

The study was conducted on the beaches of Janga and Nossa Senhora do Ó, located in the city of Paulista, Casa Caiada in the city of Olinda, and Serrambi and Merepe in the city of Ipojuca, all in the state of Pernambuco, northeastern Brazil (Fig. 1). The Seawall (Fig. 1A), located in Nossa Senhora do Ó, is a coastal defense structure made of geotextile material filled with concrete, installed parallel to the coastline. Additionally, natural reef formations are present near the Seawall. Breakwaters 1 and 2 (Fig. 1B), located at Janga Beach, are constructed of granite blocks arranged parallel to the shoreline and present differences in sediment composition, with muddy sediments prevailing as a result of water retention, particularly at Breakwater 1. Breakwaters 3 and 4 (Fig. 1C), located at Casa Caiada Beach, are also constructed of granite blocks arranged parallel to the shoreline. Furthermore, a nearby sandstone reef is partially covered by fine, muddy sediments, likely influenced by sediment accumulation around coastal defense structures. Exposed beaches (i.e., without natural reef protection) and beaches influenced by nearby reef formations were sampled at both Merepe (Fig. 1D) and Serrambi (Fig. 1E).

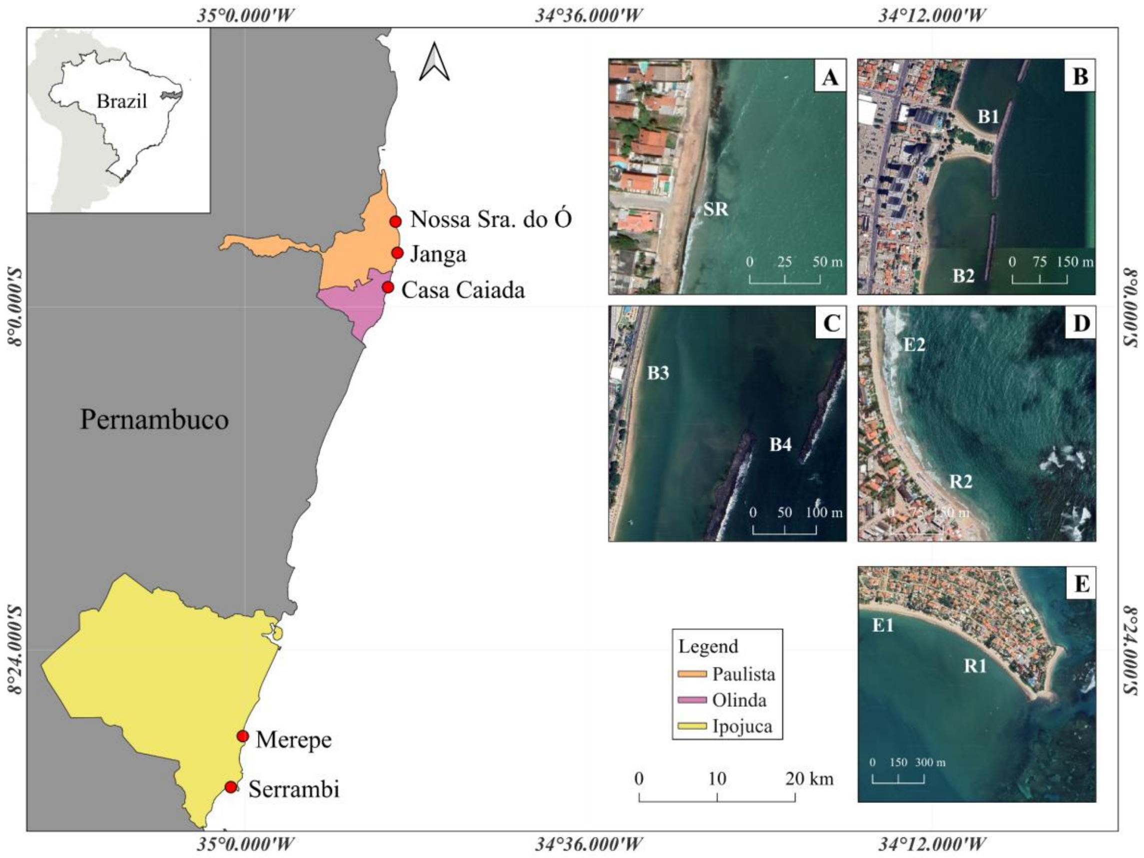

Figure 1. Location of the study area on the beaches of (A) Nossa Senhora do Ó, (B) Janga, (C) Casa Caiada, (D) Merepe and (E) Serrambi. SR: Seawall/Reef; B1: Breakwater 1; B2: Breakwater 2; B3: Breakwater 3; B4: Breakwater 4; E1: Exposed beach 1; R1: Reef 1; E2: Exposed beach 2 and R2: Reef 2.

To investigate the environmental variables, water temperature measurements and subtidal water collections were carried out (3 replicates in each area), and a refractometer was used to measure salinity. Sediment samples were collected for granulometric analyses. Measurements of the Oxidation-Reduction Potential (ORP) of the interstitial water of the samples collected in the study areas were also made using a silver electrode. A value of +200 mV was added to correct the ORP values, as suggested by Jardim (2014).

The bulletins published weekly by CPHR (Pernambuco State Environmental Agency) with information on the bathing water quality of beaches were also used. The 45 bulletins published in 2023 were used to create an average of the bathing suitability (seawater quality) for each of the beaches, considering 0 when the beach was suitable for bathing and 1 when it was unsuitable; this average value is herein considered an index of bathing suitability. Average values approaching 1 indicate lower bathing suitability and values closer to 0 indicate better bathing suitability or seawater quality.

To collect biological material, two transects were visually drawn parallel to the coastline (10 m apart) in each area, where 5 replicates were collected in each transect at a depth of approximately 60 cm. Samples were collected using a cylindrical PVC sampler with H = 10 cm and Ø = 15 cm, corresponding to a total area of 0.0177 m^2. The sampled sediment was sieved in the field through a 0.5 mm nylon screen, and the retained material was stored in plastic bags and pots, duly labeled and fixed in 4% saline formalin. The samples were subsequently washed in running water and the macrofauna was sorted and identified at the level of large groups, using a magnifying glass and stereomicroscope.

Data analyses

To assess the differences in taxonomic community structure, the abundance data were transformed to Log(x+1), then a similarity matrix was generated using Bray-Curtis similarity index. Non-metric Multidimensional Scaling (nMDS) was used to represent, in a two-dimensional model, the similarity among replicates based on taxa abundances. PERMANOVA (Permutational Multivariate Analysis of Variance) (ANDERSON, 2001) compared the differences between areas concerning their similarities. As PERMANOVA showed significant results, pairwise comparisons were performed a posteriori (pairwise permutational t-test). To determine the most important taxa in terms of contribution (%) to the dissimilarity between areas, the similarity percentage analysis (SIMPER) was applied (CLARKE, 1993). The IndVal (DUFRENE & LEGENDRE, 1997) routine (Indicator species analysis) was applied using PAST5.2 software to identify the indicator species/taxa for each area, considering the index and p(raw) values. Species/taxa were considered indicators when IndVal coefficients were significant and ≥40 (KUBOSOVA et al., 2010).

The AZTI Marine Biotic Index (AMBI) was also used to analyze the macrofauna, classifying organisms into five ecological groups (EG) according to their sensitivity to organic enrichment. Species in EG 1 are sensitive to pollution and are indicators of undisturbed environments; EG 2 includes species that are neutral to organic enrichment, generally with low densities and insignificant fluctuations; EG 3 includes species that are tolerant to increased organic matter; and EG 4 and 5 correspond, respectively, to second- and first-order opportunistic species. The assignment to these groups strictly follows the reference list of the AMBI v.5.0 program (http://www.azti.es), and for organisms identified at taxonomic levels higher than the species (such as family), a specific evaluation was carried out to define their classification based on the known ecological characteristics of the taxon. The calculation of the AMBI value, according to the methodology of Borja et al. (2004) and Bigot et al. (2008), considers the relative percentages of each group in the sample, generating a continuous index that ranges from 0 (undisturbed environment) to 6 (extremely disturbed).

The adapted version of the BOPA index, as proposed by Gesteira & Dauvin (2000) and consolidated by Dauvin & Ruellet (2007), was also applied. In this methodology, the index values are calculated using the ratio between the number of Annelida and Crustacea species, with the addition of +1, an essential mathematical adjustment to enable application in areas where crustaceans were completely absent (cases of extreme degradation). This adaptation increases the usefulness of the index by replacing the original taxonomic groups (Polychaeta and Amphipoda) with more comprehensive categories (Annelida and Crustacea), while maintaining the central logic of the indicator: the proportion between sensitive (crustaceans) and tolerant (annelids) groups in relation to the ecological quality of the area.

To evaluate the relationship between macrofauna community structure and environmental variables, a distance-based linear model (DistLM) was generated. The applied model used the sequential stepwise selection procedure with the Akaike selection criterion (AIC). Exploratory Pearson correlations with Bonferroni correction were performed to evaluate the covariation of the ecological univariate indices of community attributes [AZTI and BOPA indices, richness, Shannon–Wiener diversity ($\log_e$), evenness, and density] and the environmental variables.

The analyses were performed using the PRIMER v.6 - PERMANOVA+ and PAST5.2 software. The significance level adopted for all statistical analyses was 5%.

RESULTS

Environmental variables

The environmental variables showed variations between areas and are presented in table 1. Salinity and temperature showed small variation between areas, however, ORP, sediment texture and bathing suitability index showed greater variation.

Table 1. Average salinity, temperature, oxidation-reduction potential (ORP), sediment texture (according to Folk, 1954) and bathing suitability index values of the sampled areas.

	Salinity	Temperature	ORP	Sediment texture	Bathing suitability
Breakwater 1	34.8	31.9°C	-20	Muddy sand	0.666
Breakwater 2	30.5	31.1°C	120	Muddy sand	0.666
Breakwater 3	30.3	30.3°C	83	Muddy sand	0.666
Breakwater 4	30.8	30°C	83	Sand	0.666
Seawall/Reef	34.1	30.2°C	190	Muddy sand	0.577
Reef 1	35.6	30.2°C	160	Sand	0.155
Reef 2	36.6	28.5ºC	85	Sand	0.088
Exposed beach 1	35.6	30ºC	217	Sand	0.155
Exposed beach 2	35.5	28.2ºC	316	Sand	0.088

Ecological Univariate Indices

Breakwaters 1, 2, 3, and the Seawall/Reef structure presented moderate and relatively homogeneous H' values. These values indicate a consistent, but not high, level of taxon diversity among these artificial structures. J' values showed greater variability within this group. Breakwaters 1, 2, 3, and the Seawall/Reef structure presented lower J' than the natural areas, suggesting a less balanced distribution of abundance among the taxa present at these sites. In contrast, Breakwater 4 exhibited an exceptionally low J' value (0.14), indicating a community strongly dominated by a few species.

Breakwater 4 also stood out not only for its extremely low J', but also for its extremely low H' (0.05) and extremely high individual density (ind/m²) (62,412). This points to an extremely impoverished community dominated by a single taxon. The natural protected reefs areas (Reef 1-2) and the exposed beaches (Exposed beach 1-2) presented H' values generally lower than those of the artificial structures (except for Breakwater 4) and quite low species richness.

Among the areas with artificial structures, Breakwater 1 had the worst result (AMBI: 3.9), classified as moderately disturbed. This value, close to the upper limit of the category (3.3-5.0), suggests a significant impact. In contrast, Breakwater 2 (AMBI: 1.61) and Breakwater 3 (AMBI: 2.11) presented minimally disturbed conditions. Breakwater 4 (AMBI: 2.97), also highly disturbed, is a concern because it is close to the limits for the moderate category (3.3). The combined Seawall/Reef area (AMBI: 2.31) remained partially disturbed. Among the natural areas, Reef 1 (AMBI: 1.67) and Reef 2 (AMBI: 1.77) were classified as having minimal disturbance. The exposed beaches followed a similar trend, although with nuances: Exposed beach 1 (AMBI: 1.8) had minimal disturbance, while Exposed beach 2 (AMBI: 0.2) stood out as the only undisturbed area in the study.

Regarding BOPA, in areas with artificial structures, the results show significant variations. Breakwater 4 (BOPA: 3.34) stands out as impacted. In contrast, the other breakwaters presented less critical conditions: Breakwater 1 (BOPA: 1.36), Breakwater 2 (BOPA: 1.23), and Breakwater 3 (BOPA: 1.12) are in the slightly impacted range, although below the highest risk threshold (2.0). The Seawall/Reef (1.10) is also slightly impacted. In natural areas, healthier environmental conditions prevail. Reef 1 (BOPA: -0.07) and Reef 2 (BOPA: 0.64) present the best results, classified as unimpacted (BOPA < 1.0). Exposed beaches follow the same trend: Exposed beach 1 (BOPA: 0.78) and Exposed beach 2 (BOPA: 0.48) also remain unimpacted, with values below 1.0 (Table 2).

Pearson correlations with Bonferroni correction confirmed specific relationships between ecological indices and environmental variables. First, a highly significant correlation was observed between taxon richness and bathing suitability index ($r = 0.923$;

p = 0.006). Second, a significant and inverse correlation was identified between the AMBI and ORP (r = -0.860; p = 0.044). The negative sign reveals that high ORP values are associated with lower AMBI values. The remaining correlations were not significant.

Table 2. Values of taxon richness (S), density (ind/m²), Pielou evenness (J'), Shannon-Wiener diversity (H' $\log_e$), AMBI and BOPA indices.

	S	ind/m²	J'	H' ($\log_e$)	AMBI	BOPA
Breakwater 1	30	2130	0.68	2.30	3.9	1.36
Breakwater 2	29	2169	0.75	2.54	1.61	1.23
Breakwater 3	38	2040	0.77	2.80	2.11	1.12
Breakwater 4	19	62412	0.05	0.14	2.97	3.34
Seawall/Reef	28	1825	0.68	2.25	2.31	1.10
Reef 1	8	124	0.83	1.72	1.77	0.64
Reef 2	6	85	0.72	1.30	1.8	0.78
Exposed beach 1	9	96	0.91	2.00	1.67	-0.07
Exposed beach 2	3	28	0.96	1.05	0.2	0.48

The IndVal coefficient revealed 32 significant indicator species/taxa among all areas (Table 3). Based on the proposed IndVal scale, only taxa with high indicator values (> 40%) are considered; thus, only 8 taxa were considered indicators. For Breakwater 1, Capitellidae was considered an indicator taxon, for Breakwater 2 it was considered Ophelidae, for Breakwater 3 it was considered Magelonidae, Cirratulidae, Sipuncula and *Upogebia omissa* Gomes Correa, 1968, and for Breakwater 4 it was considered Spionidae and *Vitta virginea* (Linnaeus, 1758). For Seawall/Reef, Reefs 1 and 2 and Exposed Beaches 1 and 2, no taxon was indicated as an indicator. All taxa considered indicators were different for each area.

Table 3. Indval results per area, including all taxa that had a significant p(raw) value (>0.05). Indicator species/taxa in bold. *- denotes values that, although significant, were below 40. B=Breakwater,SR=Seawall/Reef, EB=Exposed beach and R=Reef.

	B 1	B 2	B 3	B 4	S/R	EB1	R 1	R 2	EB 2
Capitellidae	**52.2**	-	22.5*	-	-	-	-	-	-
Spionidae	-	-	-	**79.6**	-	-	-	-	-
Magelonidae	-	21.39*	**61.3**		-	-	-	-	-
Cirratulidae	-	-	**45**	-	20.6*	-	-	-	-
Opheliidae	-	**95.51**	-	-	-	-	-	-	-
Glyceridae	-	-	-	-	-	16.6*	10.6*		
Goniadidae	-	15*	19.2*	-	16*	-	-	-	-
Hesionidade	-	-	-	11.43*	-	-	-	-	-
Polynoidae	-	-	11.6*	-	38.3*	-	-	-	-
Orbiniidae	-	-	18.8*	11.4*	-	-	-	-	-
Paraonidae	-	-	-	21.2	10*	-	-	-	-
Onuphidae	-	-	12*	-	-	-	-	-	-
Dorvilleidae	-	-	-	30*	-	-	-	-	-
Oligochaeta	20*	-	-	-	-	-	-	-	-
Atylidae	18*	-	-	-	-	-	-	-	-
Cheiriphotis	14.2*	-	-	-	-	-	-	-	-
Comesoma	-	19.5*	-	-	-	-	-	-	-
Linhomoeus	-	6.6*	-	-	-	-	-	-	-
Oncholaimus	28.7*	-	-	-	-	-	-	-	-
Sabatieria	13.3*	-	-	-	-	-	-	-	-
Theristus	30*	-	-	-	-	-	-	-	-
Anomalocardia flexuosa	-	33.1*	-	-	-	-	-	-	-
Vitta virginea	-		-	**85.7**	-	-	-	-	-
Astyris lunata	-	37.8*	-	-	33.1*	-	-	-	-
Paguroidea	-	21.8*	-	-	-	-	-	-	-
Ophiuroidea	-	-	-	-	15*	-	-	-	-
Sipuncula	-	-	**44.4**	-	21.5*	-	-	-	-
Pycnogonida	20*	-		-	-	-	-	-	-
Nemertea	-	16*		-	-	-	-	-	-
Cirripedia	-	-	30*	-	-	-	-	-	-
Upogebia omissa	-	-	**40**	-	-	-	-	-	-
"Turbellaria"	-	-	-	16.8*	-	-	-	-	-

Macrofauna Community Structure

The benthic macrofauna comprised a total of 70 taxa in all study areas, with the greatest richness (Table 2) of taxa found in Breakwater 3 (38), followed by Breakwater 1

(30), and the lowest found in Exposed beach 2 (3) and Reef 2 (6). Regarding density (Table 2), Breakwater 4 had the highest number (62,412 ind/m²), followed by Breakwater 2 (2,169 ind/m²), while the lowest densities were found in Exposed beach 2 (28 ind/m²) and Reef 2 (85 ind/m²). Regarding the most abundant taxa, Spionidae was the most common among all areas, followed by Capitellidae and the nematode *Oncholaimellus* de Man, 1890. Analyzing separately by area, the most abundant taxon was Spionidae, reaching a relative abundance of 97.9% in Breakwater 4.

The nMDS demonstrates a clear separation between the areas with the presence of artificial coastal defense structures (Breakwaters 1- 4 and Seawall/Reef) on the left of the graph and the natural areas (Reef 1-2 and Exposed beach 1-2) on the right (Fig. 2). Significant differences were found using PERMANOVA for the factor area (Pseudo-F = 13. 11; p = 0.0001), and when applying the a posteriori test, significant differences were found between almost all areas ($p<0.001$), except between Exposed beaches 1 and 2 (t = 1.6243; p = 0.0508) and between Reefs 1 and 2 (t = 1.2708; p = 0.1697). The application of DistLM revealed that bathing suitability index and ORP together explained 53.12% ($R^2 = 0.53122$) of the variation in benthic macrofauna community structure.

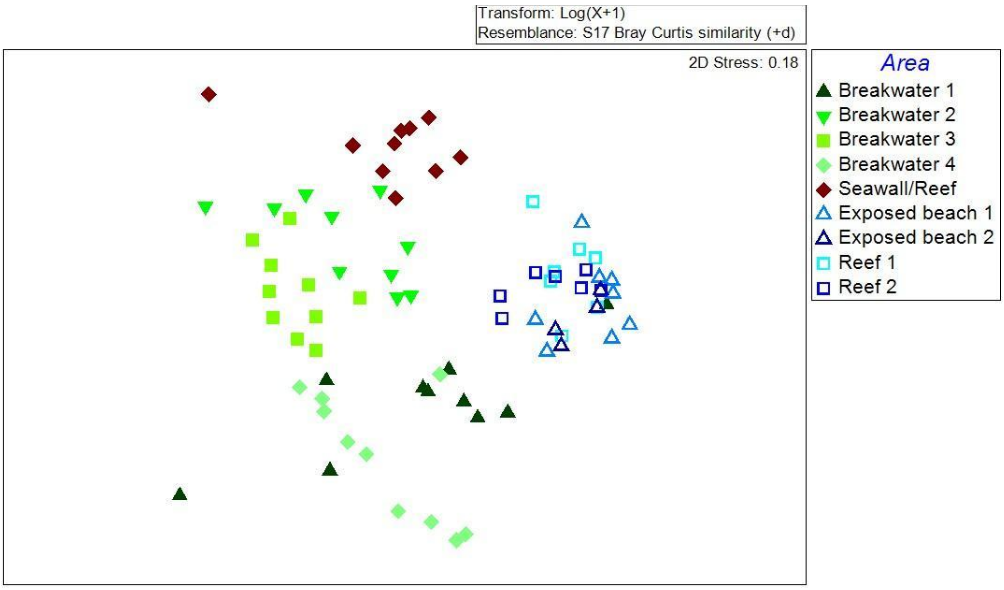

Figure 2. Non-metric multidimensional scaling ordination plot (nMDS) of the macrofaunal community structure of Breakwaters 1-4, Seawall/Reef, Exposed beach 1-2 and Reef 1-2.

The average similarity within each area was considered low, with the area with the highest value, Breakwater 3, not reaching 40%, while the lowest values were found in Exposed beach 1 (14.2%) and 2 (5.56%). It was also possible to observe that the taxa that contributed the most to Breakwater 1-4 were Polychaeta (Capitellida, Opheliidae, Magelonidae and Spionidae), while for Seawall/Reef it was the gastropod *Astyris lunata* (Say, 1826), for Exposed beach 1 the Polychaeta Glyceridae, for Exposed beach 2 "Turbellaria", and for Reef 1-2 Bivalvia (Table 4).

Table 4. Average similarity within areas and the percentage of groups that contributed most to this similarity.

Breakwater 1 Average similarity: 31.8		**Breakwater 2** Average similarity: 31.2		**Breakwater 3** Average similarity: 39.1	
Species	Contrib %	Species	Contrib %	Species	Contrib %
Capitellidae	84.52	Opheliidae	38.76	Magelonidae	28.18
Spionidae	6.42	Bivalvia	19.64	Capitellidae	19.2
		Magelonidae	11.57	Bivalvia	13.54
		Astyris lunata	9.33	Sipuncula	10.94
		Spionidae	6.41	Cirratulidae	7.96
		Goniadidae	3.76	Comesoma	3.65
		Anomalocardia flexuosa	3.34	Goniadidae	3.44
				Spionidae	3.16
Breakwater 4 Average similarity: 26.0		**Seawall/Reef** Average similarity: 29.9		**Exposed beach 1** Average similarity: 14.2	
Species	Contrib %	Species	Contrib %	Species	Contrib %
Spionidae	43.94	*Astyris lunata*	29	Glyceridae	78.13
Vitta virginea	39.59	Sipuncula	21.48	Syllidae	7.81
Capitellidae	7.35	Polynoidae	13.94	Sigalionidae	7.81
		Goniadidae	13.61		
		Cirratulidae	6.95		
		Paraonidae	5.1		
Exposed beach 2 Average similarity: 5.56		**Reef 1** Average similarity: 23.4		**Reef 2** Average similarity: 20.8	
Species	Contrib %	Species	Contrib %	Species	Contrib %
"Turbellaria"	60	Bivalvia	71.55	Bivalvia	96.88
Spionidae	40	Glyceridae	28.45		

DISCUSSION

Coastal areas, recognized globally as areas with high demand for occupation and use, have unique attractions: their scenic appeal, associated with remarkable biological wealth and multiple possibilities for exploitation, has transformed them into poles of intense human concentration and important socioeconomic sources (GOIS et al., 2013). However, this same concentration makes these regions particularly susceptible to coastal erosion. In response to this threat, coastal defense structures have become frequent elements in the coastal landscape (CHARLIER et al., 2005). The construction of these structures, in turn, has been documented as responsible for significant impacts on sandy beaches and their biodiversity, with especially pronounced effects on benthic communities (JARAMILLO et al., 2021).

The coast of Pernambuco, northeastern Brazil, presents a remarkable diversity of coastal contexts, including beaches with deep or shallow reefs, exposed areas subject to high wave energy, sheltered bays and areas with multiple defense structures such as breakwaters, groynes, ripraps and seawalls. The present study indicates that the presence of these coastal defense structures exerts a significant influence, both on environmental variables and on the structure of local soft-bottom biological communities.

Regarding sedimentary variables, a distinct pattern was observed between the areas impacted by coastal defense structures and those not modified. The sampling areas associated with defense structures (Breakwaters 1, 2, 3 and Seawall/Reef) exhibited predominantly muddy sand sediments, characteristic of environments with high deposition rates. This transition from sand sediments to finer textures after the installation of rigid structures is described by Moreira et al. (2010), who attribute the change to the reduction of local hydrodynamic energy. In contrast, beaches without artificial interventions maintained a typical sand texture sedimentary profile, reflecting natural hydrodynamic conditions.

Areas with artificial coastal defense structures also recorded the lowest seawater bathing suitability indices, an indicator that estimates the most likely number of *Escherichia coli* and thermotolerant coliforms in contact with bathers. According to the standards of the Pernambuco State Environmental Agency (2023), values classify these beaches as unsuitable for recreation during most of the year. In contrast, beaches without artificial defense structures presented high seawater bathing suitability, reflecting satisfactory environmental conditions compatible with recreational use.

The macrobenthic communities, in general, showed a numerical predominance of Polychaeta, particularly in areas with coastal defense structures (Breakwaters 1, 2, 3, and 4) and a combined Seawall/Reef. This dominance is directly associated with the physiological characteristics of the group, such as tolerance to salinity fluctuations and ability to survive in hypoxic conditions, typical of fine sand and muddy sand sediments (GLASBY et al., 2021). The environmental alteration induced by the structures (accelerated sediment deposition, accumulation of organic matter and reduction of oxygen within the sediment) favored the establishment of opportunistic species, which massively colonized the disturbed habitat. At the same time, a high abundance of Nematoda was observed in these same areas. Both taxa (Polychaeta and Nematoda) are recognized as bioindicators of organic enrichment and anthropogenic disturbance, with their population increase signaling environmentally degraded conditions (MIRTO et al., 2002; VALENÇA & SANTOS, 2012). Interestingly, the bathing suitability index presented high correlation with macrobenthic species richness, indicating the increase of tolerant/opportunistic species due to organic pollution and sediment texture shifts, as expressed by the bathing suitability index and the percentage of mud which were positively correlated (Pearson r=0.70; p=0.035) probably due to diminished water renewal induced by the coastal defense structures.

Although the natural reef protected (Reef 1-2) and exposed areas (Exposed beach 1-2) also exhibited dominance of Polychaeta, their absolute density was significantly lower. This large numerical reduction is directly associated with the higher hydrodynamic energy characteristic of these environments. As demonstrated by Alves et al. (2004), beaches exposed to greater wave action and vigorous circulation show a drastic decrease in macrofaunal abundance, an effect that contrasts with areas where artificial structures dissipate energy, allowing sediment stability and biological colonization.

In a seemingly contradictory manner, the areas Breakwaters 1, 2 and 3 and Seawall/Reef recorded the best performances in the univariate ecological indices, superficially suggesting greater environmental quality. However, this is a mistaken interpretation: high richness or abundance do not necessarily equate to healthy ecological conditions, since anthropogenic disturbances can favor tolerant species over sensitive species (GIANGRANDE et al., 2005) and the energy gradient established by defense structures turns the comparison very entangled to the main paradigm in sandy beach ecology that states that macrofaunal richness increases from reflective to dissipative beaches (MCLACHLAN & DORVLO, 2005; BARBOZA et al., 2012). However, this high richness is due to the presence of opportunistic species and may reduce the functional diversity of these ecosystems (BORJA et al., 2000; JOHNSTON & ROBERTS, 2009; VILLÉGER et al. 2017).

Areas without defensive structures (Reef 1-2 and Exposed Beach 1-2) presented the lowest values for richness and abundance, despite representing the most ecologically preserved environments. This apparent contradiction concerning the expected pattern is explained by the high hydrodynamic energy characteristic of these beaches. Even in areas partially sheltered by natural reefs, high wave action conditions persist during high tide. This occurs because the reefs themselves, in addition to being more widely spaced, allow much of the wave energy to pass through when submerged at high tide. This dynamic contrasts with that of artificial breakwaters, which, by being completely above water level, effectively block the passage of water and dissipate wave energy.

The AMBI index, globally recognized as a robust biotic indicator, has had its application predominantly concentrated in European countries (BORJA et al., 2000; MUXIKA et al., 2005; LABRUNE et al., 2006; CARVALHO et al., 2006; DAUVIN et al., 2007; ZETTLER et al., 2007; BORJA et al., 2007), due to institutional support and consolidated local databases. However, its use has expanded to other regions, including North Africa (BAZAIRI et al., 2005), Indian Ocean (BIGOT et al., 2008; FEEBARANI et al., 2016) and South America (MUNIZ et al., 2005). In tropical ecosystems, the index is often combined with other metrics to ensure greater accuracy in assessing environmental quality.

The AMBI index values (0 = unchanged, 7 = severely impacted) reflect the distinct ecological conditions of the studied areas. Exposed Beach 2 (AMBI = 0.2) stands out as having the best ecological status ("unchanged"), a direct result of its high hydrodynamics and absence of barriers, which favor sensitive benthic communities and greater oxygenation. Areas protected by natural reefs (Reef 1: 1.67; Reef 2: 1.77) and Breakwater 2 (1.61) also present low values ("undisturbed"), indicating ecological balance, the result of efficient hydrodynamics (reefs) and possible favorable environmental integration (Breakwater 2), despite the artificial structure. Environments with greater sediment deposition exhibit greater disturbance: Breakwater 3 (2.11), and Seawall/Reef (2.31) are in the “transition” range (2.0-3.3), suggesting moderate disequilibrium, probably due to the interaction between artificial structures, dynamic sandbars and retention of organic matter, while Breakwater 4 presents a peculiar situation, as it was the only artificial area characterized with sandy sediment and AMBI (2.97), which is probably related to the high number of Spionidae characterized as moderately tolerant. Breakwater 1 (AMBI = 3.9, “disturbed”) presents the worst result, consistent with its intense sediment deposition, which promotes anoxic conditions and favors tolerant species. Regarding the exposed beaches, although both share high hydrodynamic energy, Exposed Beach 1 (AMBI = 1.8) presents a slightly higher value than Exposed Beach 2 (0.2). This difference may be associated with local factors, which affect the composition of the benthic community, reinforcing that even naturally dynamic environments present natural gradients of quality, in addition to the low number of sampled organisms. In summary, AMBI shows that dynamic environments without barriers (Exposed Beach 2) or with effective natural protection (Reefs) sustain the best ecological conditions. Artificial structures can maintain acceptable quality (Breakwater 2) but tend to induce disturbance when associated with greater sediment retention (Breakwater 1 and 3) or low ORD conditions (Breakwater 4).

The DISTLM analysis demonstrated that bathing suitability index or seawater quality and ORP act as determinant factors in the structuring of benthic macrofauna, jointly explaining 53% of the observed variability. This result highlights the critical role of these variations in the configuration of biological communities, indicating that these interventions can directly influence marine biodiversity. The strong positive correlation between taxon richness and bathing suitability index indicates that environments with better environmental quality support more diverse communities. Similarly, the strong inverse correlation between AMBI and ORP reveals that sediments with higher oxidative potential (high ORP) are associated with lower AMBI values, reflecting healthier ecological conditions. This relationship highlights ORP as a strong indicator of benthic health.

Our results indicate no significant correlation between AMBI and BOPA. When comparing the results of the indices for the extremes of environmental quality, we can observe that in the artificial areas, the indices showed notable divergence in identifying the most degraded area. While AMBI identified Breakwater 1 as the worst environmental condition, with a value of 3.9 classified as "moderately disturbed," BOPA identified Breakwater 4 as the most critical, registering a value of 3.34 in the "impacted" category. However, both indices converged in diagnosing serious problems in Breakwater 4, whose AMBI of 2.97 was very close to the threshold for the moderate category (3.3), confirming its degraded condition. In the natural areas, convergence was observed regarding environmental excellence, but with a difference in identifying the optimal environment. AMBI selected Exposed Beach 2 as the highest quality area, with a score of 0.2 ("intact"), while BOPA highlighted Reef 1 as the best preserved, with a score of -0.07 ("intact"). This discrepancy does not imply a contradiction, as both areas maintained the highest ratings in their respective indices, demonstrating that Reef 1 and Exposed Beach 2 represent environments of the highest quality, although the indices have different ecological sensitivities. Another factor that may have influenced these results is the fact that AMBI is a linear measure, while BOPA is not linear, but governed by an exponential dynamic (RUELLET & DAUVIN, 2007). This implies that small changes in the structure of the benthic community can generate large variations in the BOPA value under specific conditions, revealing tipping points, which seems to have happened at Breakwater 4 (3.34) and Reef 1 (-0.07). Analyzing the relationship between AMBI and BOPA using data from 8 studies, de-la-Ossa-Carretero and Dauvin (2010) found a strong correlation between the indices for French Atlantic transitional waters, Mediterranean coastal waters and Mediterranean lagoons. Other studies, however, indicated that in muddy environments where the organic matter in the sediment is naturally high these indices presented limitations mainly when low abundances and/or number of taxa are recorded in the samples (MUNIZ et al., 2005; MUXIKA et al., 2005; BLANCHET et al., 2008).

Through the analysis of the IndVal, it was observed that each breakwater presented distinct indicator taxa (e.g., Capitellidae at Breakwater 1, Opheliidae at Breakwater 2), highlighting how coastal defense structures create unique microhabitats that favor specific benthic communities. Capitellidae polychaetes are classic opportunistic taxa, thriving in organically enriched sediments common near breakwaters due to trapped organic debris and reduced hydrodynamics (ZYSERMAN et al., 2005; OMENA et al., 2012; CARUGATI et al., 2018). Similarly, Opheliidae and Magelonidae are deposit-feeders adapted to stable, muddy substrates that accumulate in the sheltered lee of breakwaters (FAUCHALD & JUMARS,1979; FIEGE et al., 2000; PARAPAR et al., 2021). The presence of Sipuncula and *Upogebia omissa* at Breakwater 3 aligns with their preference for low-energy, burrow-friendly environments, such as sandy banks present in the area, or with a large presence of shells or tubes that serve as shelter (FERRERO-VICENTE et al., 2011; COSTA et al., 2020). Spionidae and the gastropod *Vitta virginea* at Breakwater 4 also appear to benefit from the modified environment, with Spionidae having an opportunistic tendency to colonize disturbed habitats (GRASSLE & GRASSLE, 1974), while *V. virginea* is known to rapidly colonize habitats such as breakwaters, where it finds stable substrates and abundant food resources, and already has algae and diatoms as its diet (TAN & CLEMENTS, 2008). Crucially, the absence of shared indicators among breakwaters highlights that even subtle differences in structural design, age or hydrodynamic exposure generate divergent ecological niches despite the homogenizing effect. The lack of significant indicator taxa at Seawall/Reef, Reefs 1–2, and Exposed Beaches 1–2 is likely due to the greater niche diversity at each area and to the greater ecological stability of these systems. Although Seawall/Reef has an artificial defensive structure, the nearby reef appears to have a direct influence on the community sustained in the area, meaning that, like natural areas Reef 1-2 and Exposed beaches 1-2, there is no large dominance of a few taxa.

CONCLUSIONS

This study demonstrates that coastal defense structures, although designed to mitigate erosion, significantly alter benthic community composition by establishing hydrodynamically protected environments. These modified conditions promote muddy sediments and organic matter accumulations, favoring opportunistic taxa that signal ecological disturbance. Although richness and abundance metrics suggest greater diversity in these habitats, AMBI reveals reduced functional diversity, dominated by species adapted to environmental degradation. In contrast, naturally exposed beaches and reef-protected areas exhibit superior ecological quality (AMBI) despite lower abundance and richness, indicating that low organismal densities or richness in high-energy environments reflect environmental physical constraints rather than poor environmental health. We highlight the use of redox potential (ORP) as a robust predictor of environmental quality, while indices based on specific taxonomic relationships, such as

BOPA, may have limitations depending on the sedimentary context. In support of these findings, IndVal analysis identifies indicator taxa unique to breakwaters, confirming that coastal defense structures generate unique microhabitats that promote homogenized benthic communities.

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Statements and Declarations

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Availability of data and material

Data will be made available upon reasonable request.

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Macrobenthic communities in coastal environments: artificial defense structures, natural reefs and exposed beaches

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Supplementary Material

Figure S1. List of taxa by area. 0 means absence of the taxon and 1 means presence of the taxon. B=Breakwater, SR=Seawall/Reef, R=Reef and EB=Exposed beach.

		B1	**B2**	**B3**	**B4**	**S/R**	**R1**	**R2**	**EB1**	**EB2**
Annelida										
	Polychaeta									
	Capitellidae	1	1	1	1	1	1	0	0	0
	Spionidae	1	1	1	1	0	0	1	1	1
	Nereididae	1	0	1	0	0	0	0	0	0
	Magelonidae	0	1	1	1	1	0	0	0	0
	Cirratulidae	0	1	1	1	1	0	0	0	0
	Opheliidae	0	1	1	1	0	0	1	0	0
	Glyceridae	0	1	1	0	1	1	0	1	0
	Goniadidae	0	1	1	1	1	0	0	0	0
	Hesionidade	1	1	1	1	0	0	0	0	0
	Polynoidae	0	1	1	0	1	0	1	1	0
	Sabellidae	0	1	1	0	1	0	0	0	0
	Syllidae	1	0	0	1	0	0	0	1	0
	Orbiniidae	0	1	1	1	1	0	1	0	0
	Pilargidae	0	0	1	0	0	0	0	0	0
	Paraonidae	0	0	1	1	1	1	0	0	0
	Onuphidae	1	0	1	0	1	0	0	0	0
	Eunicidae	0	0	1	0	0	0	0	0	0
	Chaetopteridae	0	0	1	0	0	0	0	0	0
	Dorvilleidae	0	0	0	1	0	0	0	0	0
	Oweniidae	0	1	1	0	1	0	0	0	0
	Sternaspidae	0	0	1	0	0	0	0	0	0
	Sigalionidae	0	0	0	0	0	0	1	1	1
	Oligochaeta	1	0	0	0	0	0	0	0	0
	Sipuncula	1	1	1	0	1	0	0	0	0

Nemertea		0	1	1	0	0	0	0	0	0
Nematoda										
	Acanthonchus	1	0	0	0	0	0	0	0	0
	Comesoma	1	1	1	1	1	0	0	0	0
	Dorylaimopsis	0	1	1	0	1	0	0	0	0
	Euchromadora	1	0	0	0	0	0	0	0	0
	Eurystomina	1	0	0	0	0	0	0	0	0
	Gomphionema	1	0	0	0	0	0	0	0	0
	Linhomoeus	1	1	1	0	0	0	0	0	0
	Oncholaimellus	0	1	0	0	1	0	0	0	0
	Oncholaimus	1	1	0	0	0	0	0	0	0
	Pespiria	1	0	0	0	0	0	0	0	0
	Prochromadorella	1	0	0	0	0	0	0	0	0
	Sabatieria	1	1	1	0	0	0	0	0	0
	Terchellingia	1	0	0	0	1	0	0	0	0
	Theristus	1	0	0	0	0	0	0	0	0
	Trissonchulus	0	1	1	0	0	0	0	0	0
	Sphaerolaimus	0	0	0	0	1	0	0	0	0
	Viscosia	0	0	1	0	1	0	0	0	0
	Metadesmolaimus	0	0	0	0	1	0	0	0	0
	Desmoslaimus	0	0	1	0	0	0	0	0	0
	Mesacanthoides	0	0	0	1	0	0	0	0	0
	Metalinhomoeus	0	0	1	0	0	0	0	0	0
	Phanoderma	0	0	1	0	0	0	0	0	0
Mollusca										
	Bivalvia	1	1	1	0	1	1	1	0	0
	Anomalocardia flexuosa	1	1	1	1	1	1	0	1	0
	Gastropoda									
	Vitta virginea	0	0	1	1	0	0	0	0	0
	Astyris lunata	0	1	0	0	1	0	0	0	0
Crustacea	**Copepoda**	1	0	0	0	0	0	0	0	0
	Cirripedia	0	0	1	0	0	0	0	0	0
	Brachyura	0	0	0	1	1	0	0	0	0
	Caridea	0	0	0	0	1	1	0	1	0
	Paguroidea	0	1	1	1	1	0	0	0	0
	Ostracoda	0	1	0	0	1	0	0	0	0
	Upogebia omissa	0	0	1	0	0	0	0	0	0
	Emerita	0	0	0	0	0	0	0	1	0
	Amphipoda									
	Atylidae	1	0	0	1	0	1	0	0	0
	Ampitoidae	1	0	1	0	0	0	0	0	0
	Cymadusa	1	0	0	0	0	0	0	0	0
	Cheiriphotis	1	1	0	0	0	0	0	0	0
	Elasmopous	1	0	0	0	0	0	0	0	0
	Caprellidae	1	0	0	0	0	0	0	0	0
Chelicerata	**Pycnogonida**	1	0	0	0	0	0	0	0	0

Echinodermata	**Ophiuroidea**	0	1	0	0	1	0	0	0	0
Platyhelminthes	"Turbellaria"	0	1	1	1	1	1	0	1	1

6 CAPÍTULO 3 - Design differences between two sets of breakwaters have different effects on the benthic macrofauna of sandy beaches

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Abstract

Breakwaters are man-made constructions installed to remedy the effects caused by coastal erosion. An inadequate breakwater design can affect water quality, alter sedimentation rates and the quantity of organic matter and nutrients in these areas. As such, the community structure of benthic macroinvertebrates can provide rapid answers about the ecological condition of these areas. Thus, the present study evaluated the community structures and trophic functioning of benthic macroinvertebrates that inhabit two sandy beaches that were highly modified after the construction of large sets of breakwaters. Here we test the hypothesis that Casa Caiada beach, despite being older, due to a breakwater built in a less impactful way, will present a more diverse benthic community with greater balance between trophic groups, compared to the Janga beach, a more recent structure but with more impactful structures. The analysis indicated that the community of Janga is more homogeneous than that of Casa Caiada, which appears to be caused by the isolation of the beach due to the formation of a tombolo. The trophic community structure pattern was broadly similar to the taxonomic community structure pattern, indicating that the changes in the community structure along the environmental gradients generated by tombolo formation and sediment accumulation also resulted in changes in the ecological functioning of the trophic groups in the beaches. In general, both Janga and

Casa Caiada sandy beaches face issues resulting from the installation of breakwaters, however, the design of the breakwaters constructed at Janga beach caused a greater impact than those installed at Casa Caiada beach.

Keywords: Anthropogenic impacts; Coastal erosion; Defense structures; Sandy coasts; Trophic guilds.

Introduction

The coastal zone is one of the most complex environments in nature, with its balance maintained by natural forces (waves, currents, tides and wind) in the short-term balance and by fluctuations in sea level in the long-term. The coast is recognized for its important socioeconomic role, providing resources, and representing an area with a highly concentrated human population (Burke et al., 2001; Manso et al., 2006).

Coastal erosion is a global phenomenon and is considered one of the most relevant issues worldwide, leading to a growing demand for protective measures. However, the installation of structures to contain the advance of the sea, which aims to remedy the effects caused by coastal erosion and consequent retreat of the coastline (Dias, 1993), is considered one of the main problems faced by coastal environments (Bulleri and Chapman, 2010). These structures have been built frequently worldwide but are almost always installed without an adequate understanding of their effects on local coastal dynamics (Mallmann and Pereira, 2014).

In Brazil, coastal erosion and sea advance are more evident in the Northeast region, and more strongly in the state of Pernambuco (Neves and Muehe, 1995; Costa et al., 2010). It is estimated that 1/3 of the coast of Pernambuco suffers, to varying degrees, from erosion processes and coastline retreat (Manso et al., 2006), a situation that is expected to worsen because of sea level rise resulting from global climate change (Marengo et al., 2016). In several stretches of Pernambuco, there has been strong public intervention with the construction of various types of coastal defense structures (breakwaters, seawalls, groynes and rip raps) to contain the advance of the sea in the municipalities of Olinda, Paulista, Recife and Jaboatão dos Guararapes (Araújo, 2007).

The history of erosion problems in Olinda dates back to 1909, after the expansion of the Port of Recife, with reports of the construction of rigid structures from the 1950s onwards. During this period, several projects were built, with the construction of a large group of seven breakwaters on the beaches of Bairro Novo, Casa Caiada and Rio Doce that were completed between the 1970s and 1980s (Pereira et al., 2007). According to Pereira et al. (2006), the construction of this group of breakwaters did not prevent the erosion process in the region; on the other hand, it promoted changes in the local topography and hydrodynamics. Along the beaches, it is possible to observe strong sediment deposition with the formation of protrusions on several breakwaters and, in some places, large sandbanks.

In Pernambuco, coastal erosion problems are generally transferred from south to north, as new coastal structures are installed and sediment flow supplied by longshore drift is reduced or obstructed. Consequently, Janga beach (Paulista), suffered a strong impact of coastal erosion between the 1980s and 1990s. This impact was intensified by the transfer of erosion from Olinda, located to the south, after the construction of defense structures on the Rio Doce-Paratibe (Pereira et al., 2007), in addition to extensive construction projects on the beaches of Casa Caiada and Rio Doce mentioned previously.

Therefore, in the 1990s, construction began on a set of nine breakwaters parallel to the coastline, to contain erosion at Janga beach, and was completed in 2001 (Silva et al., 2015). Over the years, following the construction of this set of breakwaters, a complete change in the beach landscape was observed. There were the formation of bays and a marked process of sediment deposition in the region close to and in the direction of the breakwaters, and a major erosion process in the center of the bay. In a study on the analysis of the stability of Janga beach, Silva et al. (2015) indicated some changes that occurred on this beach, in addition to projections on the stability of these bays. They concluded that the bays analyzed, for the most part, had not yet reached their static equilibrium and that the erosion process in these areas would likely continue.

Currently, 20 years after the completion of the project, it is possible to observe the formation of two tombolos on Janga beach, due to the erosion and deposition process of dammed areas. In other breakwaters, strong depositions are observed in some areas, forming protrusions close to the structures. The present study does not aim to make projections about the changes in the morphology of this beach, but given the accentuated deposition process in the area, it is quite likely that over time other tombolos will be formed, transforming the beach that was once continuous into small individual beaches.

Although studies on the impacts of the installation of coastal defense structures have been carried out in several locations around the world (Wilding, 2006; Walker et al., 2008; Wehkamp and Fischer, 2013 a, b; Munari, 2013; Laurino et al., 2022), changes in beach morphology and the response of the benthic macrofauna community that inhabits the shallow infralittoral have not been effectively studied.

Changes in the structure and population dynamics of benthic macroinvertebrates can provide rapid answers about the ecological condition of these areas. The macrobenthic fauna (i.e., organisms retained in a 0.5 mm mesh opening) are highlighted as excellent bioindicators of the environmental quality of beaches. They are frequently used in studies that seek to investigate the spatial and temporal patterns of species in areas with different degrees of urbanization (Veloso et al., 2006; Coutinho and Bernardino, 2017; Santos et al., 2021) or with distinct morphodynamic characteristics (Barros et al., 2001; Veloso and Cardoso, 2001; Vieira et al., 2012), and therefore, act as a useful environmental monitoring tool. These organisms are numerous and diverse, in addition to being sensitive to natural and anthropogenic disturbances. They are in direct contact with the sediment, and, consequently, with the available pollutants that are consumed and/or absorbed by these organisms and other life forms, since they connect primary producers and higher trophic levels (Laetz, 1998).

As such, studies of highly modified environments, such as areas created after the formation of overhangs, due to strong sediment deposition and the formation of tombolos that laterally close the beach, or regions that present intense sediment deposition, but that still present continuity where there may be sedimentary dynamics, can provide important information on how benthic communities respond to the different impacts caused by the construction of a breakwater. Thus, the present study evaluated the structure of benthic macroinvertebrate communities, comparing both the taxonomic and the trophic functional structures, which inhabit two beaches that were highly modified after the construction of large sets of breakwaters.

We hypothesize that there will be a difference in community structure between the two beaches. We expect the beach with a breakwater built in a less impactful way (Casa Caiada) to present a benthic community with greater taxonomic diversity and balance between trophic groups, compared to the beach with more impactful structures (Janga). We also expect the community to be dominated by a few opportunistic species and specific trophic guilds, reflecting the intensity of hydrodynamic and sedimentary alterations.

Materials and methods

Study area

The study areas (Fig. 1) were Janga Beach, located in Paulista, with an extension of approximately 4 km, and Casa Caiada Beach, located in Olinda, with an extension of approximately 3 km, both located in the Metropolitan Region of Recife. The two beaches have large sets of breakwaters built with granite blocks arranged parallel to the coastline. Both areas were studied during the rainy season that extends from March to August with monthly rainfall averages above 200 mm according to the monthly bulletins published by the Pernambuco Water and Climate Agency (APAC). Although each beach was sampled in different years (2022 and 2023), the climatic seasonal pattern was very similar for both years (Fig. S1). The rainy period remained the same, but rainfall was more concentrated in May and June compared to the historic average, with values slightly below average in April and June for both years (Fig. S1). The tides are classified as meso-tides (range up to 2.5 m) and are semi-diurnal.

Sampling at Janga beach was carried out on March 18, 2022, at four stations (S1 to S4) located in two areas: (1) the first of which was an area where there was large sediment deposition due to the presence of the breakwater, creating a tombolo over the years, closing one side of the beach, causing this stretch of the beach to become perpendicular to the coastline, and not parallel as usual (Fig 1). Two stations were established (S1 and S2) here. The second area (2), between two breakwaters, is an area with large sediment deposition, but without the formation of a tombolo, keeping both sides of the beach open. In this area, both stations (S3 and S4) were located in a shallow area adjacent to the breakwater's opening.

At Casa Caiada beach, sampling took place on July 5, 2023, also at four stations (S5 to S8) located in two distinct areas: the first (3), located in the shallow infralittoral, parallel to the coastline, where two stations were established (S5 and S6). The second (4) constitutes an area with strong sediment deposition located close to the breakwater opening, in an inlet with high water flow and where the other two stations (S7 and S8) were located. A table with information on breakwater lengths and opening lengths of both beaches is available in Table S1.

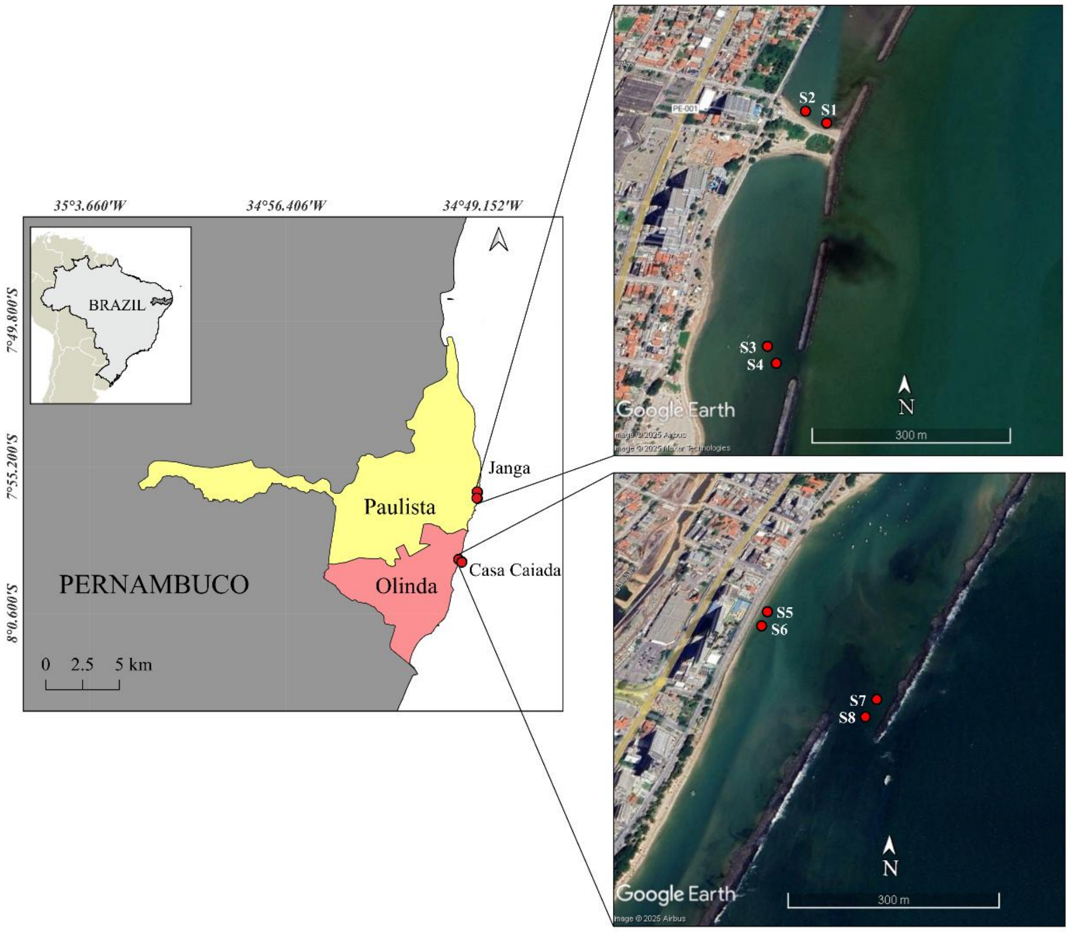

Figure 1. Location of sampling areas and stations on Janga and Casa Caiada beaches.

Sample design

To investigate the environmental parameters (salinity and granulometry), water samples were collected from the infralittoral (3 replicates in each station) and sediment was collected using a cylindrical sampler (17.4 cm² opening, 5 cm depth). In the laboratory, a refractometer was used to measure seawater salinity and the granulometric analysis was performed following the methodology of Suguio (1973). The result of this parameter was expressed in percentages of silt clay.

For spatial/morphological characterization of the area, Google Earth® was used, and measurements were made using satellite images of breakwater length, in addition to the number and width of openings between the breakwaters that allow water exchange.

To sample biological material, samples were taken during low tide at 2 stations located in the shallow infralittoral (60 cm deep at low tide) parallel to the beach line, at least 10 m apart, for each area. Five replicates were collected at each station, totaling 20 sample replicates for each beach.

A cylindrical PVC sampler with H = 10 cm and Ø = 15 cm was used, corresponding to an area of 0.0177 m^2. The sampled sediment was sieved through a 0.5 mm nylon screen, and the retained material was stored in plastic bags or pots, duly labeled and fixed in 4% saline formalin. At the laboratory, the samples were subsequently washed under running water and the macrofauna was sorted and identified to the lowest possible taxonomic level.

The Polychaeta taxa present in the samples were classified into feeding guilds according to Fauchald and Jumars (1979) and Jumars et al. (2015) using the three-letter codes. The letter in the first position indicates the main feeding mode (B, subterranean deposit feeder; C, carnivore; F, filter feeder; H, herbivore; S, surface deposit feeder); families that have more than one feeding mode were classified as O, omnivorous. The second letter indicates the motility pattern (D, discretely mobile; M, mobile; S, sessile;) and the last letter indicates the morphological structure used in feeding (J, jaw; P, pumping; T, tentacle; X, other structures, usually pharyngeal in the form of an eversible sac).

Nematoda were processed using three solutions (De Grisse, 1969): solution I, containing formaldehyde (4%) and glycerin; solution II, with 5 parts of ethanol (96%) with 95 parts of glycerin; and solution III, containing 50 parts of ethanol (96%) and 50 parts of glycerin. This diaphanization process is necessary for observing the internal structures used for identification. Identification to the genus level was obtained using the pictorial key of Warwick et al. (1998). Subsequently, they were grouped into one of the four trophic groups provided by Wieser (1953), based on the morphology of the oral cavity: 1A, selective deposit feeders; 1B, nonselective deposit feeders; 2A, epistrate scrapers; and 2B, predators/omnivores. Due to the use of Polychaeta and Nematoda in trophic analyses, the term trophic group will be adopted.

Data analysis

To assess the differences in taxonomic community structure, the abundance data were transformed to Log(x+1), then a similarity matrix was generated using Bray-Curtis index. Non-metric Multidimensional Scaling (nMDS) was used to represent, in a two-dimensional model, the similarity among replicates based on taxa abundances. The Permutational Multivariate Analysis of Variance (PERMANOVA) (Anderson, 2001) compared the differences between beaches and between stations (nested within beach) concerning their similarities. As the PERMANOVA showed significant results, pairwise comparisons were performed a posteriori (pair-wise t-test). For all analyses, 9999 random permutations were used. To determine the most important taxa in terms of contribution (%) to the dissimilarity between areas, the similarity percentage analysis (SIMPER) was applied (Clarke, 1993). To detect non-random distributions of species (indicator species) or trophic groups between stations, the Indicator Species Analysis (IndVal), developed by Dufrêne and Legendre (1997), was used. The IndVal coefficient combines both the species' relative abundance (specificity) with its relative frequency of occurrence (fidelity) in a defined group of samples. The statistical significance of the species or group indicator values was evaluated using a Monte Carlo test. Species/taxa or trophic groups were considered indicators when IndVal coefficients were significant and ≥40 (Kubosova et al., 2010). The trophic groups' abundance data was also Log(x+1) transformed and the same tests used to assess community structure were used to assess the trophic groups. The analyses were performed using the PRIMER v.6+PERMANOVA and PAST software. The significance level adopted for all statistical analyses was 5%.

Results

Environmental and morphological parameters

On Janga beach, where S1 and S2 were located, it is possible to observe an area protected by two breakwaters (8 and 9). The closest (8), where the tombolo is located, is approximately 270 m long, while breakwater 9, located further north, is approximately 655 m long. Between these two breakwaters, there is only one water inlet, parallel to the beach line, approximately 50 m long. The area comprised by stations S3 and S4 is in the shape of a bay, approximately 495 m long, and is protected by part of breakwater 8 (87 m), breakwater 7 (185 m) and part of breakwater 6 (128 m). This bay has two water inlets parallel to the beach line, approximately 45 and 40 m long (Fig. S2a).

At Casa Caiada beach, stations S5 and S6 are in an area with strong sediment deposition, parallel to a deeper channel that limits the shallowest part of the beach, which is protected by breakwater 4. Stations S7 and S8 are close to the opening between breakwater 3, which is approximately 550 m long, and breakwater 4, which is approximately 850 m long. This opening is perpendicular to the beach line and measures approximately 70 m (Fig. S2b). The environmental parameters of the Janga and Casa Caiada beaches are demonstrated in Table 1.

Table 1 Percentage of silt/clay and mean and standard deviation of salinity and temperature per station on Janga and Casa Caiada beaches.

	Stations	**Silt Clay (%)**	**Salinity**	**Temperature**
Janga	S1	79.5 sM	35.7±0.6	32.3±0.2
	S2	39.3 mS	35.3±0.6	31.5±0.1
	S3	40.7 mS	34.6±0.6	31.6±0.1
	S4	50.9 sM	34±0	30.6±0.1
Casa Caiada	S5	28.8 mS	29.3±0.1	30.2±0.2
	S6	24.7 mS	31.3±0.1	30.4±0.1
	S7	5.3 S	31±0.1	29.9±0.2
	S8	2.9 S	30.6±0.1	30.1±0.1

Macrobenthic community structure

The benthic macrofauna of Janga beach comprised a total of 48 taxa with an average density of 2,150 ind/m² (minimum of 915 ind/m² in S3 and maximum of 3,424 ind/m² in S4). The group with the largest number of taxa was Polychaeta with 15 families, with Capitellidae being the most abundant. In S1, with a density of 3,299 ind/m², Capitellidae was the dominant taxon, accounting for 21.2% of the individuals, followed by the nematodes *Euchromadora* and *Oncholaimus,* both comprising 20.2% of the individuals. In S2, with a density of 960 ind/m², Capitellidae was the most abundant taxon (72% of the individuals), in S3 the density was 915 ind/m^2 and Opheliidae was the most abundant taxon (28% of the individuals) and in S4, with a density of 3,424 ind/m², Opheliidae also accounted for the greatest abundance (20% of the individuals).

At Casa Caiada beach, the benthic macrofauna comprised 43 taxa and presented an average density of 32,226 individuals/m² (minimum of 1,571 ind/m² in S5 and maximum of 122,531 ind/m² in S8), with great variation between stations. Polychaeta was the most abundant taxon with 21 families, of which Spionidae was dominant accounting for 95% of the individuals, mostly recorded in station S8. Among the stations, S5 presented a density of 1,571 ind/m², and Capitellidae was the most abundant taxon (20.9% of individuals), S6 presented a density of 2,508 ind/m², where Magelonidae had the greatest abundance (21.2% of individuals), S7 presented a density of 2,294 ind/m², and the mollusk *Vitta virginea* (Linnaeus, 1758) was the most abundant taxon (53.7% of individuals), while S8 presented an extremely high density of 122,531 ind/m², with Spionidae presenting the highest dominance with 99.6% of organisms.

The nMDS ordination shows a clear differentiation between the groups, with clusters between stations S1 and S2, S3 and S4, and S5 and S6. We can also observe a separation between the stations closest to the breakwater opening to the sea (S3, S4, S7 and S8) and those furthest from the construction site and close to the beach face (S1, S2, S5 and S6 (Fig. 2). Significant differences were found using a PERMANOVA for the factor beach (Pseudo-F = 8.204; p = 0.0001). Regarding the factor stations (nested within beach), significant differences were also found between stations (Pseudo-F=6.0606; p = 0.0001), and when applying the a posteriori test, significant differences were found between almost all stations ($p<0.05$), except between S1 and S2 (t = 0.94108; p= 0.5359) and between S3 and S4 (t =1.3504; p= 0.0638).

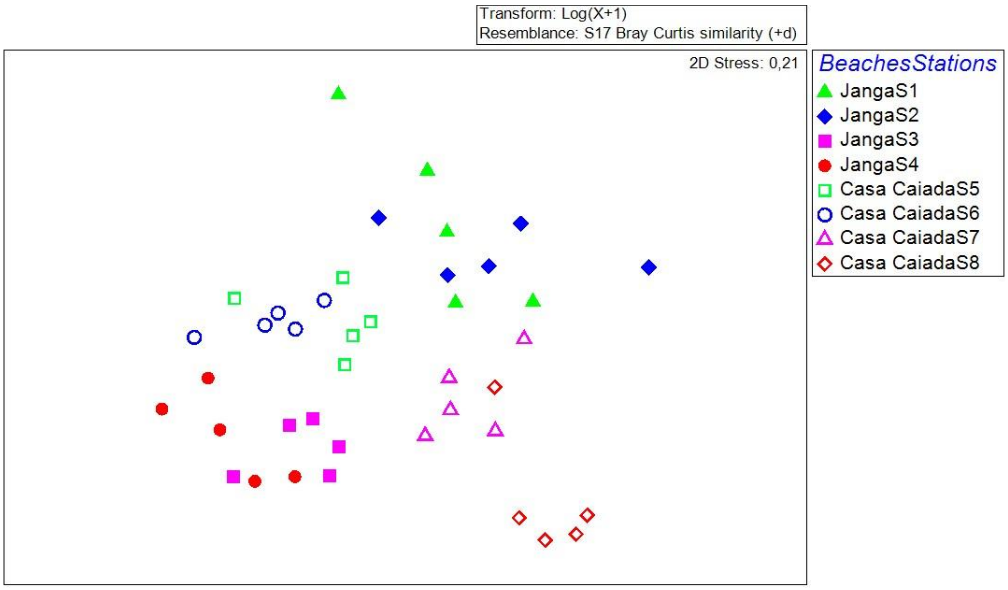

Figure 2. Non-metric multidimensional scaling ordination plot (MDS) of the macrofaunal community structure of Janga and Casa Caiada beaches. Stations are indicated as S1, S2, S3, S4, S5, S6, S7 and S8.

The SIMPER analysis indicated a low similarity between replicates for both Janga beach (18.5%) and for Casa Caiada beach (21.4%), while the dissimilarity between the beaches was quite high, at 79.4%. The taxa that contributed most to this high dissimilarity were Spionidae, *V. virginea* and Capitellidae (Table 2). As for the similarities between the station replicates, the highest occurred in S7 (55.7%) and S6 (54%). The stations with the greatest similarity were for Janga S3 with 40.7% and for Casa Caiada S7 with 55.7% (Table S2). The stations with the higher dissimilarities for Janga beach were S2 and S4 (96.5%) and S1 and S4 (94.8%), while at Casa Caiada beach, S6 and S8 (97.6%) and S5 and S8 (95.1%) (Tables S3 and S4) had the highest dissimilarity.

Table 2 Average similarity within Janga and Casa Caiada beaches and the percentage of groups that contributed most to the dissimilarity between beaches.

Janga		**Casa Caiada**		**Janga and Casa Caiada**	
Average similarity: 18.5		**Average similarity: 21.4**		**Average dissimilarity = 79.4**	
Species	Contrib%	Species	Contrib%	Species	Contrib%
Capitellidae	38.98	Capitellidae	20.99	Spionidae	25.41
Bivalvia	15.77	Spionidae	16.58	*V. virginea*	10.95
Opheliidae	15.46	Magelonidae	16.41	Capitellidae	10.86
		V. virginea	14.68	Magelonidae	6.24
		Bivalvia	5.86	Opheliidae	4.92
				Sipuncula	3.68
				Comesoma	3.47
				Bivalvia	3.29
				A. lunata	3.26

The IndVal coefficient revealed 16 significant indicator species/taxa at Casa Caiada and Janga beaches (Table 3). Based on the proposed IndVal scale, only taxa with high indicator values (>40%) are considered. Pycnogonida was considered indicative in S1, no indicator taxon in S2, Glyceridae in S3, Opheliidae, *Anomalocardia flexuosa* (Linnaeus, 1767) and *Astyris lunata* (Say, 1826) in S4, Cirripedia in S5, Magelonidae, Cirratulidae, Sipuncula and *Upogebia omissa* Gomes Corrêa, 1968 in S6, Paraonidae, Dorvilleidae, *V. virginea* in S7 and Spionidae and “Turbellaria” in S8. All indicator taxa found were different for each beach, Casa Caiada had 23.3% of indicator taxa (10 out of 43 taxa), while Janga had only 10.4% (5 out of 48 taxa).

Table 3 Indval results per station (S1 to S8), including all taxa that had a significant p(raw) value (>0.05). Indicator species/taxa in bold. *- denotes values that, although significant, were below 40.

	S1	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**
Spionidae	-	-	-	-	-	-	-	**99.4**
Magelonidae	-	-	-	21.6*	-	**48.4**	-	-
Cirratulidae	-	-	-	-	-	**77.5**	-	-
Opheliidae	-	-	26.1*	**70.4**	-	-	-	-
Glyceridae	-	-	**48**	-	-	-	-	-
Paraonidae	-	-	-	-	-	-	**59.1**	-
Dorvilleidae	-	-	-	-	-	-	**60**	-
Comesoma	-	-	-	39.5*	-	-	-	-
Anomalocardia flexuosa	-	-	-	**47.5**	-	-	-	-
Vitta virginea	-	-	-	-	-	-	**85.1**	-
Astyris lunata	-	-	-	**74.6**	-	-	-	-
Sipuncula	-	-	-	-	-	**52.1**	-	-
Pycnogonida	**40**	-	-	-	-	-	-	-
Cirripedia	-	-	-	-	**60**	-	-	-
Upogebia omissa	-	-	-	-	-	**80**	-	-
"Turbellaria"	-	-	-	-	-	-	-	**43.2**

Polychaeta and Nematoda Trophic Groups

Polychaetes were classified into 11 trophic groups: OMX, ODT, OMJ, SDT, SMT, BMX, CDJ, CMJ, ODJ and OSP, where SDT was the most abundant, accounting for 95.7% of all organisms found. For Polychaeta, the trophic group OMX comprised the families Capittelidae and Paraonidae, ODT included Sabellidae and Oweniidae, OMJ included Nereididae, Glyceridae, Hesionidae, Syllidae, Eunicidae and Dorvilleidae, SDT included Spionidae and Magelonidae, SMT included Cirratulidae, BMX included Opheliidae, Orbiniidae and Sternaspidae, CDJ included Goniadidae, CMJ included Polynoidae and Pilargidae, ODT included Onuphidae and OSP included Chaetopteridae. To group the families into trophic groups, it was also necessary to consult the available bibliography and define the most common group or structure for the family. For

Nematoda, trophic group 1A was composed of the genera *Perspiria* and *Terchellingia*, 1B by *Comesoma*, *Linhomolus*, *Sabatieria*, *Theristus*, *Sphaerolaimus*, *Metadesmolaimus*, *Desmoslaimus*, *Gonioinchus* and *Metalinhomoeus*, 2A by *Acanthonchus*, *Dorylaimopsis*, *Euchromadora*, *Gomphionema*, *Prochromadorella*, *Trissonchulus* and *Phanoderma* and 2B *Eurystomina*, *Oncholaimellus*, *Oncholaimus*, *Viscosia* and *Mesacanthoides*.

Table 4 Relative abundance of Polychaeta and Nematoda trophic groups (TG) distributed in stations (S1 to S8).

TG	S1	S2	S3	S4	S5	S6	S7	S8
BMX	0	0	44.44	61.20	12.50	2.11	12.94	0.01
CDJ	0	0	7.40	9.48	15.0	2.81	3.52	0.05
CMJ	0	0	1.85	1.72	10.0	1.40	0	0
ODJ	1.08	0	0	0	0	4.22	0	0
ODT	0	0	0	2.58	0	2.81	0	0
OMJ	7.60	1.53	9.25	0	0	3.52	8.23	0.01
OMX	67.39	93.84	5.55	0.86	36.25	21.12	50.58	0.02
OSP	0	0	0	0	0	0.70	0	0
SDT	23.91	4.61	25.92	22.41	22.50	38.02	23.52	99.88
SMT	0	0	5.55	1.72	3.75	23.23	1.17	0
1A	3.06	0	0	0	0	0	0	0
1B	15.33	33.33	0	67.81	93.75	61.53	50.0	0
2A	37.42	8.33	0	27.58	6.25	30.76	0	0
2B	44.17	58.33	0	4.59	0	7.69	50.0	0

Regarding the relative abundance of trophic groups for Polychaeta and Nematoda, we obtained the highest values for SDT with 99.8% in S8, OMX and 1B with 93% in S2 and S5, respectively (Table 4).

The nMDS of the Polychaeta and Nematoda trophic groups did not present a pattern as clear as that of the community structure, but we were still able to observe greater proximity between most station replicates (except S8) that were closest to breakwaters' opening to the sea (S3, S4 and S7) and those furthest from the construction site and close to the beach face (S1, S2, S5 and S6) (Fig. 3). Significant differences were also found using a PERMANOVA to compare the beaches (Pseudo-F = 3.9652; p = 0.0022) and the stations (within the beach) (Pseudo-F = 5.6082; p = 0.0001). Applying the a posteriori test, it was possible to detect that the differences occurred between almost all stations, except between S1 and S2 (Pseudo-F= 0.8042; p = 0.6484) and between S3 and S4 (Pseudo-F = 1.353; p = 0.1111), as occurred for taxonomic analysis.

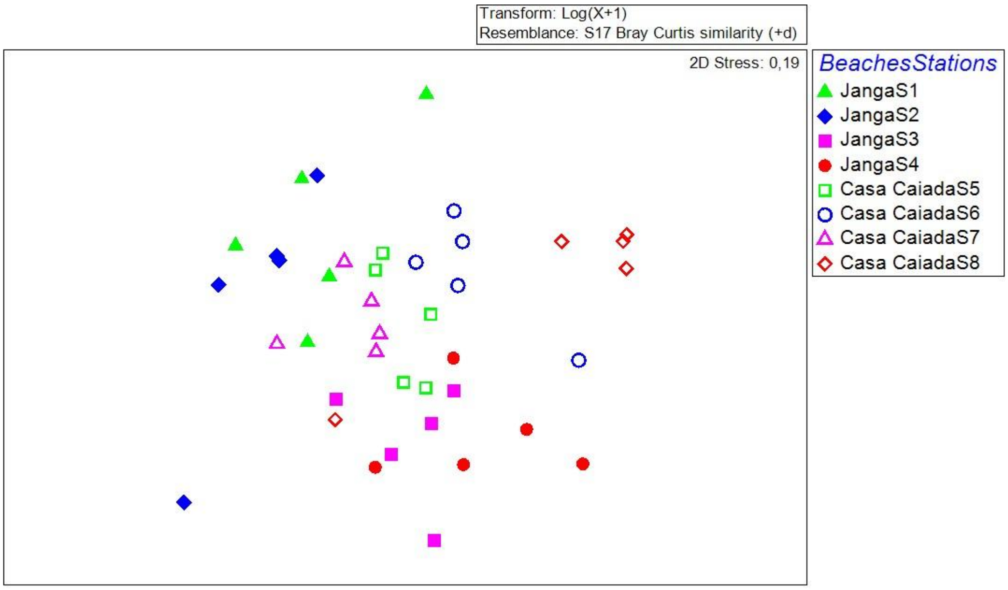

Figure 3. Non-metric multidimensional scaling ordination plot (MDS) of the Polychaeta and Nematoda trophic groups distributed in stations. Stations are indicated as S1, S2, S3, S4, S5, S6, S7 and S8.

Using a SIMPER analysis, it was possible to observe a low similarity concerning the trophic group samples for each of the beaches (Janga and Casa Caiada, 22.4% and 30.1%, respectively), while the dissimilarity between both was high (75.9%). The trophic groups that contributed most to the dissimilarity between beaches were SDT (38.5%), OMX (20.08%), BMX (10.9%), 1B (7.42%) and SMT (5.15%) (Table 5). As for the similarities between the station replicates, on Janga beach the highest similarity occurred in S3 (49.0%), where BMX represented the trophic group with the highest contribution (58.23%) and for Casa Caiada in S6 (58.4%), where SDT represented the group with the highest contribution 51.7% (Table S5). Regarding the dissimilarities between stations for Janga Beach, they were S2 and S4 (95.4%) and for Casa Caiada S6 and S8 (96.0%) (Tables S6 and S7).

Table 5 Average similarity of Polychaeta and Nematoda trophic groups on Janga and Casa Caiada beaches, and the percentage of groups that contributed most to the dissimilarity between beaches.

Janga Average similarity: 22.4		**Casa Caiada Average similarity: 30.1**		**Janga and Casa Caiada Average dissimilarity = 75.9**	
Species	Contrib%	Species	Contrib%	Species	Contrib%
OMX	38.15	SDT	41.24	SDT	38.51
SDT	30.82	OMX	36.28	OMX	20.08
BMX	22.96	BMX	8.34	BMX	10.96
		SMT	4.20	1B	7.42
				SMT	5.15
				CDJ	4.40
				2A	3.40
				2B	3.19

The IndVal coefficient revealed 13 significant indicator trophic groups at Casa Caiada and Janga beaches (Table 6). However, based on the proposed IndVal scale, only 3 Polychaeta trophic groups were above 40%. They were BMX (59.1%) in S4, SMT (42.8%) in S5 and (78.5%) in S6 and SDT (98.5%) in S8.

Table 6 Indval results per station (S1 to S8), including all trophic guilds that had a significant p(raw) value (>0.05). *values that, although significant, were below 40.

	S1	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**
BMX	-	-	-	**59.17**	-	-	-	-
CDJ	-	-	-	-	-	-	-	-
CMJ	-	-	-	-	36.92*	-	-	-
ODJ	-	-	-	-	-	34.29*	-	-
ODT	-	-	-	25.71*	-	34.29*	-	-
OMJ	-	-	-	-	-	-	-	-
OMX	26.7*	21.0*	-	-	12.5*	-	-	-
OSP	-	-	-	-	-	-	-	-
SDT	-	-	-	-	-	-	-	**98.57**
SMT	-	-	-	-	**42.86**	**78.57**	-	-
1A	-	-	-	-	-	-	-	-
1B	-	-	-	31.61*	-	-	-	-
2A	-	-	-	-	-	-	-	-
2B	33.8*	-	-	-	-	-	-	-

Discussion

The history of coastal erosion records in the state of Pernambuco is long-standing, with the first report dating back to 1914 in Olinda, which documented damage caused by the construction of a groin located near the Isthmus of Olinda (FINEP/UFPE, 2009 *apud* Mallmann et al., 2014). However, in the following decades, this issue was not confined to Olinda alone, as it also affected other municipalities, leading to the subsequent construction of numerous coastal defense structures.

In this study, when comparing the two sets of breakwaters, one of the main differences lies in the design of their construction. In Casa Caiada, the breakwaters are spaced at varying distances, either closer to or farther from the shoreline, resulting in openings between the breakwaters that are more perpendicular to the structures and, consequently, to the beach. In contrast, at Janga beach, the breakwaters were built side by side, with openings parallel to the coastline. These openings vary in length, ranging from 40 to 50 meters at Janga and from 70 to 105 meters at Casa Caiada. Another difference is the size of the breakwaters, which range from 125 to 654 meters at Janga and from 495 to 870 meters at Casa Caiada. We also observed differences in the distances of the area in which the structures were built from the shoreline. Janga was built closer to the shoreline (the shortest distance of around 90m and the longest distance 150m) and Casa Caiada was constructed further away (the shortest distance of around 120m and the longest distance of around 240m). The differences in the arrangement of the structures and the distances between them have influenced sediment retention and deposition in these areas. Although the structures at Janga are more recent, they have intensified the process of sediment redistribution (erosion/deposition), resulting in more points with the formation of tombolos and salients. Casa Caiada beach, despite having older structures, exhibiting sediment deposition with the formation of several sandbanks visible at low tide, did not develop the formation of complete tombolos. It is known that the presence of artificial defense structures generally affects local hydrodynamic regimes, potentially leading to changes in sedimentation rates, increased siltation, and organic enrichment (Zyserman et al., 2005; Carugati et al., 2018). These changes result in alterations in the physical parameters of the sediment, influencing grain size, which, in turn, affects the composition and structure of macrofaunal assemblages (Martin et al., 2005).

The most abundant group found on both beaches was Polychaeta, which is the group that generally contributes the most to the diversity and abundance of benthic communities, often representing more than half of the organisms in mobile bottom habitats (Olsgard et al., 2003). This dominance is influenced by several characteristics of the group, including their tolerance to salinity variations and their ability to survive in low-oxygen environments, as typically occurs in depositional environments with fine and eutrophic substrates (Glasby et al., 2021). Our granulometric analyses indicated that both beaches had a significant presence of fine sediments, which could explain the high dominance of these organisms.

At Janga beach, a high abundance of Polychaeta Capitellidae (in S1 and S2) and Opheliidae (S3 and S4), the Nematoda *Comesoma* (in S4), *Oncholaimus* and *Euchromadora* (in S1) and the Gastropoda *A. lunata* (in S4) were observed. S1 and S2 were characterized as areas with muddy, dark and viscous sediment, with a large volume of algae. Thus, the presence of Capitellidae which presents a greater representation in fine substrates, characterized by high organic matter content, such as muddy sediments, and *Euchromadora* and *Oncholaimus* that were already found in artificial hard substrates as well as in *Sargassum* banks, were expected (Fonsêca-Genevois et al., 2006; Blake, 2009). S3 and S4 are areas with more sandy sediment, and are consequently more aerated, with greater water flow due to the proximity to the opening between two breakwaters. Thus, the presence of Opheliidae was expected, since this family is frequently associated with aerated substrates with low concentrations of organic matter (Hartmann-Schröder, 1996). The presence of *Comesoma*, was also expected as, due to its feeding habit as a non-selective deposit feeder, it is found in greater abundance in somewhat disturbed sediments (Ingels et al., 2011). *A. lunata* is a micrograzer, common in seagrass ecosystems and a facultative carnivore on tunicates, which has also been reported to inhabit breakwaters (Cote et al., 2001; Locke et al., 2007).

At Casa Caiada beach, a high density of small Spionidae (mainly *Streblospio*) was found in four replicates of S8, which we associated with the local recruitment of these organisms, resulting in densities exceeding 120,000 individuals per square meter - unusual values for sandy beaches. S8 was located in a sandy area closer to the opening between two breakwaters with higher water flow and near remnants of sandstone reefs. The hydrodynamics near the breakwater sampling site may also favor larval retention and recruitment. Spionids exhibit a variety of reproductive and larval development patterns, including benthic and planktonic development (Blake and Arnofsky, 1999). They primarily inhabit soft-bottom habitats such as mud and sand in coastal waters (Blake and Arnofsky, 1999; Dagli et al., 2011). It should be highlighted that *Streblospio* species commonly present seasonal peaks of juveniles and this strategy is considered an adaptative strategy of opportunistic polychaetes to accommodate environmental variability (Lardici et al., 1997). Spionids are also known as opportunistic colonizers of disturbed habitats, as they exhibit rapid responses to changes in hydrodynamics, disturbances and the availability of organic resources (Grassle and Grassle, 1974; Omena et al., 2012). The physiological tolerance of Spionidae to mechanical stress conditions, combined with reproductive strategies of high fecundity and synchronized larval dispersal, suggests that the environment modified by the breakwater may act as an ecological filter, selecting r-strategist species (Bulleri and Chapman, 2010; Shimabukuro et al., 2016).

Considering the community structure, the MDS indicated a separation between groups, mainly highlighting stations collected at greater proximity to each other, as well as stations closest to breakwater openings compared to those farthest from them and closer to the shoreline for both beaches. Through statistical tests, we were able to observe that stations S1 and S2 did not present significant differences, nor did S3 and S4 taxonomically and functionally. In the case of Janga beach, what we can infer is that the presence of the tombolo, which laterally closes to one side of the beach, may have caused stations in each area to become more homogeneous, and more heterogeneous between areas (stations located on the tombolo and stations close to opening of the breakwaters). On the contrary, in Casa Caiada, despite sedimentation and formation of sandbanks, significant differences were found between all stations, which could indicate greater heterogeneity of the area. This phenomenon of environmental homogenization has already been described by McKinney (2006), who associated urbanization as a major cause of biotic homogenization, which can cause changes in species composition, which in turn, can lead to the functional homogenization of systems.

The climatic similarity of 2022 and 2023 (Fig. S1) increases our confidence that macrofauna community differences between beaches are the result of the impact of differently designed breakwaters on beach morphology and sediment composition and are not related to interannual variability. A study by Menezes (2019) on Brazilian tropical sandy beach midlittoral macrofauna, carried out over three years, sampling rainy and dry periods, showed that samples from each season group together, independent of the year. Regarding the differences between the beaches, the taxa that contributed the most to the dissimilarities were Spionidae, due to their high abundance at Casa Caiada in S8. The Gastropoda *V. virginea* was also found in high abundance at Casa Caiada, particularly in S7, an area where large sandbanks with high densities of this gastropod are visible at low tide. Tan and Clements (2008) reported that populations of most marine Neritidae, which includes *V. virginea*, have rapidly colonized artificial habitats such as breakwaters and seawalls. The Polychaeta Capitellidae, also contributed to dissimilarities between beaches, with high abundance at Janga beach, especially in S1 and S2. These stations are in an area dominated by viscous, very dark muddy sand with a strong odor, indicative of an anoxic or hypoxic sediment.

Through the Indval analysis, it was possible to identify a series of indicator taxa that may reflect changes in the composition of the benthic macrofauna on the beaches of Olinda and Janga. It was observed that in Casa Caiada, twice as many taxa were considered indicators when compared to Janga, further supporting the hypothesis of higher homogeneity in the later beach. An indicator taxon must have high fidelity within the assessed ecological state and only species with IndVal values above 70% are characteristic or indicative of the ecological state of the studied environment. Species with significant values that are lower than 70%, are considered detector taxa (Dufrêne and Legendre, 1997; Mcgeoch et al., 2002). These detector taxa are potential indicators of habitat change because they can change their preferred habitat more quickly than indicator species (Van Rensburg et al., 1999). Some of these species are associated with opportunistic behavior. Among the taxa that presented the highest values was Spionidae, which, as previously mentioned, is known for being opportunistic, and *V. virginea,* which is known to feed mainly on algae and diatoms (Matthews-Cascon and Martins, 1999). It is possible that this species may have benefited from the environment formed in Casa Caiada, where the sandy or muddy-sand banks are partly covered with macroalgae patches (*Ulva*, *Halodule*, *Bryopsis*, *Caulerpa*, and *Cladophora*) during part of the year and are completely covered from January to May (Marinho et al., 2023).

In non-impacted marine regions, the trophic groups of benthic macroinvertebrates exhibit a balanced functional structure, characterized by the dominance of organisms specialized in the processing of stable resources, such as deposit feeders and suspension feeders (Jumars et al., 2015). Concomitantly, there is a greater occurrence of nematodes classified as selective deposit feeders (1A), which exhibit efficiency in the assimilation of sedimentary organic compounds under low disturbance conditions. The reduction in the representation of non-selective nematodes or predators/omnivores (1B and 2B) is generally correlated with environmental stability, where competition for resources is minimized and trophic niches become more defined (Schratzberger and Ingels, 2018). This pattern reflects the reduction of environmental stress, allowing the coexistence of guilds with marked functional specialization, a characteristic of undisturbed benthic systems (Jumars et al., 2015). With regard to the trophic groups between the stations, we can observe that just like the community structure in S1 and S2 (stations most impacted by the formation of the tombolo), no significant differences were found, with a predominance of omnivorous organisms (OMX and 2B), considered less specialized. Likewise, S3 and S4 (closest to the opening of the breakwaters) did not present significant differences, with the prevalence of deposit feeders (BMX, SDT and 1B), that is, more specialized organisms. Stations S5, S6, S7 and S8 were significantly different, where S5 was more distributed in terms of the relative abundance of organisms among the trophic groups, with the highest percentages of 1B, OMX, SDT and CDJ. In S6 it is also possible to observe a greater distribution of organisms among the trophic groups, with a predominance of deposit feeders and epistrate scrapers (SDT, SMT, 1B and 2A). As for S7, there was a predominance of omnivorous and deposit feeders (OMX, SDT, 1B and 2B), while in S8 there was a high dominance of the surface deposit feeder (SDT). The trophic pattern was broadly similar to the community structure pattern, indicating that the changes in the community structure along the environmental gradients generated by tombolo formation and mud sediment accumulation also resulted in changes in the ecological functioning of the trophic groups in the areas.

In general, both Janga and Casa Caiada face issues resulting from the installation of breakwaters. Despite being built later, the way the breakwaters were constructed at Janga Beach caused a greater impact than at Casa Caiada, which was built earlier. Nevertheless, for both beaches, we can observe that following the alterations caused by the installation of the structures, both communities are impoverished and less structured, both in terms of taxon composition and the trophic groups present, resulting in a greater composition of opportunistic organisms. The characterization and monitoring of beaches with existing defense structures are of great importance, as the information obtained from these environments can be used to assist in decisions regarding the type of structure, materials used, arrangement, extent, and other important characteristics that may be applied in future defense structures.

In conclusion, the presence of coastal defense structures significantly affected the composition of benthic macrofauna on the beaches of Olinda and Janga, creating a gradient of conditions that influenced the distribution and abundance of opportunistic indicator taxa. These results highlight the importance of considering the impacts of coastal structures on benthic biodiversity and sediment dynamics, providing support for the management and conservation of these ecosystems.

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Statements and Declarations

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Author contributions

NCMM and PJPS contributed to Conceptualization, Formal analysis, and Investigation; NCMM contributed to Writing- original draft preparation and Methodology; MCFL and LBC contributed to Writing- review and editing; CCCC, EGS, PFN contributed to investigation; AME contributed to resources and investigation; PJPS contributed to review, Funding acquisition and Supervision. All authors have read and agreed to the published version of the manuscript.

Co-author declarations

All co-authors confirm that they have read, fully agree with the manuscript's content and approved its final version for submission.

Availability of data and material

Data will be made available upon reasonable request.

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Design differences between two sets of breakwaters have different effects on the benthic macrofauna of sandy beaches

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Supplementary Material

Figure S1. Pluviometry for the years 2022 and 2023 and historical average (avg) of 30 years for the cities of Paulista (Pa) and Olinda (Ol).

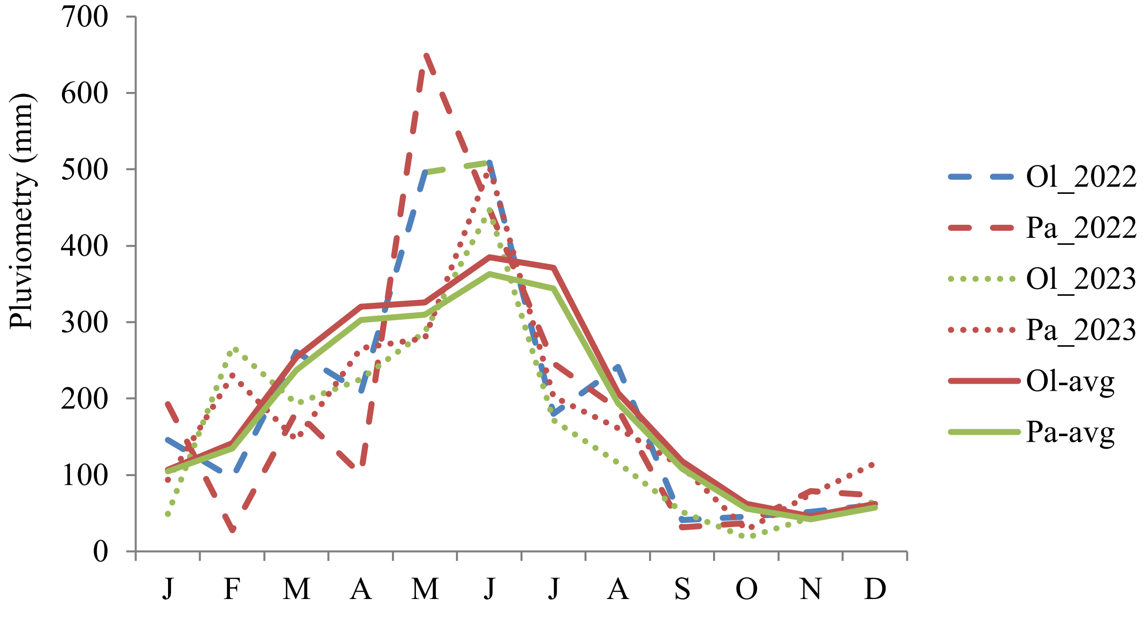

Table S1. Breakwater lengths and opening lengths of Janga and Casa Caiada beaches.

Janga			Casa Caiada		
Breakwater	**(m)**	**Opening (m)**	**Breakwater**	**(m)**	**Opening (m)**
1	125	45	1	567	85
2	255	50	2	870	80
3	205	50	3	578	105
4	245	50	4	860	70
5	315	45	5	500	80
6	245	40	6	495	70
7	185	40	7	500	
8	270	50			
9	654				

Figure S2. Location of the set of breakwaters on Janga and Casa Caiada beaches. The numbers indicate the numbering of each breakwater, from south to north.

Table S2. Average similarity of each station of Janga and Casa Caiada beaches and the percentage of groups that contributed most to this similarity.

Janga S1		**Janga S2**		**Janga S3**		**Janga S4**	
Average similarity: 25.9		Average similarity: 33.8		Average similarity: 40.7		Average similarity: 28.9	
Species	Contrib%	Species	Contrib%	Species	Contrib%	Species	Contrib%
Capitellidae	73.03	Capitellidae	94.04	Opheliidae	40.95	Opheliidae	30.35
Spionidae	12.73			Bivalvia	24.28	*A. lunata*	25.29
Ampitoidae	3.67			Spionidae	9.5	Magelonidae	15.44
Bivalvia	3.42			Magelonidae	7.2	Bivalvia	12.09
				Cirratulidae	4.28	*A. flexuosa*	7.29
				Glyceridae	4.11		
Casa Caiada S5		**Casa Caiada S6**		**Casa Caiada S7**		**Casa Caiada S8**	
Average similarity: 39.4		Average similarity: 54.0		Average similarity: 55.7		Average similarity: 48.6	
Species	Contrib%	Species	Contrib%	Species	Contrib%	Species	Contrib%
Capitellidae	27.03	Magelonidae	34.34	*V. virginea*	71.04	Spionidae	99.7
Magelonidae	24.27	Cirratulidae	16.56	Capitellidae	11.45		
Bivalvia	11.74	Bivalvia	11.97	Paraonidae	7.48		
Orbiniidae	7.73	Sipuncula	11.15	Orbiniidae	3.15		
Goniadidae	6.26	Capitellidae	7.7				
Sipuncula	4.75	Spionidae	4.27				
Comesoma	4.23	*U. omissa*	3.78				
V. virginea	3.35	"Turbellaria"	2.3				
Polynoidae	3.31						

Table S3. Average dissimilarity on the stations of Janga beach and the percentage of groups that contributed most to this dissimilarity.

S1 and S2		**S1 and S3**		**S1 and S4**	
Average dissimilarity = 66.6		Average dissimilarity = 88.8		Average dissimilarity = 94.8	
Species	Contrib%	Species	Contrib%	Species	Contrib%
Capitellidae	36.62	Capitellidae	25.43	Capitellidae	15.28
Oncholaimus	11.28	Opheliidae	12.53	*A. lunata*	13.86
Spionidae	9.58	*Oncholaimus*	6.42	Opheliidae	13.2
Euchromadora	7.75	Spionidae	6.35	*Comesoma*	7.49
Bivalvia	3.48	*Euchromadora*	5.82	Magelonidae	5.76
Comesoma	3.1	Magelonidae	3.89	*Oncholaimus*	5.2
Theristus	2.96	Bivalvia	3.78	Euchromadora	4.68
Ampitoidae	2.88	Nemertea	2.71	*A. flexuosa*	4.56
Sabatieria	2.83	*A. lunata*	2.2	Bivalvia	3.67
Oligochaeta	2.59	Glyceridae	2.1	Orbiniidae	3.6
Cheiriphotis	2.26	Sabatieria	2.08	Spionidae	3.47
Atylidae	2.21	Goniadidae	2.05	*Dorylaimopsis*	2.72
Eurystomina	1.71	*Theristus*	1.98	Goniadidae	2.4
Linhomoeus	1.62	*Cheiriphotis*	1.95	*Sabatieria*	1.36
		Paguroidea	1.91	*Theristus*	1.3
		Ampitoidae	1.79	*Eurystomina*	1.03
		Atylidae	1.69	*Linhomoeus*	0.94
		Cirratulidae	1.69		
		Comesoma	1.68		
		Eurystomina	1.28		
		Linhomoeus	1.19		

S2 and S3		**S2 and S4**		**S3 and S4**	
Average dissimilarity = 92.3		Average dissimilarity = 96.5		Average dissimilarity = 71.6	
Species	Contrib%	Species	Contrib%	Species	Contrib%
Capitellidae	34.96	Capitellidae	20.0	*A. lunata*	21.18
Opheliidae	16.39	*A. lunata*	16.94	Opheliidae	15.84
Bivalvia	6.36	Opheliidae	16.02	*Comesoma*	10.39
Magelonidae	5.32	*Comesoma*	8.2	Magelonidae	7.58
Spionidae	4.44	Magelonidae	7.09	*A. flexuosa*	7.11
Nemertea	3.75	*A. flexuosa*	5.57	Orbiniidae	6.19
Oncholaimus	2.99	Bivalvia	5.26	Bivalvia	4.43
A. lunata	2.87	Orbiniidae	4.57	*Dorylaimopsis*	4.11
Glyceridae	2.73	*Dorylaimopsis*	3.05	Goniadidae	3.95
Goniadidae	2.64	Goniadidae	2.9	Spionidae	3.14
Paguroidea	2.58	*Oncholaimus*	2.1	Nemertea	2.25
Oligochaeta	2.3			Glyceridae	1.88
Cirratulidae	2.23			Paguroidea	1.82
Atylidae	1.38			Capitellidae	1.49

Table S4. Average dissimilarity on the stations of Casa Caiada beach and the percentage of groups that contributed most to this dissimilarity.

S5 and S6		**S5 and S7**		**S6 and S7**	
Average dissimilarity = 67.0		Average dissimilarity = 78.0		Average dissimilarity = 83.8	
Species	Contrib%	Species	Contrib%	Species	Contrib%
Magelonidae	13.63	*V. virginea*	38.23	*V. virginea*	30.58
Cirratulidae	11.91	Capitellidae	8.33	Magelonidae	11.21
Capitellidae	10.03	Paraonidae	6.62	Cirratulidae	8.61
Sipuncula	9.15	Sipuncula	5.67	Bivalvia	6.9
Bivalvia	6.51	Magelonidae	4.95	Capitellidae	6.54
Goniadidae	5.09	Spionidae	4.83	Sipuncula	6.49
Comesoma	4.83	Goniadidae	4.61	Paraonidae	4.19
Orbiniidae	3.5	Comesoma	4.39	Spionidae	3.42
U. omissa	3.34	Bivalvia	3.47	Orbiniidae	2.54
Onuphidae	3.0	Orbiniidae	3.37	*U. omissa*	2.25
Polynoidae	2.86	Polynoidae	2.52	Onuphidae	1.96
Paraonidae	2.68	*A. flexuosa*	1.82	"Turbellaria"	1.63
V. virginea	2.47	Cirripedia	1.78	*A. brasiliana*	1.33
"Turbellaria"	2.39			*Dorvilleidae*	1.2
Spionidae	2.38			*Comesoma*	1.18
A. flexuosa	2.17				
Cirripedia	2.0				
Sabellidae	1.01				
Dorylaimopsis	0.92				
Phanoderma	0.92				
S5 and S8		**S6 and S8**		**S7 and S8**	
Average dissimilarity = 95.1		Average dissimilarity = 97.6		Average dissimilarity = 94.8	
Species	Contrib%	Species	Contrib%	*Species*	Contrib%
Spionidae	83.34	Spionidae	80.32	Spionidae	83.69
Capitellidae	2.91	Magelonidae	4.22	*V. virginea*	8.7
Magelonidae	1.99	Cirratulidae	2.7		
Sipuncula	1.82	Bivalvia	2.11		
		Sipuncula	1.96		

Table S5. Average similarity on the trophic groups of Janga and Casa Caiada beaches and the percentage of groups that contributed most to this similarity.

Janga S1		**Janga S2**		**Janga S3**		**Janga S4**	
Average similarity: 27.3		Average similarity: 35.6		Average similarity: 49.0		Average similarity: 33.3	
Species	Contrib%	Species	Contrib%	Species	Contrib%	Species	Contrib%
OMX	80.97	OMX	96.4	BMX	58.23	BMX	54.59
SDT	14.76			SDT	25.92	SDT	35.74
				OMJ	6.7		
Casa Caiada S5		**Casa Caiada S6**		**Casa Caiada S7**		**Casa Caiada S8**	
Average similarity: 50.4		Average similarity: 58.4		Average similarity: 53.5		Average similarity: 48.6	
Species	Contrib%	Species	Contrib%	Species	Contrib%	Species	Contrib%
SDT	30.4	SDT	51.7	OMX	62.57	SDT	99.94
OMX	29.91	SMT	22.28	SDT	23.06		
BMX	13.25	OMX	16.35	BMX	10.61		
1B	11.37						
CDJ	6.95						

Table S6. Average dissimilarity on the trophic groups of Janga beach and the percentage of groups that contributed most to this dissimilarity.

S1 and S2		**S1 and S3**		**S1 and S4**	
Average dissimilarity = 64.7		Average dissimilarity = 85.1		Average dissimilarity = 87.2	
Species	Contrib%	Species	Contrib%	Species	Contrib%
OMX	45.59	OMX	33.77	BMX	25.43
2B	15.27	BMX	19.33	OMX	24.37
SDT	13.07	SDT	10.1	1B	14.34
1B	11.48	2B	9,.5	SDT	9.57
2A	9.99	1B	8.05	2A	9.32
		2A	7.36	2B	8.4
		OMJ	4.34		
S2 and S3		**S2 and S4**		**S3 and S4**	
Average dissimilarity = 93.1		Average dissimilarity = 95.4		Average dissimilarity = 62.0	
Species	Contrib%	Species	Contrib%	Species	Contrib%
OMX	43.17	OMX	29.78	BMX	37.58
BMX	24.46	BMX	29.71	1B	16.61
SDT	13.74	SDT	14.78	SDT	16.55
OMJ	4.94	1B	11.34	CDJ	6.9
2B	3.63	2A	4.11	2A	6.01
CDJ	3.45	CDJ	3.69	OMJ	4.99
				OMX	3.36

Table S7. Average dissimilarity on the trophic groups of Casa Caiada beach and the percentage of groups that contributed most to this dissimilarity.

S5 and S6		**S5 and S7**		**S6 and S7**	
Average dissimilarity = 62.1		Average dissimilarity = 51.7		Average dissimilarity = 58.2	
Species	Contrib%	Species	Contrib%	Species	Contrib%
SDT	23.34	OMX	28.96	SDT	24.38
SMT	18.59	1B	14.6	SMT	21.98
OMX	15.6	SDT	13.55	OMX	18.62
1B	8.49	CDJ	12.89	BMX	6.97
CDJ	7.64	BMX	9.23	OMJ	5.46
BMX	5.85	CMJ	8.74	ODJ	4.83
CMJ	5.32	OMJ	7.09	1B	4.72
ODJ	4.35			2A	3.43
2A	3.4				
S5 and S8		**S6 and S8**		**S7 and S8**	
Average dissimilarity = 93.1		Average dissimilarity = 96.0		Average dissimilarity = 93.9	
Species	Contrib%	Species	Contrib%	Species	Contrib%
SDT	87.45	SDT	87.75	SDT	87.98
OMX	4.16	SMT	3.99	OMX	7.59

7 CONSIDERAÇÕES FINAIS

Esta tese, estruturada em artigos interligados, traça um arco investigativo que se inicia na síntese crítica do conhecimento global e se encerra na validação através de amostragens.

O primeiro capítulo, fundamentado em revisão sistemática, bibliométrica e meta-análise, não apenas mapeou o estado da arte sobre os impactos de estruturas de defesa costeira (EDC) sobre a macrofauna bentônica, mas estabeleceu as bases metodológicas que orientaram toda a pesquisa subsequente. Ao analisar criticamente duas décadas de produção científica (2002-2024), revelou-se um crescimento exponencial de estudos após 2015, sinalizando a urgência global do tema. Identificou-se que os quebra-mares de concreto dominam as intervenções costeiras (68% dos casos), e que 65,3% das pesquisas reportam impactos negativos significativos, particularmente na abundância/densidade da macrofauna, cuja magnitude de efeito supera em 54% os danos à riqueza taxonômica. Esta revisão expôs ainda lacunas especialmente no que diz respeito a informações muito importantes. Por exemplo, a maioria dos artigos (40%) não menciona o material utilizado na construção das estruturas de defesa, pouco mais de 1/3 dos trabalhos se concentrou nos efeitos das estruturas sobre os organismos que vivem no substrato inconsolidado, e dados abióticos importantes estavam ausentes na maioria dos estudos.

O segundo capítulo realizou uma avaliação pioneira no litoral de Pernambuco, contrastando nove áreas representando áreas com EDCs, áreas com recifes como proteção natural e praias expostas. Os resultados validaram os padrões globais identificados na revisão, mas com nuances críticas: as EDCs artificiais promoveram sedimentos finos e comunidades dominadas por espécies oportunistas, refletindo qualidade ecológica comprometida. Embora as métricas de riqueza e abundância tenham sugerido maior diversidade nos habitats com EDCs, o índice de qualidade ambiental AMBI revelou uma diversidade funcional reduzida, dominada por espécies adaptadas à degradação ambiental. A análise de espécies indicadoras (IndVal) identificou táxons indicadores exclusivos de quebra-mares, confirmando que as estruturas de defesa costeira geram microhabitats únicos com comunidades bentônicas homogeneizadas Em contraste, praias naturalmente expostas e áreas protegidas por recifes apresentam qualidade ecológica superior (AMBI), apesar da menor abundância e riqueza, indicando que baixas densidades ou riqueza de organismos em ambientes de alta energia refletem restrições físicas ambientais e não problemas de qualidade ambiental. Neste capítulo também destacamos o uso do potencial redox (ORP) como um preditor robusto da qualidade ambiental.

O terceiro capítulo, por sua vez, respondeu à necessidade de discriminar como características específicas de projeto das EDCs modulam seus impactos. Ao comparar o conjunto de quebra-mares do Janga e de Casa Caiada, revelou-se que decisões de engenharia determinam destinos ecológicos distintos. No Janga, a configuração mais próxima à costa e com menor distância entre as estruturas, apesar de mais recente, induziu formação de tômbolos, causando isolamento sedimentar e homogeneização biológica. Em contraste, o desenho paralelo mais afastado e com maior distância entre si de Casa Caiada teve menor impacto, quando comparado ao Janga. Este trabalho evidencia que estruturas de defesa costeira reconfiguram profundamente os ecossistemas bentônicos, e embora seus impactos negativos sejam predominantes, demonstra-se que estruturas ecologicamente projetadas podem mitigar danos, reforçando a urgência de novas pesquisas que, com protocolos padronizados e monitoramento de longo prazo, quantifiquem de forma robusta a resiliência ecológica em áreas que já sofreram intervenções.

Os resultados obtidos ao longo desta tese fornecem subsídios científicos relevantes para aprimorar o planejamento e a gestão de estruturas de defesa costeira, contribuindo para uma abordagem mais sustentável na junção entre engenharia e ecossistemas marinhos. De modo geral, as evidências demonstraram que as estruturas artificiais modificam de forma significativa as condições sedimentares e a composição da macrofauna bentônica, resultando em ambientes biologicamente mais homogêneos e com menor qualidade ecológica. No entanto, observou-se também que o impacto dessas obras pode ser substancialmente mitigado quando aspectos de projeto são ecologicamente orientados e baseados em parâmetros ambientais locais.

Os achados deste trabalho indicam que a altura das estruturas de defesa é um fator crítico na determinação da qualidade ambiental. Obras excessivamente elevadas, com alturas muito superiores às dos recifes naturais comuns nessas regiões, impedem a passagem da água sobre sua superfície durante a maré alta, reduzindo a renovação hídrica e favorecendo o acúmulo de sedimentos finos. Essa condição limita a oxigenação do substrato e favorece a dominância de espécies oportunistas, características de ambientes degradados. Assim, recomenda-se que futuras intervenções considerem a altura média dos recifes naturais de áreas próximas como referência para o dimensionamento das estruturas, de forma a minimizar o isolamento hidrodinâmico e os efeitos negativos sobre a macrofauna.

Outro aspecto determinante refere-se à distância das estruturas em relação à linha de costa. Os resultados demonstraram que obras construídas muito próximas à praia, como observado na praia do Janga, intensificam o isolamento sedimentar, favorecem a formação de tômbolos e resultam em comunidades bentônicas empobrecidas e homogeneizadas. Por outro lado, estruturas posicionadas mais afastadas da costa, como em Casa Caiada, apresentaram menor impacto ecológico. Dessa forma, recomenda-se que futuras obras de defesa costeira sejam construídas considerando, em conjunto com a altura da estrutura, a distância à linha de praia, permitindo maior troca entre o ambiente marinho e o sistema praial e reduzindo os efeitos de homogeneização biológica.

O espaçamento entre as estruturas também é considerado um fator determinante para a manutenção da circulação de água e sedimentos. Estruturas muito próximas entre si, sem aberturas adequadas, limitam o fluxo transversal e agravam o acúmulo de material fino, enquanto arranjos mais espaçados, com aberturas amplas entre os blocos, favorecem a renovação hídrica e melhoram a qualidade ambiental. Assim, sugere-se que as futuras obras sejam planejadas de forma a manter intervalos maiores entre as estruturas, permitindo o fluxo contínuo da água e o transporte de sedimentos, evitando o bloqueio lateral característico de projetos contínuos.

Além desses parâmetros relacionados à construção em si, destaca-se a importância de integrar a engenharia costeira à ecologia marinha desde as etapas iniciais de planejamento. Recomenda-se que o desenvolvimento de novas obras seja precedido por avaliações ambientais detalhadas e pelo uso de protocolos padronizados de monitoramento ecológico, considerando tanto variáveis físicas, como a granulometria, quanto biológicas, como índices de qualidade ambiental. O uso contínuo desses indicadores permitirá não apenas a mitigação de impactos, mas também a quantificação da resiliência ecológica em áreas já modificadas.

Em síntese, a adoção de projetos ecologicamente orientados, com dimensões compatíveis às formações recifais naturais, maior afastamento da costa e espaçamento adequado entre estruturas, representa um caminho promissor para conciliar a proteção costeira com a conservação da biodiversidade marinha. A integração efetiva entre ciência e gestão pública é, portanto, essencial para o desenvolvimento de soluções de engenharia costeira mais resilientes, sustentáveis e compatíveis com a dinâmica natural dos ecossistemas.

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