UNIVERSIDADE FEDERAL DO RIO GRANDE DO NORTE

PROGRAMA DE PÓS-GRADUAÇÃO EM ECOLOGIA

Thayná Jeremias Mello

Impactos locais e globais sobre ambientes marinhos do Arquipélago de Fernando de Noronha

NATAL – RN

2023

Thayná Jeremias Mello

Impactos locais e globais sobre ambientes marinhos do Arquipélago de Fernando de Noronha

Tese submetida ao programa de Pós-Graduação em Ecologia da Universidade Federal do Rio Grande do Norte para a obtenção do Título de Doutora em Ecologia.

Orientador:

Dr. Guilherme Ortigara Longo

Coorientadora:

Dra. Cybelle Menolli Longhini

NATAL – RN

2023

Universidade Federal do Rio Grande do Norte - UFRN
Sistema de Bibliotecas - SISBI
Catalogação de Publicação na Fonte. UFRN - Biblioteca Central Zila Mamede

Mello, Thayná Jeremias.
Impactos locais e globais sobre ambientes marinhos do Arquipélago de Fernando de Noronha / Thayná Jeremias Mello. - 2023.
153 f.: il.

Tese (doutorado) - Universidade Federal do Rio Grande do Norte, Centro de Biociências, Programa de Pós-Graduação em Ecologia,, Natal, RN, 2023.
Orientador: Prof. Dr. Guilherme Ortigara Longo.
Coorientadora: Profa. Dra. Cybelle Menolli Longhini.

1. Recifes - Tese. 2. Unidades de conservação - Tese. 3. Poluição - Tese. 4. Turismo - Tese. 5. Mudanças climáticas - Tese. 6. Ilhas oceânicas - Tese. I. Longo, Guilherme Ortigara. II. Longhini, Cybelle Menolli. III. Título.

RN/UF/BCZM CDU 574.5

Elaborado por Ana Cristina Cavalcanti Tinoco - CRB-15/262

AGRADECIMENTOS

Este trabalho é fruto de observação, vivência e aprendizado construídos ao longo de uma jornada profissional e pessoal em Fernando de Noronha, e por isso agradeço a todos que fizeram parte dessa trajetória que vai além dos quatro anos do Doutorado.

Minha primeira ida ao Arquipélago em 2009 me deixou eufórica e instigou a vontade de entender a natureza dessa ilha oceânica e como a interferência humana afeta seu funcionamento, o que me levou ao meu trabalho de Mestrado em 2011 a 2013. Por conta dele, fui convidada para trabalhar como Coordenadora de Pesquisa do NGI Noronha, e vivi a experiência de ser moradora da ilha de 2014 a 2018. Cada mergulho e cada chance de vivenciar a sazonalidade do arquipélago foi uma grande escola. Como analista ambiental do ICMBio tive a chance de interagir com inúmeros grupos de pesquisa, e de aprender com cada pesquisador com quem tive contato. Ao mesmo tempo, as dificuldades, limitações e conflitos de ser moradora da ilha me fizeram deixar a euforia um pouco de lado, e buscar uma visão crítica e abrangente sobre a importância de conservar o Arquipélago e os obstáculos para isso. Encarar o desafio do Doutorado teve a ver a vontade de registrar e consolidar algumas das percepções e ideias que surgiram ao longo deste tempo.

Agradeço a todos os colegas do ICMBio Noronha, pelo acolhimento, suporte e trocas de experiência. Agradeço especialmente ao Ricardo Araújo pelas portas sempre abertas, ao Eduardo Macedo pela confiança, e a Carla Guaitanele pelo apoio a valorização do trabalho. Agradeço à Viviane Vilella pela amizade em todos os momentos, e aos voluntários que viraram amigos, Débora Gutierrez, Tati Micheletti, Francielly Furlan e Ana Carolina Grillo.

Agradeço aos pesquisadores com quem convivi neste período pela disponibilidade em compartilhar seus conhecimentos e receptividade durante o acompanhamento dos trabalhos de campo e laboratório, especialmente ao Paulinho, Thali, Tati e Ricardo Pescador Krul; Patrícia Serafini e Carlos Feliz Abrahão; Cadu Ferreira, Guilherme, Ronaldo e Sérgio; e Guilherme Nunes pela amizade e oportunidades de colaboração.

Agradeço aos participantes do Encontro de Pesquisa realizado em 2015, evento em que tive contato com várias das questões que levaram à construção da tese anos depois.

Agradeço ao Ronaldo Francini Filho pela palestra chamando a atenção para os impactos do esgoto nos ambientes recifais. Nunca mais consegui colocar detergente na esponja sem pensar que eu estava matando um coral, mas entendi o quanto esse assunto era negligenciado e a necessidade de dar atenção a ele em Noronha.

Agradeço aos amigos que foram minha família na ilha ao longo destes anos, e tornaram a vida muito mais leve e alegre. Tive a sorte e o privilégio de ter como amigos alguns dos maiores conhecedores sobre a vida marinha, o mergulho e o mar de Noronha, conhecimento que sempre compartilharam comigo com muito entusiasmo. Alice, Zaira, Lola, Marenga e Ismael, sou muito grata a vocês pela amizade, cuidado e generosidade.

Agradeço ao ICMBio por valorizar a qualificação dos seus servidores e mais uma vez investir na minha capacitação, me concedendo o afastamento para cursar o Doutorado.

Agradeço ao meu orientador, Guilherme Longo por ter me acolhido no Laboratório de Ecologia Marinha (LECOM) e aceitado me orientar mesmo me conhecendo há pouco tempo.

A combinação Doutorado + bebê + pandemia + mudança de PE para PB para SP trouxe desafios que muitas vezes me fizeram muitas vezes pensar que seria impossível conseguir concluir o trabalho, e algumas vezes pensar em desistir. Agradeço à minha mãe, que se desdobrou no papel de vovó Sandra todas as vezes em que eu apertei o "botão pânico" e tornou possível que eu conseguisse trabalhar em momentos cruciais, e ao meu irmão Christian pelo apoio e disposição em ajudar. Agradeço ao Guilherme por sempre dizer "entendo seu pânico, mas tenho uma solução e vai dar certo". E sempre correr atrás para fazer dar certo quando eu não conseguia. Agradeço à Ana Grillo por toda paciência e parceria na elaboração do artigo do Capítulo 1, e por manter o trabalho andando quando eu mal podia ligar o computador. Agradeço ao Edson Vieira pelo suporte, amizade e por me ajudar a organizar e fazer acontecer o Capítulo 2 quando a bagunça tomava conta da minha vida.

Agradeço à Coordenação do PPGECO, especialmente à Juliana Dias e ao Guilherme, por toda a compreensão, apoio e proatividade para garantir que eu pudesse conciliar as demandas da vida pessoal com o Doutorado da melhor forma possível.

Agradeço a todos que fazem parte do LECOM. Num ambiente tão competitivo como é a academia, é inspirador ver um laboratório onde o respeito, solidariedade e companheirismo são prioridades. Mesmo que boa parte da nossa convivência tenha sido virtual, fazer parte de um ambiente inclusivo e acolhedor como o do LECOM fez toda a diferença nestes tempos conturbados. Destaco a importância da liderança e exemplo do Gui em promover este ambiente, e a minha sorte por ele ter um sido acima de tudo amigo, além de orientador nesta fase.

Agradeço à Cybelle, minha coorientadora, que surgiu como se fosse um pedido atendido pelo gênio da lâmpada, e trouxe as soluções que faltavam, além de muita competência e vontade de fazer o trabalho dar certo.

O trabalho de diversos voluntários do Programa de Voluntariado do ICMBio Noronha foi essencial para a coleta de dados usados nos dois primeiros capítulos da tese, e agradeço a cada um deles pela dedicação e comprometimento com a qualidade do trabalho. Agradeço também a todos que contribuíram para as análises de laboratório e à Mariana e Aline pela ajuda com a análise dos fotoquadrados do Capítulo 3.

Agradeço ao Guilherme, Ana, Jéssica e Kelly (a.k.a. equipe maravilhosa) por fazerem a expedição "volta à ilha" ser um sucesso. Sempre que eu listo os participantes desta expedição acho que faltou alguém, por que é surpreendente a quantidade de coisa que fizemos em tão pouco tempo. Obrigada por toda a faina pré e pós expedição, igualmente importantes para o trabalho. Obrigada também à Cybelle por garantir que tudo estivesse perfeito no campo.

Agradeço à operadora de mergulho Sea Paradise e toda equipe pelo apoio nos mergulhos e coleta de dados, e ao Fernando por sempre se esforçar para viabilizar o trabalho da melhor forma possível.

Agradeço ao Neto e Joseph da ATM Esportes de Aventura pelo apoio nas coletas terrestres.

Agradeço ao Conselho Nacional de Desenvolvimento Científico e Tecnológico CNPq (Auxílio 443329/2019-2, concedido a Guilherme Ortigara Longo) pelo apoio financeiro. Este estudo foi financiado parcialmente pela Coordenação de Aperfeiçoamento de Pessoal de Nível Superior–Brasil (CAPES)– Código 001. Um agradecimento especial ao Instituto Serrapilheira por ter financiado a ida da minha filha e da minha mãe para me acompanharem no piloto do trabalho de campo em Natal, através da linha de financiamento para promoção da diversidade na ciência.

Agradeço ao Ronaldo pelo apoio e estímulo desde quando o Doutorado ainda era uma ideia distante, e pela ajuda em todas as etapas do trabalho. Agradeço sobretudo pelo companheirismo e por enfrentar as batalhas que se apresentaram nos últimos anos ao meu lado.

Agradeço à Nina por me permitir contemplar e viver o que há de mais belo e importante na vida.

Sumário

RESUMO

Os recifes e demais ambientes marinhos enfrentam ameaças significativas devido a impactos humanos, incluindo poluição, turismo desordenado e mudanças climáticas globais. Dentre as estratégias de conservação desses ambientes, destacam-se as Unidades de Conservação (UCs). No Brasil, o arquipélago de Fernando de Noronha (FN) é protegido por duas UCs, uma delas de proteção integral (Parque Nacional – PN) e uma de uso sustentável (Área de Proteção Ambiental – APA). Entretanto, o local tem sofrido com impactos crescentes devido à expansão do turismo e ocupação da Ilha, e às mudanças climáticas. O objetivo geral desta tese foi avaliar múltiplos impactos locais e globais sobre ambientes marinhos de FN, e o papel das UCs do arquipélago na mitigação destes impactos. No Capítulo 1 investigamos a presença de lixo marinho nas praias de FN e sua distribuição relacionada a fatores oceanográficos e turismo. Encontramos maior abundância de plásticos em áreas desabitadas a barlavento, apesar de estarem dentro do PN com normas mais restritivas, enquanto na costa de sotavento, próxima a áreas urbanas e dentro de uma UC menos restritiva, havia mais plásticos descartáveis e bitucas de cigarro. No Capítulo 2 investigamos a dinâmica temporal do bentos de uma poça de maré e observamos que a anomalia de temperatura foi o principal fator que afetou a saúde dos corais e sua comunidade de endossimbiontes. Embora o soterramento sazonal tenha causado alta variabilidade temporal na cobertura bentônica, não afetou a saúde dos corais. A quantidade de turistas mergulhando no local também não influenciou a saúde dos corais, indicando que os impactos do turismo podem ser reduzidos quando adequadamente manejados. No Capítulo 3, mapeamos e quantificamos a poluição marinha em FN (nutrientes, metais, microplásticos, esteróis e hidrocarbonetos policíclicos aromáticos - HPAs), caracterizando suas fontes e avaliando seus impactos nas comunidades recifais. Detectamos poluentes tanto na APA quanto no PN. Embora as concentrações de nutrientes na água do mar estivessem em conformidade com a legislação na maior parte das amostras, os efluentes das estações de tratamento de esgoto não atendiam os padrões legais. A assinatura isotópica de macroalgas indicou a presença de nitrogênio derivado de esgoto em toda a APA, sendo que áreas com enriquecimento de nutrientes apresentaram comunidades bentônicas distintas e com maior biomassa algal. A análise dos sedimentos revelou baixos níveis de HPAs e esteróis fecais, com exceção das amostras do Porto. Observamos concentrações de metais relativamente altas no

sedimento em algumas estações amostrais, o que pode estar relacionado tanto a fontes antrópicas quanto à origem vulcânica do arquipélago. Microplásticos foram encontrados em todas as amostras de água e sedimentos, tanto na APA quanto no PN. Destacamos a necessidade urgente de melhorar a gestão do esgoto para proteger o ecossistema marinho em FN, promovendo sua resiliência em face das mudanças climáticas. Ressaltamos a vulnerabilidade do PN, na costa sotavento a poluentes gerados na APA e na costa barlavento a poluentes transportados pelas correntes a partir de fontes distantes. As informações geradas por este trabalho servirão como referência para iniciativas de monitoramento e para melhorar estratégias de gestão para a conservação de FN e de outras UCs.

ABSTRACT

Reefs and other marine ecosystems face significant threats from human impacts, including pollution, poorly regulated tourism and global climate change. Marine Protected Areas (MPAs) stand out among the most important strategies for marine conservation. In Brazil, the Fernando de Noronha archipelago (FN) is protected by two MPAs: a no-take area and a multiple-use area. However, the archipelago has been experiencing increasing impacts due to the expansion of tourism and human occupation, and climate change. In this context, the main objective of this thesis was to assess multiple local and global impacts on marine environments in FN, and the role of the MPAs in mitigating these impacts. In Chapter 1, we investigated the presence of marine litter on the beaches of FN and its distribution related to oceanographic factors and tourism. We found a higher abundance of plastics in uninhabited areas on the windward side, despite being within the no-take area, a more restricted MPA, while on the leeward coast, near urban areas and within a less restricted MPA, there were more disposable plastics and cigarette butts. In Chapter 2, we investigated the temporal dynamics of benthic organisms in a tide pool and observed that temperature anomalies were the main factor affecting coral health and their endosymbiont communities. Although seasonal burial caused high temporal variability in benthic cover, it did not affect coral health. The number of tourists diving in the area also did not influence coral health, indicating that proper management can reduce tourism impacts. In Chapter 3, we mapped and quantified marine pollution in FN (nutrients, metals, microplastics, sterols, and polycyclic aromatic hydrocarbons - PAHs), characterizing their sources and evaluating their impacts on reef communities. We detected pollutants both in the multiple-use and in the no-take area. While nutrient concentrations in the water were in accordance with legislation in most samples, effluents from wastewater treatment plants did not meet legal standards. The isotopic signature of macroalgae indicated the presence of sewage-derived nitrogen throughout the multiple-use area, and nutrient-enriched areas showed distinct benthic communities with higher algal biomass. Sediment analysis revealed generally low levels of PAHs and fecal sterols, except in samples from Porto. We observed relatively high metal concentrations in sediments at some sampling stations, which may be related to anthropogenic sources and the archipelago's volcanic origin. Microplastics were found in all sediment and water samples, from the multiple-use and the no-take area. We emphasize the urgent need to improve wastewater management to protect the marine ecosystem in FN, promoting its

resilience facing climate change. We also highlight the vulnerability of the no-take area, with pollutants originating from the multiple-use area on the leeward coast and pollutants transported by currents from distant sources on the windward coast. The information generated by this work will serve as an important reference for monitoring initiatives and improving management strategies for conserving FN and other MPAs.

APRESENTAÇÃO

Esta tese é composta por três capítulos, escritos em inglês, na forma de artigos científicos. Além deles, a tese inclui uma Introdução e Conclusão em português, com o propósito de comunicar os objetivos, conclusões e recomendações do estudo para um público mais amplo, incluindo não só cientistas, mas também gestores, técnicos e comunidade interessada.

O primeiro capítulo traz o manuscrito "*Marine debris in the Fernando de Noronha Archipelago, a remote oceanic marine protected area in tropical SW Atlantic*". Neste trabalho avaliamos a composição e abundância do macro lixo marinho e os fatores que influenciam a sua distribuição espacial em Fernando de Noronha - FN. Utilizamos dados de monitoramento mensal do lixo marinho em seis praias da ilha de FN, coletados de 2016 a 2018. Verificamos que a distribuição e características do lixo marinho estão relacionadas a fatores oceanográficos, com o transporte de lixo gerado em fontes distantes do arquipélago, e também aos padrões de uso das praias por moradores e turistas, que geram lixo localmente. Discutimos os resultados relacionando-os às peculiaridades da gestão de cada uma das Unidades de Conservação - UCs de FN e fizemos recomendações de manejo para mitigação do problema. Este manuscrito foi publicado no periódico *Marine Pollution Bulletin* em janeiro de 2021.

O segundo capítulo é composto pelo manuscrito "*Drivers of temporal variation in benthic cover and coral health of an intertidal reef in Fernando de Noronha Archipelago, Southwestern Atlantic*". Neste estudo monitoramos mensalmente a dinâmica de organismos bentônicos por dois anos (2017-2019) e avaliamos a resposta de corais e seus endossimbiontes ao soterramento por areia, anomalias de temperatura e turismo na piscina do Atalaia, um dos principais atrativos turísticos do Parque Nacional -PN, onde a visitação é sujeita a uma série de regras com o objetivo de garantir a conservação do ambiente. Observamos alta variabilidade temporal na cobertura bentônica devida à dinâmica natural da areia trazida sazonalmente por ondas vindas da direção sudeste a sul-sudeste. Apesar de a cobertura de coral diminuir quando a areia se acumula, com algumas colônias permanecendo soterradas por até 60 dias, a saúde dos corais não foi afetada pelo soterramento, mas sim pela anomalia de temperatura (*i.e.*, temperaturas acima da média histórica). A anomalia de temperatura foi intensa durante o segundo ano de monitoramento, e nestes períodos houve aumento de branqueamento e alterações nos simbiontes associados. Não detectamos efeito significativo do número de visitantes acessando o local na saúde dos corais. Isso indica que, considerando as condições ambientais atuais, a gestão do turismo está sendo eficaz para evitar danos aos corais. Este é um estudo de caso interessante para ser utilizado em outras UCs marinhas, mostrando explicitamente como o monitoramento biológico pode auxiliar na avaliação e potenciais ajustes nas estratégias de gestão. Este manuscrito foi publicado no periódico *Regional Studies in Marine Science* em fevereiro de 2023.

O terceiro capítulo é composto pelo manuscrito *"Marine Pollution in Fernando de Noronha Archipelago, an oceanic marine protected area in tropical SW Atlantic"* que será submetido ao periódico *Science of the Total Environment*. Neste estudo mapeamos e quantificamos a poluição marinha em Fernando de Noronha, caracterizando suas fontes e impactos sobre as comunidades recifais. Para isso, diversos poluentes (nutrientes, metais, microplásticos, esteróis e contaminantes orgânicos) foram quantificados em amostras de água e sedimento obtidas ao redor do arquipélago. A origem dos nutrientes incorporados no recife foi avaliada através de análise isotópica de amostras de macroalgas. As comunidades bentônicas e de peixes foram caracterizadas nos pontos de coleta, e a relação entre a estrutura da comunidade e os dados de qualidade ambiental foi avaliada. Detectamos poluentes na APA e no PN, incluindo nutrientes, metais e microplásticos. Observamos que o tratamento de efluentes tem sido ineficiente, afetando as comunidades bentônicas (*i.e.,* organismos que vivem associados ao fundo, como algas). Discutimos a vulnerabilidade das UCs à poluição e indicamos urgências e prioridades de gestão para proteger o ecossistema marinho em FN e aumentar sua resiliência às mudanças climáticas.

INTRODUÇÃO GERAL

Os ecossistemas recifais estão entre os ambientes mais ameaçados do mundo, com perdas aceleradas sendo causadas por diversas atividades antrópicas (Hughes *et al.* 2017, Barlow *et al.* 2018). Os impactos antrópicos sobre os ambientes recifais ocorrem em diferentes escalas (Adam *et al.* 2015). Em escala local, a sobrepesca e a redução da qualidade da água, em conjunto com os impactos associados ao turismo desordenado são algumas das ameaças mais significativas aos recifes (Knowlton & Jackson 2008; Andrello *et al.* 2022). Em escala mais ampla, os impactos das mudanças climáticas globais sobre os recifes de coral, acelerados pela ação antrópica, são cada vez mais evidentes (Attrill 2009, Doney *et al.* 2012).

Na última década o mundo vem registrando recordes sucessivos de temperaturas elevadas, consequência da queima de combustíveis fósseis e a emissão de gases do efeito estufa (Miller *et al.* 2017). No oceano, as temperaturas estão recorrentemente acima das médias históricas (*i.e.*, anomalias térmicas positivas) e a absorção do gás carbônico (CO_2) da atmosfera pelos oceanos vem deixando o oceano mais ácido (Gaylord *et al.* 2015, Hoegh-Guldberg *et al.* 2007). Devido ao aquecimento da água, a frequência e extensão de eventos de branqueamento em massa e proliferação de doenças em corais têm aumentado consideravelmente (Hughes *et al.* 2018, Lough *et al.* 2018), acarretando em impactos ecológicos e sociais (Jones *et al.* 2008, Weatherdon *et al.* 2016).

Diversos impactos em escala local estão relacionados ao turismo costeiro, agravados quando realizado de maneira intensiva (Andrello *et al.* 2022). Danos físicos diretos nos organismos bentônicos podem resultar do mergulho, ancoragem e pisoteio (Giglio *et al.* 2017, Forrester, 2020). O ruído das embarcações (Simpson *et al.* 2016, Ferrier-Pagès *et al.* 2021) e a presença de mergulhadores também podem alterar o comportamento de algumas espécies de peixes (Benevides *et al.* 2019, Giglio *et al*. 2019). O desenvolvimento urbano desordenado associado ao turismo costeiro pode alterar ou destruir os recifes e habitats adjacentes e frequentemente está relacionado à redução da qualidade ambiental devido à poluição (Araújo *et al.* 2021).

A poluição marinha ocorre devido ao descarte inadequado de lixo, falta de tratamento de efluentes, uso de fertilizantes e mudanças no uso do solo relacionadas à agricultura, introduzindo uma variedade de substâncias nocivas nos ecossistemas naturais (Tilley *et al.* 2014, Wear & Thurber 2015). Os impactos do esgoto, que são evidentes desde a década de 1980 (Walker & Ormond 1982, Pastorok & Bilyard 1985), incluem desestabilização do microbioma, aumento de doenças, diminuição da cobertura de corais, proliferação de patógenos, macroalgas e organismos filtradores que competem com os corais por espaço, além do declínio na abundância de peixes e das mudanças nas interações entre organismos (Hunter & Evans 1995, Moreira *et al.* 2014, Zeneveld *et al.* 2016, Duprey *et al.* 2017). A poluição pode afetar negativamente a abundância e riqueza de espécies de corais, bem como seu crescimento e capacidade reprodutiva (Pastorok & Bilyard 1985). Em ambientes naturalmente oligotróficos, como as ilhas oceânicas, qualquer adição de macronutrientes limitantes nas águas costeiras pode causar mudanças na comunidade recifal (Wear & Thurber 2015). No entanto, apesar destes impactos, há poucos investimentos em estudos e soluções para esse problema (Wear & Thurber, 2015, Abessa 2018).

Os resíduos sólidos, conhecidos como lixo marinho, em especial o plástico, também causam impactos importantes, como adesão de microplásticos aos pólipos de coral (Rani-Borges *et al.* 2023), inibição do crescimento de algas mutualísticas e danos ao sistema imunológico dos corais (Gregory 2009, Nama *et al.* 2023). Além disso, animais como mamíferos marinhos, peixes, aves e tartarugas marinhas podem ficar presos em detritos, afetando sua alimentação, locomoção, reprodução e sobrevivência (Bergmann *et al.* 2015). A ingestão de lixo marinho afeta os mais diversos tipos de organismos no planeta (Santos *et al.* 2021), e pode facilitar a propagação de espécies invasoras e patógenos (Lamb *et al.* 2018, Mantelatto *et al.* 2020), além de prejudicar o turismo e causar perdas econômicas devido à diminuição do apelo estético das praias e recifes (Rodríguez *et al.* 2020, Hayati *et al.* 2020).

Os recifes de coral são afetados tanto por impactos antropogênicos locais, como a poluição, o turismo e a pesca excessiva, quanto por mudanças climáticas globais (Donovan *et al.* 2020, França *et al.* 2020). Tais impactos podem resultar em uma diminuição da efetividade das medidas de proteção, como as áreas marinhas protegidas (Abessa 2018). Embora as áreas protegidas - áreas especialmente dedicadas a proteção em manutenção da diversidade biológica, e dos recuros naturais e recursos culturais associados (Dudley 2008) - sejam amplamente adotadas para a conservação marinha (Roberts & Hawkins 2000, Ferreira *et al.* 2022), a mitigação das mudanças climáticas está além de sua capacidade (Edgar *et al.* 2014; Bruno *et al.* 2018). No entanto, boas práticas de manejo local, como a manutenção da qualidade da água, podem aumentar a resistência e resiliência dos recifes de coral aos impactos climáticos (Wooldridge & Done

2009, Bruno *et al.* 2018). Portanto, as áreas marinhas protegidas – AMPs que implementam essas práticas podem ajudar a mitigar os impactos em escalas espaciais mais amplas.

Um dos mais importantes conjuntos de AMPs do Brasil está localizado no arquipélago de Fernando de Noronha, e é composto por uma área de proteção integral (Parque Nacional - PN) e outra de uso sustentável (Área de Proteção Ambiental - APA) (Fig. 1). Essas AMPs foram criadas com o objetivo de proteger a fauna, flora e recursos naturais do arquipélago, conciliando o turismo com a preservação ambiental (Brasil 1986, Brasil 1988). No entanto, a efetividade dessas regulamentações pode ser influenciada pela categoria de proteção, pela gestão, pela conexão com os ambientes marinhos ao redor e por fatores externos, como as mudanças climáticas (Ferreira *et al.* 2022, Wilson *et al.* 2020).

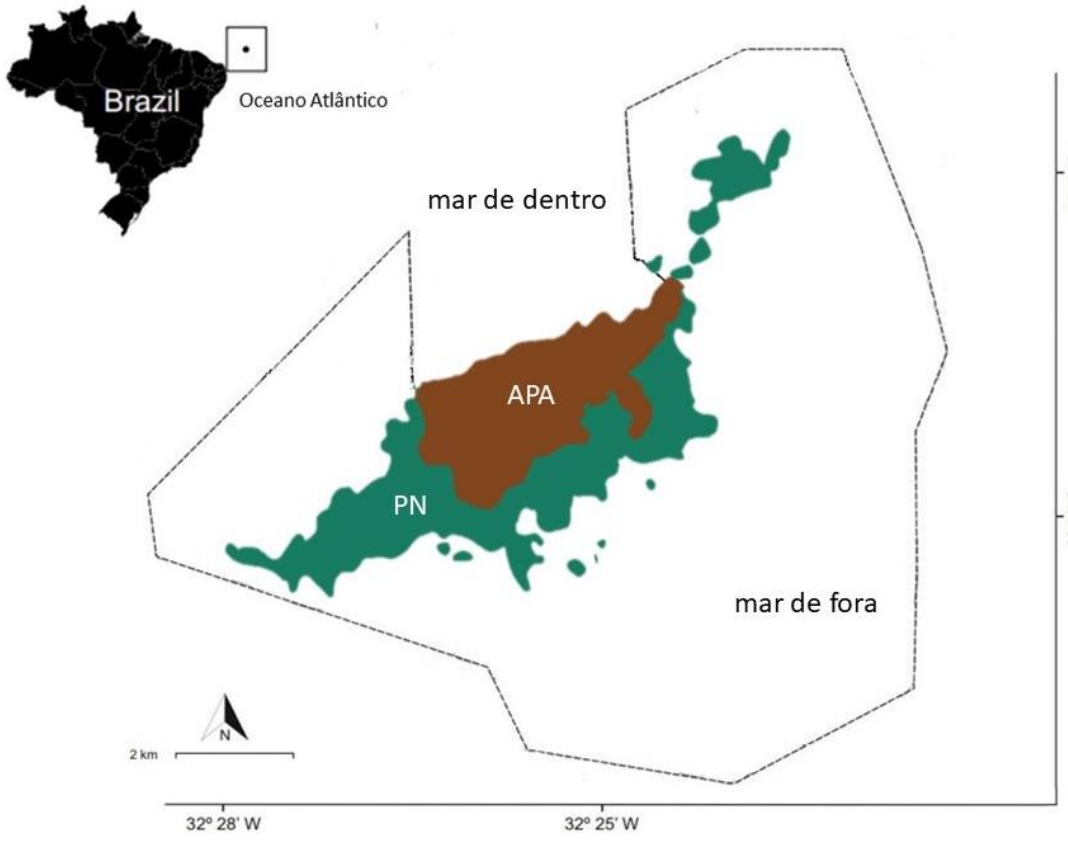


Figura 1: Arquipélago de Fernando de Noronha: localização e áreas marinhas protegidas. APA: Área de Proteção Ambiental de Fernando de Noronha. PN: Parque Nacional Marinho de Fernando de Noronha. Mar de dentro: face sotavento, voltada para o norte. Mar de fora: face barlavento, voltada para o sul. As linhas pontilhadas indicam os limites marinhos do Parque Nacional.

Fernando de Noronha é uma área de grande importância para a conservação da biodiversidade, sendo reconhecida como Sítio do Patrimônio Marinho pela UNESCO. Trata-se de uma das poucas áreas insulares do Atlântico Sul, que além de abrigar espécies endêmicas e ameaçadas de extinção, é fonte de propágulos para as áreas costeiras (Peluso *et al.* 2018). O turismo é a principal atividade econômica de FN, tornando a conservação marinha essencial para sua sustentabilidade a longo prazo. No entanto, as ilhas oceânicas brasileiras, incluindo FN, estão sujeitas a pressões antrópicas crescentes, como turismo, poluição, lixo marinho e mudanças climáticas (Halpern *et al.* 2015, Soares 2018). Apesar das medidas de manejo eficazes na redução da pressão da pesca e de estarem sujeitas a impactos menores quando comparados a áreas mais próximas aos centros urbanos (Magris *et al.* 2018), essas ilhas são ambientes sensíveis e com menor capacidade de recuperação devido ao seu isolamento geográfico e às correntes oceanográficas que dificultam a chegada de propágulos do continente mais próximo.

As pressões em escala local e global sobre Fernando de Noronha tendem a se agravar. Considerando a infraestrutura disponível, foi elaborado um estudo de capacidade de carga de FN estabelecendo o número máximo de pessoas que a ilha poderia comportar sem gerar impactos ambientais significativos e sem prejudicar a qualidade dos serviços. A capacidade de carga estimada na época era de 4 mil pessoas, e a ilha já registrava uma sobrecarga de 1400 pessoas (ICMBio 2009). Porém, o número de turistas mais que dobrou desde então, passando de 61 mil em 2009 para 149 mil em 2022, resultando em sobrecarga e impactos ambientais significativos. O aumento do turismo também causa preocupações em relação ao impacto sobre a biota, geração de lixo, sobrecarga dos sistemas de abastecimento e transporte, poluição veicular e aumento do volume de esgoto, sendo que metade do esgoto já não era tratado adequadamente em 2009 (ICMBio 2009, Monteiro *et al.* 2018). Os vazamentos frequentes afetam a qualidade das águas e resultam na interdição de praias (https://g1.globo.com/platb/pe-viver-noronha/2014/03/31/esgoto-estoura-e-praia-do-cachorro-e-interditada/).

Buscando reduzir a poluição por plástico em FN, a entrada de alguns tipos de produtos descartáveis plásticos de uso único foi proibida no local (Pernambuco 2018). No entanto, o problema do lixo marinho é mais amplo e medidas locais podem não ser suficientes, considerando a chegada de detritos, especialmente plástico, através das correntes marinhas (Monteiro *et al.* 2018, Onink *et al.* 2021). A poluição por plástico é global, irreversível e está em constante aumento (Ford *et al.* 2022, Williams & Rangel-Buitrago 2022).

Assim como a poluição por plástico, uma ameaça global significativa que já está afetando todas as regiões do mundo e tende a se intensificar são as ondas de calor e anomalias térmicas mais frequentes (Hughes *et al.* 2018). Os recifes do Atlântico Sul têm características que os tornam menos suscetíveis a eventos de branqueamento em massa seguidos de alta mortalidade (Mies *et al.* 2020). Entretanto, considerando os cenários futuros de aquecimento dos oceanos, impactos negativos poderão ser observados em FN (Diele-Viegas & Rocha 2021), com diminuição na cobertura de corais prevista na área do Parque Nacional (Albuquerque *et al.* 2023).

Compreender como esses estressores em diferentes escalas afetam a biodiversidade e os serviços ecossistêmicos é crucial para orientar medidas de manejo. É importante realizar levantamentos para obter valores de referência antes que os impactos se intensifiquem ainda mais, especialmente diante do aumento da população, turismo e mudanças climáticas (Underwood 1994, Stewart-Oaten & Bence 2001). O conjunto de UCs de FN constitui um cenário ideal para avaliações da influência relativa de impactos locais e globais sobre ecossistemas recifais, bem como suas interações.

Neste contexto, o objetivo geral desta tese é avaliar múltiplos impactos locais e globais sobre ambientes marinhos de Fernando de Noronha, e o papel das UCs do arquipélago na mitigação destes impactos. Especificamente, os capítulos tem como objetivo:

1) Avaliar a composição e abundância do macro lixo marinho e os fatores que influenciam a sua distribuição espacial nas praias em FN.
2) Investigar a dinâmica temporal do bentos de uma poça de maré e sua relação com variáveis ambientais e número de visitantes.
3) Descrever os padrões espaciais de poluição marinha em FN e seus efeitos sobre as comunidades recifais.

Referências

Abessa, D.M.S., Albuquerque, H.C., Morais, L.G., Araújo, G.S., Fonseca, T.G., Cruz, A.C.F., Campos, B.G., Camargo, J.B.D.A., Gusso-Choueri, P.K., Perina, F.C., Choueri, R.B., Buruaem, L.M., 2018. Pollution status of marine protected areas worldwide and the consequent toxic effects are unknown. Environ. Pollut. https://doi.org/10.1016/j.envpol.2018.09.129

Adam, T.C., Burkepile, D.E., Ruttenberg, B.I., Paddack, M.J., 2015. Herbivory and the resilience of Caribbean coral reefs: Knowledge gaps and implications for management. Mar. Ecol. Prog. Ser. 520, 1–20. https://doi.org/10.3354/meps11170

Albuquerque, F.C., Bleuel, J., Pinto, M.P., Longo, G.O., 2023. In the right place at the right time: representativeness of corals within marine protected areas under warming scenarios in Brazil. Ocean Coast. Manag. 233, 106469. https://doi.org/10.1016/j.ocecoaman.2022.106469

Andrello, M., Darling, E.S., Gelfand, S., Wenger, A., Ahmadia, G.N., Suárez-castro, A.F., 2022. A global map of human pressures on tropical coral reefs. Conserv. Lett. 15, 1–12. https://doi.org/10.1111/conl.12858

Araújo, M.P., Hamacher, C., Farias, C. de O., Soares, M.L.G., 2021. Fecal sterols as sewage contamination indicators in Brazilian mangroves. Mar. Pollut. Bull. 165, 112149. https://doi.org/10.1016/j.marpolbul.2021.112149

Attrill, M.J., 2009. Changes in Coral Reef Ecosystems as an Indicator of Climate and Global Change, 1st ed, Climate Change: Observed Impacts on Planet Earth. Elsevier B.V. https://doi.org/10.1016/B978-0-444-53301-2.00013-0

Barlow, J., França, F., Gardner, T., Hicks, C., Lennox, G., Berenguer, E., Castello, L., Economo, E.P., Ferreira, J., Guénard, B., Gontijo Leal, C., Isaac, V., Lees, A., Parr, C., Wilson, S.K., Young, P., Graham, N., 2018. The future of hyperdiverse tropical ecosystems. Nature 559, 517.

Benevides, L.J., Cardozo-Ferreira, G.C., Ferreira, C.E.L., Pereira, P.H.C., Pinto, T.K., Sampaio, C.L.S., 2019. Fear-induced behavioural modifications in damselfishes can be diver-triggered. J. Exp. Mar. Bio. Ecol. 514–515, 34–40. https://doi.org/10.1016/j.jembe.2019.03.009

Bergmann, M., Gutow, L., Klages, M., 2015. Marine Anthropogenic Litter, Marine Anthropogenic Litter. Springer International Publishing, Cham. https://doi.org/10.1007/978-3-319-16510-3

Brasil, 1986. Decreto nº 92.755, de 5 de junho de 1986.

Brasil, 1988. Decreto nº 96.693, de 14 de setembro de 1988.

Bruno, J.F., Bates, A.E., Cacciapaglia, C., Pike, E.P., Amstrup, S.C., Van Hooidonk, R., Henson, S.A., Aronson, R.B., 2018. Climate change threatens the world's marine protected areas. Nat. Clim. Chang. 8, 499–503. https://doi.org/10.1038/s41558-018-0149-2

Diele-Viegas, L.M., Rocha, C.F.D., 2021. Will climate change be harmful for small tropical islands? The case of Fernando de Noronha Archipelago, Brazil. Brazilian J. Climatol. 29, 341–356. https://doi.org/10.5380/abclima

Doney, S.C., Ruckelshaus, M., Duffy, J.E., Barry, J.P., Chan, F., English, C.A., Galindo, H.M., Grebmeier, J.M., Hollowed, A.B., Knowlton, N., Polovina, J., Rabalais, N.N., Sydeman, W.J., Talley, L.D., 2012. Climate Change Impacts on Marine

Ecosystems. Ann. Rev. Mar. Sci. 4, 11–37. https://doi.org/10.1146/annurev-marine-041911-111611

Donovan, M.K., Adam, T.C., Shantz, A.A., Speare, K.E., Munsterman, K.S., Rice, M.M., Schmitt, R.J., Holbrook, S.J., Burkepile, D.E., 2020. Nitrogen pollution interacts with heat stress to increase coral bleaching across the seascape. Proc. Natl. Acad. Sci. U. S. A. 117, 5351–5357. https://doi.org/10.1073/pnas.1915395117

Dudley, N. (Editor) (2008). Guidelines for Applying Protected Area Management Categories. Gland, Switzerland: IUCN. x + 86pp. WITH Stolton, S., P. Shadie and N. Dudley (2013). IUCN WCPA Best Practice Guidance on Recognising Protected Areas and Assigning Management Categories and Governance Types, Best Practice Protected Area Guidelines Series No. 21, Gland, Switzerland: IUCN. xxpp.

Duprey, N.N., Wang, X.T., Thompson, P.D., Pleadwell, J.E., Raymundo, L.J., Kim, K., Sigman, D.M., Baker, D.M., 2017. Life and death of a sewage treatment plant recorded in a coral skeleton Δ15N record. Mar. Pollut. Bull. 120, 109–116. https://doi.org/10.1016/j.marpolbul.2017.04.023

Edgar, G.J., Stuart-Smith, R.D., Willis, T.J., Kininmonth, S., Baker, S.C., Banks, S., Barrett, N.S., Becerro, M.A., Bernard, A.T.F., Berkhout, J., Buxton, C.D., Campbell, S.J., Cooper, A.T., Davey, M., Edgar, S.C., Försterra, G., Galván, D.E., Irigoyen, A.J., Kushner, D.J., Moura, R.L., Parnell, P.E., Shears, N.T., Soler, G., Strain, E.M.A., Thomson, R.J., 2014. Global conservation outcomes depend on marine protected areas with five key features. Nature 506, 216–220. https://doi.org/10.1038/nature13022

Ferreira, H.M., Magris, R.A., Floeter, S.R., Ferreira, C.E.L., 2022. Drivers of ecological effectiveness of marine protected areas : A meta-analytic approach from the Southwestern Atlantic Ocean (Brazil). J. Environ. Manage. 301, 113889. https://doi.org/10.1016/j.jenvman.2021.113889

Ferrier-Pagès, C., Leal, M.C., Calado, R., Schmid, D.W., Bertucci, F., Lecchini, D., Allemand, D., 2021. Noise pollution on coral reefs? — A yet underestimated threat to coral reef communities. Mar. Pollut. Bull. 165, 112129. https://doi.org/10.1016/j.marpolbul.2021.112129

Ford, H. V, Jones, N.H., Davies, A.J., Godley, B.J., Jambeck, J.R., Napper, I.E., Suckling, C.C., Williams, G.J., Woodall, L.C., Koldewey, H.J., 2022. The fundamental links between climate change and marine plastic pollution. Sci. Total Environ. 806, 150392. https://doi.org/10.1016/j.scitotenv.2021.150392

Forrester, G.E., 2020. The influence of boat moorings on anchoring and potential anchor damage to coral reefs. Ocean Coast. Manag. 198, 105354. https://doi.org/10.1016/j.ocecoaman.2020.105354

França, F.M., Benkwitt, C.E., Peralta, G., Robinson, J.P.W., Graham, N.A.J., Tylianakis, J.M., Berenguer, E., Lees, A.C., Ferreira, J., Louzada, J., Barlow, J., 2020. Climatic and local stressor interactions threaten tropical forests and coral reefs. Philos. Trans. R. Soc. B Biol. Sci. 375. https://doi.org/10.1098/rstb.2019.0116

Gaylord, B., Kroeker, K.J., Sunday, J.M., Anderson, K.M., Barry, J.P., Brown, N.E., Connell, S.D., Dupont, S., Fabricius, K.E., Hall-Spencer, J.M., Klinger, M.M., Munday, P.L., Russell, B.D., Sanford, E., Schreiber, S.J., Thiyagarajan, V., Vauhgan, M.L.H., Widdicombe, S., Harley, C.D.G., 2015. Ocean acidification through the lens of ecological theory. Ecology 96, 3–15. https://doi.org/10.1525/auk.2008.1408

Giglio, V.J., Ternes, M.L.F., Mendes, T.C., Cordeiro, C.A.M.M., Ferreira, C.E.L., 2017. Anchoring damages to benthic organisms in a subtropical scuba dive hotspot. J. Coast. Conserv. 21, 311–316. https://doi.org/10.1007/s11852-017-0507-7

Giglio, V.J., Ternes, M.L.F., Kassuga, A.D., Carlos, E.L., 2019. Scuba diving and sedentary fish watching: effects of photographer approach on seahorse behavior. J. Ecotour. 18, 142–151. https://doi.org/10.1080/14724049.2018.1490302

Gregory, M.R., 2009. Environmental implications of plastic debris in marine settings—entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. Philos. Trans. R. Soc. B Biol. Sci. 364, 2013–2025. https://doi.org/10.1098/rstb.2008.0265

Hayati, Y., Adrianto, L., Krisanti, M., Pranowo, W.S., Kurniawan, F., 2020. Magnitudes and tourist perception of marine debris on small tourism island : Assessment of Tidung Island , Jakarta , Indonesia. Mar. Pollut. Bull. 158, 111393. https://doi.org/10.1016/j.marpolbul.2020.111393

Halpern, B.S., Frazier, M., Potapenko, J., Casey, K.S., Koenig, K., Longo, C., Lowndes, J.S., Rockwood, R.C., Selig, E.R., Selkoe, K.A., Walbridge, S., 2015. Spatial and temporal changes in cumulative human impacts on the world's ocean. Nat. Commun. 6, 1–7. https://doi.org/10.1038/ncomms8615

Hoegh-Guldberg, O., Mumby, P.J., Hooten, A.J., Steneck, R.S., Greenfield, P., Gomez, E., Harvell, C.D., Sale, P.F., Edwards, A.J., Caldeira, K., Knowlton, N., Eakin, C.M., Iglesias-Prieto, R., Muthiga, N., Bradbury, R.H., Dubi, A., Hatziolos, M.E., 2007. Coral Reefs Under Rapid Climate Change and Ocean Acidification. Science 318, 1737–1742. https://doi.org/10.1126/science.1152509

Hughes, T.P., Anderson, K.D., Connolly, S.R., Heron, S.F., James, T., Lough, J.M., Baird, A.H., Baum, J.K., Berumen, M.L., Tom, C., Claar, D.C., Eakin, C.M., Gilmour, J.P., Graham, N.A.J., Harrison, H., Hobbs, J.A., Hoey, A.S., Hoogenboom, M., Lowe, R.J., Mcculloch, M.T., Pandolfi, J.M., Pratchett, M., Schoepf, V., Wilson, S.K., Cook, J., Watch, C.R., Oceanic, U.S.N., 2018. Spatial and temporal patterns of mass bleaching of corals in the Anthropocene. Science 359, 80–83. https://doi.org/10.1126/science.aan8048

Hunter, C.L., Evans, C.W., 1995. Coral reefs in Kaneohe Bay, Hawaii: Two centuries of western influence and two decades of data. Bull. Mar. Sci. 57, 501–515.

ICMBio, 2009. Estudo e Determinação da Capacidade de Suporte e seus Indicadores de Sustentabilidade com vistas à implantação do Plano de Manejo da Área de Proteção Ambiental do arquipélago de Fernando de Noronha. Disponível em:

https://www.parnanoronha.com.br/_files/ugd/2b2627_de1274fb1b404642a890f1c3e29756ed.pdf

Jones, L., Acolado, P., Cala, Y., Án, D.C., Coelho, V., Hernández, A., Jones, R.J., Mallela, J., Manfrino, C., 2008. Status of Caribbean Coral Reefs after Bleaching and Hurricanes in 2005. Faculty Books.

Lamb, J.B., Willis, B.L., Fiorenza, E.A., Couch, C.S., Howard, R., Rader, D.N., True, J.D., Kelly, L.A., Ahmad, A., Jompa, J., Harvell, C.D., 2018. Plastic waste associated with disease on coral reefs. Science 359, 460–462. https://doi.org/10.1126/science.aar3320

Lough, J.M., Anderson, K.D., Hughes, T.P., 2018. Increasing thermal stress for tropical coral reefs: 1871-2017. Sci. Rep. 8, 1–8. https://doi.org/10.1038/s41598-018-24530-9

Magris, R.A., Grech, A., Pressey, R.L., 2018. Cumulative human impacts on coral reefs: Assessing risk and management implications for brazilian coral reefs. Diversity 10. https://doi.org/10.3390/d10020026

Mantelatto, M.C., Póvoa, A.A., Skinner, L.F., Araujo, F.V., Creed, J.C., 2020. Marine litter and wood debris as habitat and vector for the range expansion of invasive corals (*Tubastraea* spp.). Mar. Pollut. Bull. 160, 111659. https://doi.org/10.1016/j.marpolbul.2020.111659

Mies, M., Francini-Filho, R.B., Zilberberg, C., Garrido, A.G., Longo, G.O., Laurentino, E., Güth, A.Z., Sumida, P.Y.G., Banha, T.N.S., 2020. South Atlantic Coral Reefs Are Major Global Warming Refugia and Less Susceptible to Bleaching. Front. Mar. Sci. 7, 1–13. https://doi.org/10.3389/fmars.2020.00514

Miller, D.D., Ota, Y., Sumaila, U.R., Cisneros-Montemayor, A.M., Cheung, W.W.L., 2017. Adaptation strategies to climate change in marine systems. Glob. Chang. Biol. 24, 1–14. https://doi.org/10.1111/gcb.13829

Monteiro, R.C.P., Ivar do Sul, J.A., Costa, M.F., 2018. Plastic pollution in islands of the Atlantic Ocean. Environ. Pollut. 238, 103–110. https://doi.org/10.1016/j.envpol.2018.01.096

Moreira, A.P.B., Chimetto Tonon, L.A., Do Valle P. Pereira, C., Alves, N., Amado-Filho, G.M., Francini-Filho, R.B., Paranhos, R., Thompson, F.L., 2014. Culturable heterotrophic bacteria associated with healthy and bleached scleractinian madracis decactis and the fireworm hermodice carunculata from the remote St. Peter and St. Paul Archipelago, Brazil. Curr. Microbiol. 68, 38–46. https://doi.org/10.1007/s00284-013-0435-1

Nama, S., Shanmughan, A., Nayak, B.B., Bhushan, S., 2023. Impacts of marine debris on coral reef ecosystem : A review for conservation and ecological monitoring of the coral reef ecosystem. Mar. Pollut. Bull. 189, 114755. https://doi.org/10.1016/j.marpolbul.2023.114755

Onink, V., Jongedijk, C.E., Hoffman, M.J., van Sebille, E., Laufkötter, C., 2021. Global simulations of marine plastic transport show plastic trapping in coastal zones. Environ. Res. Lett. 16, 064053. https://doi.org/10.1088/1748-9326/abecbd

Peluso, L., Tascheri, V., Nunes, F.L.D., Castro, C.B., Pires, D.O., Zilberberg, C., 2018. Contemporary and historical oceanographic processes explain genetic connectivity in a Southwestern Atlantic coral. Sci. Rep. 8, 1–12. https://doi.org/10.1038/s41598-018-21010-y

Pernambuco 2018. Decreto Distrital nº 02, de 12 de dezembro de 2018.

Rani-Borges, B., Gomes, E., Maricato, G., de Carvalho Lins, L.H.F., de Moraes, B.R., Lima, G.V., Côrtes, L.G.F., Tavares, M., Pereira, P.H.C., Ando, R.A., Queiroz, L.G., 2023. Unveiling the hidden threat of microplastics to coral reefs in remote South Atlantic islands. Sci. Total Environ. 165401. https://doi.org/10.1016/j.scitotenv.2023.165401

Roberts, C.M., Hawkins, J.P., 2000. Fully-protected marine reserves : a guide, WWF Endangered Seas Campain. University of York, NW, Washington, DC.

Rodríguez, Y., Ressurreição, A., Pham, C.K., 2020. Socio-economic impacts of marine litter for remote oceanic islands : The case of the Azores. Mar. Pollut. Bull. 160, 111631. https://doi.org/10.1016/j.marpolbul.2020.111631

Santos, R.G., Machovsky-Capuska, G.E., Andrades, R., 2021. Plastic ingestion as an evolutionary trap: Toward a holistic understanding. Mar. Mammal Sci. 373, 56–60. https://doi.org/10.1126/science.abh0945

Simpson, S.D., Radford, A.N., Nedelec, S.L., Ferrari, M.C.O., Chivers, D.P., McCormick, M.I., Meekan, M.G., 2016. Anthropogenic noise increases fish mortality by predation. Nat. Commun. 7, 10544. https://doi.org/10.1038/ncomms10544

Soares, M. O., 2018. Climate change and regional human pressures as challenges for management in oceanic islands, South Atlantic. Mar. Pollut. Bull. 131, 347–355. https://doi.org/10.1016/j.marpolbul.2018.04.008

Stewart-Oaten, A., Bence, J.R., 2001. Temporal and spatial variation in environmental impact assessment. Ecol. Monogr. 71, 305–339.

Tilley, E., Ulrich, L., Lüthi, C., Reymond, P., Schertenleib, R., Zurbrügg, C., 2014. Compendium of sanitation systems and technologies. 2nd revised Edition, 2nd ed. Swiss Federal Institute of Aquatic Science and Technology (Eawag), Dübendorf, Switzerland. Wear, S.L., Thurber, R.V., 2015. Sewage pollution : mitigation is key for coral reef stewardship. Ann. N. Y. Acad. Sci. 1355, 15–30. https://doi.org/10.1111/nyas.12785

Underwood, A.J., 1994. On Beyond BACI : Sampling Designs that Might Reliably Detect Environmental Disturbances. Ecol. Appl. 4, 3–15.

Walker, D.I., Ormond, R.F.G., 1982. Coral death from sewage and phosphate pollution at Aqaba, Red Sea. Mar. Pollut. Bull. 13, 21–25. https://doi.org/10.1016/0025-326X(82)90492-1

Wear, S.L., Thurber, R.V., 2015. Sewage pollution : mitigation is key for coral reef stewardship. Ann. N. Y. Acad. Sci. 1355, 15–30. https://doi.org/10.1111/nyas.12785

Weatherdon, L. V., Magnan, A.K., Rogers, A.D., Sumaila, U.R., Cheung, W.W.L., 2016. Observed and Projected Impacts of Climate Change on Marine Fisheries, Aquaculture, Coastal Tourism, and Human Health: An Update. Front. Mar. Sci. 3. https://doi.org/10.3389/fmars.2016.00048

Williams, A.T., Rangel-Buitrago, N., 2022. The past, present, and future of plastic pollution. Mar. Pollut. Bull. 176, 113429. https://doi.org/10.1016/j.marpolbul.2022.113429

Wilson, K.L., Tittensor, D.P., Worm, B., Lotze, H.K., 2020. Incorporating climate change adaptation into marine protected area planning 3251–3267. https://doi.org/10.1111/gcb.15094

Wooldridge, S.A., Done, T.J., 2009. Improved water quality can ameliorate effects of climate change on corals. Ecol. Appl. 19, 1492–1499.

Zaneveld, J.R., Burkepile, D.E., Shantz, A.A., Pritchard, C.E., McMinds, R., Payet, J.P., Welsh, R., Correa, A.M.S., Lemoine, N.P., Rosales, S., Fuchs, C., Maynard, J.A., Thurber, R.V., 2016. Overfishing and nutrient pollution interact with temperature to disrupt coral reefs down to microbial scales. Nat. Commun. 7, 1–12. https://doi.org/10.1038/ncomms11833

CAPÍTULO 1

Marine debris in the Fernando de Noronha Archipelago, a remote oceanic marine protected area in tropical SW Atlantic

Capítulo 1

Marine debris in the Fernando de Noronha Archipelago, a remote oceanic marine protected area in tropical SW Atlantic*

Ana Carolina Grillo[a,✉] Thayná Jeremias Mello[a,b]

a Parque Nacional Marinho de Fernando de Noronha, Instituto Chico Mendes de Conservação da Biodiversidade, Alameda do Boldró, s/n, Boldró, 53990-00, Fernando de Noronha, PE, Brazil

b Programa de Pós-Graduação em Ecologia, Centro de Biociências, Universidade Federal do Rio Grande do Norte, 59072-970, Natal, RN, Brazil

✉ *Corresponding author*

*Manuscript published in the *Marine Pollution Bulletin* 164 (2021) 11 2021 https://doi.org/10.1016/j.marpolbul.2021.112021

Abstract

Marine debris is widespread worldwide, from coastal areas to remote protected oceanic islands. We assessed marine macro-debris on the shores of Fernando de Noronha, an archipelago 360 km off Brazil that encompasses no-take and multiple-use areas. The windward uninhabited coast, more exposed to oceanic currents and winds and inside a no-take area, presented higher abundance of plastic debris. The leeward coast, within the multiple-use urban area, presented more disposable plastics and cigarette butts. These patterns may be explained by the marine debris transportation by ocean currents to the windward side and by locally generated debris by the high quantity of beach users in the leeward coast. These results indicate that oceanographic characteristics and tourism infrastructure play important roles in the accumulation of marine debris in a protected archipelago. They also serve as a baseline for future monitoring initiatives and to improve strategies to tackle plastic pollution within this remote archipelago.

Keywords

Oceanic island; Marine litter; Plastic pollution; Solid waste; Environmental management; Brazil

Oceanic islands, which rise beyond continental shelves, are among remote areas frequently considered as near-pristine environments (Quimbayo *et al.*, 2019). Because of their evolution in isolation, they can harbor unique terrestrial and marine biotas, representing important conservation hotspots. However, some anthropogenic impacts, such as pollution and climate change, respect no boundaries (Halpern *et al.*, 2015), leading to negative impacts on remote and protected locations such as oceanic islands (Barnes, 2005; Lavers and Bond, 2017).

Marine debris found on insular shores originates either locally or are transported from elsewhere by ocean currents. Inhabited islands are sources of marine debris because of human occupation and tourism (Wilson and Verlis, 2017). However, some types of debris, particularly plastics, can be found several kilometers from their original source (Ivar do Sul and Costa, 2007; de Scisciolo *et al.*, 2016). They can be dropped in distant areas or dumped at sea, subsequently accumulating on inhabited and unpopulated remote island's coastlines (Pérez-Venegas *et al.*, 2017; Andrades *et al.*, 2018) by the action of currents and winds (Barnes *et al.*, 2009; Eriksson *et al.*, 2013). Due to this oceanic transport, islands tend to accumulate proportionally more plastic debris than continental sites (Pérez-Venegas *et al.*, 2017). Although marine debris found in islands can be both locally and distantly generated (Carson *et al.*, 2013; de Scisciolo *et al.*, 2016), the later source is more prevalent (Donohue *et al.*, 2001).

Marine debris severely threatens marine wildlife (Sheavly and Register, 2007) as it can entangle and/or be ingested by organisms such as mammals, fish, seabirds, and turtles, affecting their fitness, breeding, and survival (Kühn *et al.*, 2015). There is evidence that even sessile organisms such as corals can ingest microplastic (Hall *et al.*, 2015; Rotjan *et al.*, 2019). Marine debris can also facilitate the dispersal of organisms by rafting, thus promoting the introduction of invasive species (Gregory, 2009; Kiessling *et al.*, 2015). Therefore, regardless of its origin, marine debris negatively affects the local biodiversity of oceanic islands and their conservation (Dias, 2016). In addition, tourism and local economies relying on it can be negatively impacted by the presence of debris (Jang *et al.*, 2014).

The Fernando de Noronha Archipelago (FNA), 360 km off the northeastern coast of Brazil (3º51' S; 32º25' W), is formed by the main island of Fernando de Noronha (18 km^2) plus 20 other smaller islands and islets. The FNA comprises two protected areas: a multiple-use Environmental Protection Area, which covers the inhabited part of the main island (~30%) and allows for human occupation and sustainable use of its resources; and a Marine National Park, a no-take area covering 70% of the main island and all secondary islands. Considered a glamorous national and international destination, the FNA receives an increasing number of tourists each year. In 2018, the number of tourists exceeded 100,000 (ICMBio, 2018). In addition, there are 3,000 residents, including permanent residents and temporary workers (IBGE, 2020). The multiple-use area encompasses all the urban facilities, such as streets, guesthouses, restaurants, banks, a hospital, and an airport, as well as historical constructions and beaches with bars. Visitation can thus be done without many restrictions. The remaining 70% of the main island is a no-take area, with more deserted beaches and forest trails that can be accessed only under supervision of a trained guide. Tourists must pay a fee to visit the no-take area and previous scheduling is mandatory to access some of the areas.

The entire coast of the multiple-use area and a part of the no-take area are located on the leeward coast of the archipelago, which faces north and is sheltered from ocean currents and winds. The windward coast of the main island, comprising the no-take area, faces south and is influenced by relatively strong winds (Matheus *et al.*, 2019) and the South Equatorial Current (Lumpkin and Garzoli, 2005). These differences in exposure levels between the leeward and windward sides of the FNA affect the human activities performed on the island, like fishing and diving, and the insular biodiversity (Batistella, 1996; Matheus *et al.*, 2019).

Previous studies on marine debris in Fernando de Noronha have focused mainly on microplastics (< 5 mm; Ivar do Sul *et al.*, 2017; Monteiro *et al.*, 2020). Although some have included larger plastic debris (< 100 mm; Ivar do Sul *et al.*, 2009), sampling items included mostly fibers and strands (Ivar do Sul *et al.*, 2009; Ivar do Sul *et al.*, 2014). Therefore, the objective of this study was to describe the distribution, composition, and abundance of marine macro-debris, including plastic, paper, glass, and metal, on the sheltered and exposed shores of the FNA. The southern coast of the island is expected to present more marine debris because of its oceanographic characteristics, despite the protection status granted by the Marine National Park. This assessment comparing different types of marine macro-debris among sites complements the knowledge already available on the composition of marine debris, serves as a baseline for future monitoring programs, and may contribute to evaluate and adjust strategies for the management of marine debris in the FNA.

This study was conducted on five no-take sites in the windward coast of the FNA and one site within the leeward multiple-use area. The no-take sites were Abreus (13,000 m^2), Atalaia (10,600 m^2), Alagados da Raquel (30,000 m^2), Leão (22,000 m^2), and Sueste (10,200 m^2), and the multiple use site was Conceição (28,000 m^2) (Fig. 1). Alagados da

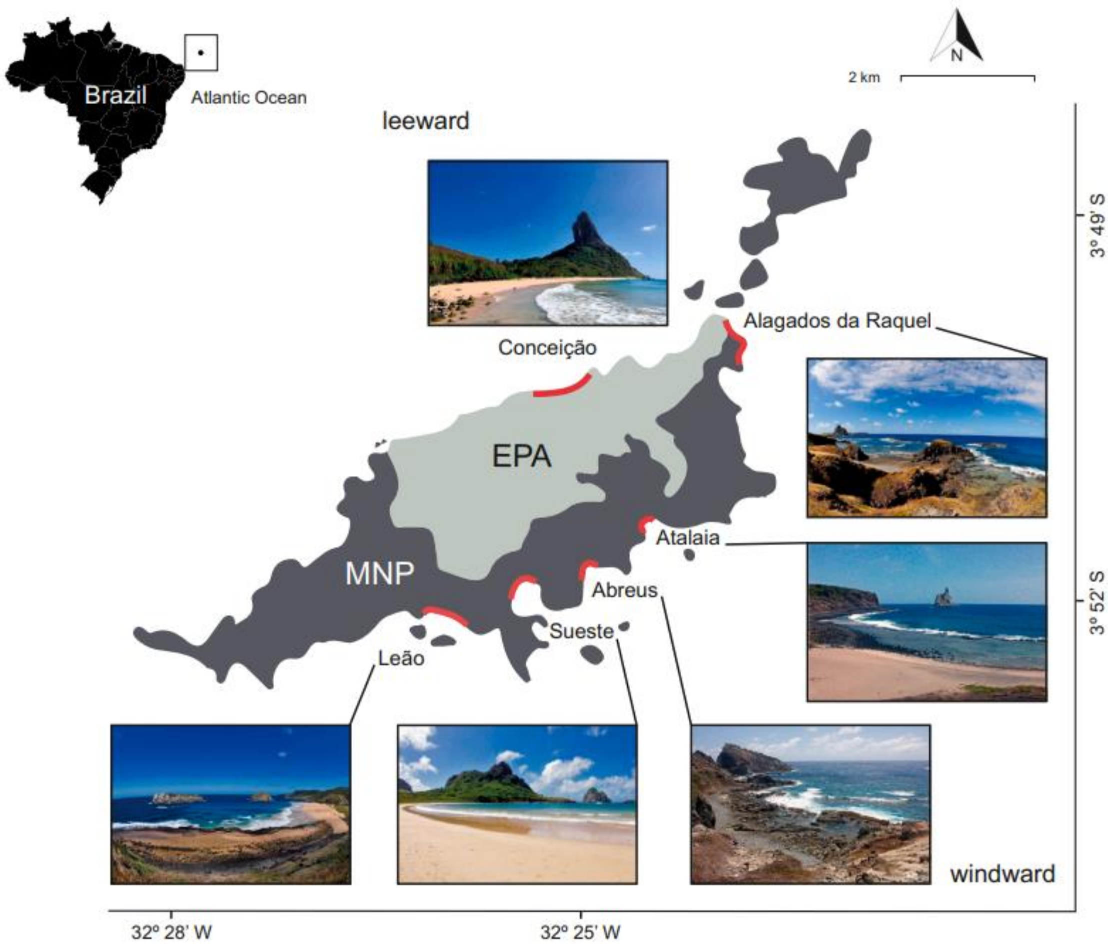


Figure 1. Study sites in the Fernando de Noronha Archipelago (3º51' S; 32º25' W). Protected areas: multiple-use Environmental Protection Area (EPA) and no-take Marine National Park (MNP). Extension of the sampled sites in the leeward and windward coast of the main island (in red). Photos: Grillo, A.C.; Pires, J.M; Teixeira, F.C.

Raquel is a restricted access shore platform where only researchers can enter with previous authorization. Abreus is another shore platform and Atalaia is a sandy beach with a neighbouring shore platform. The main attraction of Abreus and Atalaia are tide pools in which the number of visitors (tourists and residents) and the time for free dive are limited. Leão is a beach visited by few people due to difficulty in access and Sueste is a highly visited bay with a free diving area. Conceição is an easily accessible beach with bars heavily visited by tourists wishing to perform leisure activities.

The sampling sites are enclosed by natural features such as cliffs and dense vegetation, that act as boundaries. The entire area of each sampling site was surveyed from the edge of the water to the beginning of the vegetated area or cliffs during low tide and all marine macro-debris (> 2.5 cm) were collected. Samplings were carried out from September 2016 to September 2018. Marine debris was weighted and categorized into four broad groups: plastic, paper, glass, and metal. Plastics were further classified into four subcategories: bottle caps, disposable plastics (cups, cutlery), hospital waste (plastic syringes), and cigarette butts (based on UNEP/IOC marine debris classification, UN Environmental Programme/International Oceanic Commission; Cheshire *et al.*, 2009). For these subcategories, the number of items were counted. Other kinds of plastic objects and fragments were generically classified as "plastic".

Since the area varies among sites, data was standardized by dividing the total weight and the amount of each debris category by the area of each site, resulting in a comparable value by m^2. Kruskal-Wallis tests were used to test for differences on the weight and amount of marine debris categories and subcategories among the sites and between the windward and leeward coasts of the FNA using samples as replicates. A non-metric multidimensional scaling (MDS) ordination was used to summarize spatial similarities (Bray-Curtis) on the composition and abundance of plastic marine debris subcategories, and a separate one-way analysis of similarity (ANOSIM) was used to test for differences according to sampling sites.

In sum, 294.5 kg of plastic, 15.3 kg of glass, 8.2 kg of metal, and 3 kg of paper were collected. Plastic was the most abundant category of marine debris. Plastic weight differed among the sampling sites (Kruskal-Wallis H=33.55, $p < 0.001$), and the sites with higher and lower mean weight of plastics were Atalaia and Conceição, respectively (Fig. 2). The amount of plastic debris subcategories also differed among the sampling sites (Kruskal-Wallis: bottle caps H=21.22, $p < 0.001$; disposable plastics H=22.64, $p < 0.001$; hospital waste H=19.94, $p < 0.01$; cigarette butts H=23.98, $p < 0.001$). Although bottle caps and disposable plastics were collected in all sites, the higher amounts of bottle caps were recorded in Atalaia, Sueste, and Abreus, and disposable plastics were mostly found in Conceição (Table 1, Fig. 3). Hospital waste was collected in all sites, except in Conceição. Cigarette butts were mostly found in Conceição, and among the other sites, 66% of all cigarette butts belonged to Sueste. The MDS ordination diagram showed a distinction among the sampling sites for the plastic subcategories, with plastic debris of Conceição clustering apart from the remaining sites (Fig. 4; ANOSIM: R=0.3, p = 0.001).

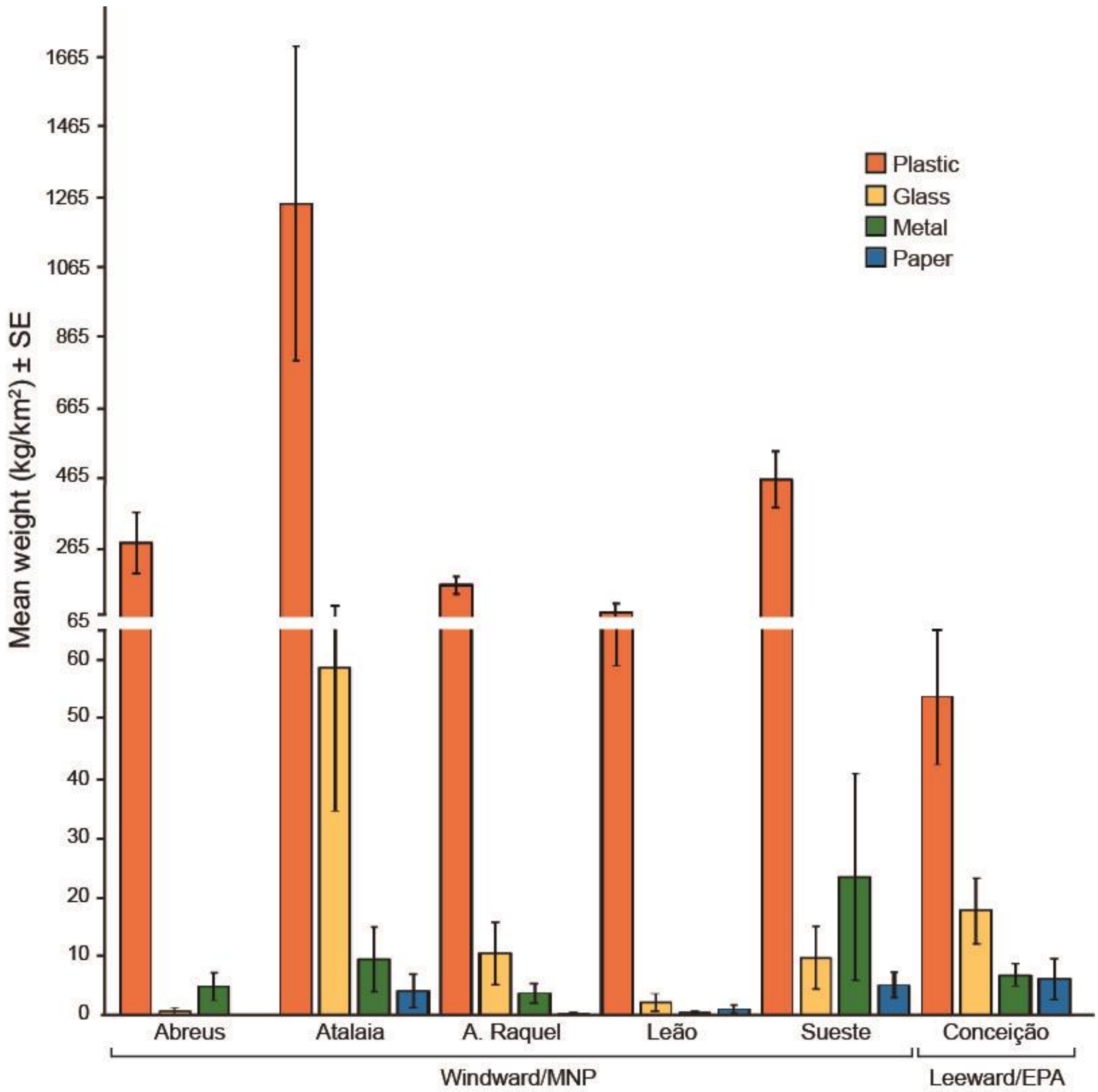


Figure 2. Weight (mean ± SE) of marine macro-debris categories in the windward no-take Marine National Park (MNP) and leeward multiple-use Environmental Protection Area (EPA) sites.

Table 1. Mean amount (items/km^2) of the plastic marine debris subcategories counted on the sampled sites in the windward no-take Marine National Park (MNP) and leeward multiple-use Environmental Protection Area (EPA) coasts of Fernando de Noronha.

Plastic debris subcategories	Mean amount (items/km^2) ± SE					
	Windward (MNP)					Leeward (EPA)
	Abreus	Atalaia	A. Raquel	Leão	Sueste	Conceição
Bottle caps	23,673.08 ± 8,881.88	51,442.05 ± 30,467.13	14,356.41 ± 2,813.60	8,015.15 ± 3,284.63	39,908.96 ± 9,485.59	842.86 ± 252.04
Disposable plastics	25.64 ± 25.64	121.29 ± 49.21	28.21 ± 17.64	5.05 ± 5.05	42.02 ± 22.31	196.43 ± 59.11
Hospital waste	25.64 ± 17.29	229.11 ± 137.88	51.28 ± 25.77	75.76 ± 43.52	175.07 ± 53.58	0
Cigarette butts	0	0	12.82 ± 10.36	35.35 ± 19.72	168.07 ± 79.23	732.14 ± 255.90

Fig. 3. Marine macro-debris collected in shores of Fernando de Noronha: windward coast samplings (a, b, d, e); leeward coast sampling (c); hospital waste, including a syringe with a needle (d); packages of brands not commercialized in Brazil, in foreign languages (e). Photos: Furlani, F., Grillo, A.C., Santana, J.

The mean weight (g/m^2) of glass (Kruskal-Wallis: H=21.63, p < 0.001) and paper (Kruskal-Wallis: H=19.96, p < 0.01) was different among the sampled sites, and metal debris did not differ among the sites (Kruskal-Wallis: H=5.42, $p > 0.05$; Fig. 2). Glass was mostly represented by beer and other alcoholic beverage bottles. In Conceição, 80% of the surveys contained paper (napkins and food packages; Fig. 3), followed by Sueste and Atalaia (64% and 57% of the surveys respectively). Metal was represented by beer and soda cans, wires, and bottle caps. Additionally, packages of identifiable brands that are not commercialized in Brazil, in foreign languages (mostly Asiatic languages, but also English and Spanish), were collected only in the windward sites, totalizing 115 items among all samples. They were mostly represented by plastic beverage bottles in a low state of decay, but also included carton packaging and metal bottle caps (Fig. 3).

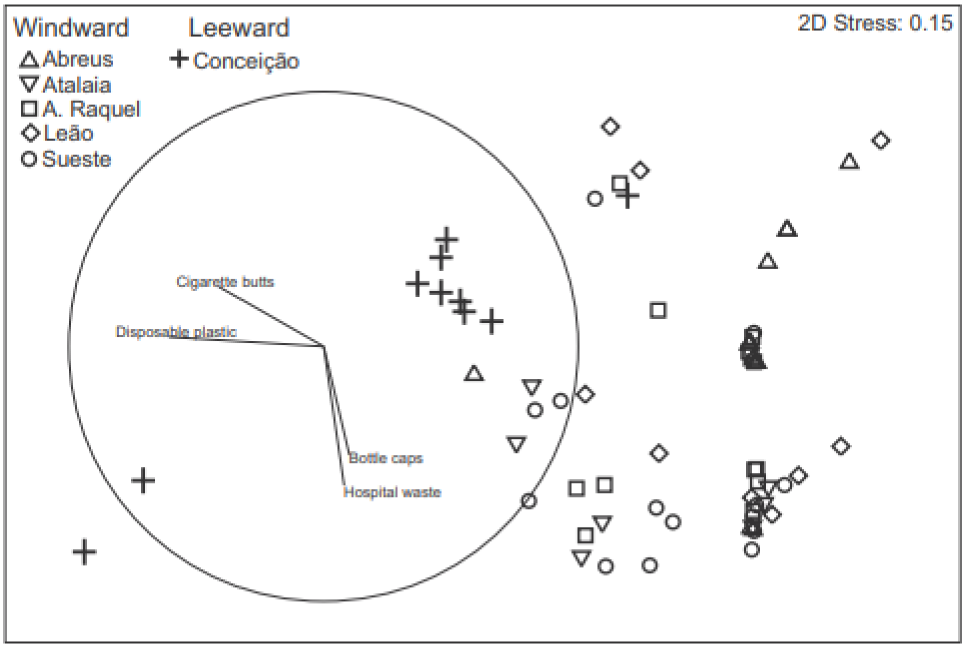


Fig. 4. Multidimensional scaling (MDS) of the composition and abundance of plastic marine debris subcategories between windward and leeward sampling sites of Fernando de Noronha based on Bray-Curtis similarities.

The mean total weight of marine debris on the windward coast was significantly higher than on the leeward coast (Table 2). Since the sampling effort was higher on the windward coast, the same analyses were done after randomly sampling 1000 times the windward data by bootstrap to obtain equally sized samples in both coasts (n=10) and the pattern remained. The mean weight of plastic debris was significantly higher on the windward than on the leeward coast. Also, the mean debris amount regarding bottle caps and hospital waste was significantly higher on the windward coast. On the leeward coast, mean amounts of disposable plastics and cigarette butts were significantly higher. The mean weight (g/m^2) of glass, paper, and metal debris was not significantly different between the leeward and the windward coasts.

Table 2. Mean total abundance of marine macro-debris in the windward no-take Marine National Park (MNP) and leeward multiple-use Environmental Protection Area (EPA) coasts of Fernando de Noronha. Significance levels for differences between windward and leeward coasts are indicated (* $p < 0.005$, ** $p < 0.001$).

Marine debris	Mean abundance ± SE		
	Windward (MNP)	Leeward (EPA)	
Total debris (kg/km^2)	411.64 ± 77.97	84.62 ± 16.08	*
Plastic (kg/km^2)	387.51 ± 77.15	53.75 ± 11.49	**
Bottle caps (items/km^2)	26,575.75 ± 5,181.77	842.86 ± 252.04	**
Disposable plastics (items/km^2)	39.22 ± 11.47	196.43 ± 59.11	**
Hospital waste (items/km^2)	103.84 ± 25.28	0	*
Cigarette butts (items/km^2)	51.6 ± 22.09	732.14 ± 255.9	**
Glass (kg/km^2)	12.90 ± 4.18	17.62 ± 5.58	
Metal (kg/km^2)	9.14 ± 4.56	6.79 ± 1.88	
Paper (kg/km^2)	2.07 ± 0.70	6.09 ± 3.43	

The composition and abundance of marine macro-debris differed according to the distinct characteristics of the sampled sites in the FNA. The windward coast, a no-take Marine National Park, presented higher plastic abundance than the leeward coast, a multiple-use Environmental Protection Area. Plastic bottle caps and hospital waste were collected in higher amounts in the windward sites. Contrarily, the leeward site presented a higher density of disposable plastics and cigarette butts.

Higher abundance of plastic debris on the windward coast may be explained by the transportation of plastic from its origin to the FNA through oceanic forces, such as winds and currents (Barnes *et al.*, 2009). Bottle caps abundance, for example, was more than twenty-fold greater on the windward than on the leeward coast of the FNA, which can be explained by their efficiency in being transported by the ocean (Andrady, 2011). Some polymers such as polyethylene and polypropylene, used for making bottle caps, plastic bags, bottles, and straws, are less dense than seawater and therefore float in the ocean until they are washed ashore on beaches, sometimes far away from where they entered the sea (Andrady, 2011). The lightweight, high buoyancy, elevated durability, and super slow environmental degradation of plastic objects (from 58 years for bottles to 1200 years for pipes; Chamas *et al.*, 2020) make them perfect to accumulate in the ocean (Andrady, 2011, 2015).

Moreover, hospital waste was completely absent in leeward surveys. There is only one public hospital on the FNA, which is located in the urban area and closer to the leeward coast, and there are no perennial watercourses on the island, turning less plausible the possibility of debris transportation from the hospital to windward shores. In addition, the fact that one of the sites surveyed in the windward side (Alagados da Raquel) is closed for public use suggests that most of the marine debris on its shores might have come from elsewhere. Finally, the presence of packages with foreign language labels at early decomposition states among the debris collected only in windward sites also points to the hypothesis that some of these objects might have been thrown overboard and then traveled through currents (Duhec *et al.*, 2015). These results agree with a previous study conducted in the FNA (Ivar do Sul *et al.*, 2009), and previous studies of insular shores facing winds and oceanic currents elsewhere, such as in the Seychelles (Duhec *et al.*, 2015), Easter Island and Salas & Gómez, Chile (Miranda-Urbina *et al.*, 2015), Trindade Island, Brazil (Andrades *et al.*, 2018), and Hawaiian Archipelago, USA (McDermid and McMullen, 2004; Ribic *et al.*, 2012), evidencing a higher vulnerability to debris on windward sides of isolated locations.

The marine debris recorded in Conceição beach, on the leeward coast, may have originated from local uses and infrastructure. The higher quantity of disposable plastics, such as single-use cups and cutlery, and cigarette butts suggests that this kind of debris was generated locally by beach users. Similarly, other studies have pointed touristic use of urban beaches as the cause of accumulation of this kind of debris on oceanic and continental islands (Martinez-Ribes *et al.*, 2007; de Scisciolo *et al.*, 2016; Wilson and Verlis, 2017). Together, these results indicate improper disposal of local debris in the FNA and elsewhere.

Cigarette butts are usually one of the top three most abundant items collected in beach surveys and clean-ups on islands (Martinez-Ribes *et al.*, 2007; Pieper *et al.*, 2015; de Scisciolo *et al.*, 2016; Wilson and Verlis, 2017). Although in the windward sampling sites of the FNA the abundance of cigarette butts was much lower than in the leeward site, most of them were collected at Sueste, which is explained by the far greater number of visitors compared to other windward sites (Sueste: ~98,000; Atalaia: ~ 20,000; Abreus ~ 6,000 visitors/year; ICMBio, 2018). Despite Alagados da Raquel being a restricted access site, the presence of cigarette butts suggests that they might have been thrown from a viewpoint located just above the cliff. Installing public ashtrays in the viewpoints and handing out portable ashtrays at the beaches can help diminish the number of cigarette butts littered in inadequate areas and raise smokers' concern about their behavior as polluters (Araújo and Costa, 2019).

Inside the windward no-take area, paper was more frequent in Sueste, represented by food packages, sales receipts, and diving equipment rental receipts. The only marine activity allowed in Sueste is free diving, which explains the high quantity of equipment rent tickets surveyed. Sueste was the only sampling site in the windward coast to bear an Information and Control Point in front of the beach (when the study was conducted), which offers food, bathroom, and store services for visitors. Therefore, these results suggest the role of visitors in the presence and density of marine debris on insular shores. After concluding the samplings, another Control Point was inaugurated in Leão beach, and a viewpoint located in the leeward Boldró beach has been conceded for the construction of a bar. It is important to keep marine debris monitoring in the long term to verify if the installation of these new infrastructures will cause an increase in the amount of debris related to beach users, and take steps to minimize the problem if necessary.

Despite being a no-take and supervised Marine National Park, the high abundance of marine debris on the windward coast of Fernando de Noronha poses negative impacts to marine biodiversity. During this study, a classic example of entanglement was recorded in Atalaia. A plastic debris collar was observed in a reef resident grunt (*Haemulon parra*) for several days, limiting the movements of its pelvic fins (Fig. 5a). This kind of entanglement has already been recorded in other Brazilian locations (Sazima *et al.*, 2002; Nunes *et al.*, 2018), threatening the supposedly protected marine fauna and implying a risk for conservation efforts. Consequently, the protection status of an area does not prevent the ubiquity of marine debris nor protect its inhabitants from the negative effects of this type of pollution (Barnes *et al.*, 2018; Luna-Jorquera *et al.*, 2019).

Another example of inappropriate debris disposal that could affect the marine fauna was documented in Porto beach, located in the leeward multiple-use area, a site highly visited by residents and tourists. This is the only port on the island, where all vessels arrive and depart, and several activities are performed, including scuba diving due to the abundant marine life and to a shallow shipwreck nearby. In 2016 a high quantity of underwater debris was recorded alongside the pier, composed mainly of tires, ropes, plastic big bags, building materials, single-use plastics, and cans (Fig. 5b). This debris points to several anthropogenic in-situ sources, such as island supply (i.e., objects falling

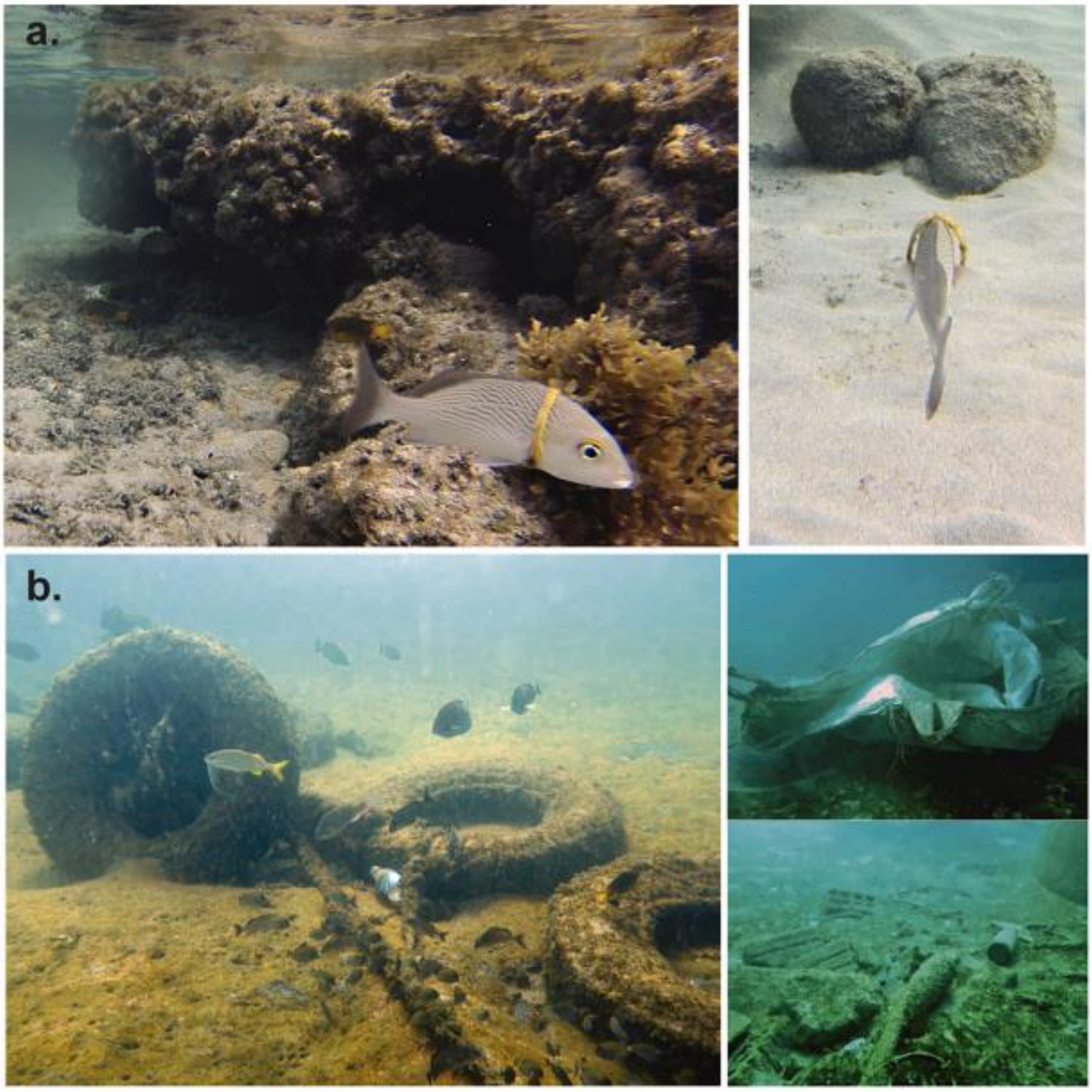


Fig. 5. Sailor´s grunt (*Haemulon parra*) with a plastic debris collar in Atalaia's tide pool, in the windward no-take area (a); debris disposal recorded underwater in Porto beach, in the leeward multiple-use area of Fernando de Noronha (b). Photos: Grillo, A.C.; Mello, T.J.

from ships while transferred to the island and vice versa), pier construction, and recreational activities by beach users. These examples add information to the managers, citizens, and tourists for an improvement of the waste disposal and accumulation on the island. Moreover, ineffective urban solid waste management is a recurrent environmental problem in the FNA, with a monthly average production of about 300 tons of garbage on the island (Fioravanso, 2017) mainly due to the large number of tourists throughout the year.

Despite being internationally known as a glamorous ecotourism destination, a Protected

Area, a UNESCO World Heritage and Ramsar Wetland Site, and having a beach ranked among the most beautiful in the world, Fernando de Noronha suffers from anthropogenic debris on its shores. A large amount of marine debris, especially plastic, was surveyed in all sites. The distinct debris composition and abundance along the coast of Fernando de Noronha could indicate that debris accumulation on its shores is influenced by the exposure to winds and currents and by the increasing number of tourists to the island. It also evidences that distance from the coast and protection status do not prevent pollution by marine debris on oceanic islands. These pollutants present a threat to the entire ecosystem, posing consequences from the marine wildlife to social and economic levels, and also impacting tourism (Cristiano *et al.*, 2020).

In 2018 a decree was published (District Decree nº 002/2018) forbidding the use and commercialization of single-use plastics in the island. Therefore, this study serves as a baseline to evaluate the effectiveness and orient improvements of this and other environmental measures adopted in Noronha to reduce the input of debris to its shores and the sea. Socio-environmental programs, mainly directed to tourists and beach users, international cooperation, enforced supervision, and integrated prevention strategies are urgent and essential to address the marine debris question worldwide and thus reduce its accumulation, production, and impacts on the environment.

CRediT authorship contribution statement

Ana Carolina Grillo and Thayná Jeremias Mello contributed equally in this study, from the conception and methods development to data analyses and manuscript preparation.

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.

Acknowledgments

We thank ICMBio Noronha Volunteer Program for supporting and continuing this monitoring and all the volunteers and workers of NGI Noronha and EcoNoronha who helped to collect data for this study, specially Francielly Furlani and Jéssika Santana. We also thank Vinícius de Souza for providing the map of the archipelago, Guilherme O. Longo, Ronaldo B. Francini-Filho, and Ryan Andrades for their important contributions.

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

References

Andrades, R., Santos, R.G., Joyeux, J.C., Chelazzi, D., Cincinelli, A., Giarrizzo, T., 2018. Marine debris in Trindade Island, a remote island of the South Atlantic. Mar. Pollut. Bull. 137, 180–184. https://doi.org/10.1016/j.marpolbul.2018.10.003

Andrady, A.L., 2011. Microplastics in the marine environment. Mar. Pollut. Bull. 62, 1596–1605. https://doi.org/10.1016/j.marpolbul.2011.05.030

Andrady, A.L., 2015. Persistence of plastic litter in the oceans, in: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer International Publishing, Cham, pp. 57–72. https://doi.org/10.1007/978-3-319-16510-3_3

Araújo, M.C.B., Costa, M.F., 2019. A critical review of the issue of cigarette butt pollution in coastal environments. Environ. Res. 172, 137–149. https://doi.org/10.1016/j.envres.2019.02.005

Barnes, D.K.A., 2005. Remote islands reveal rapid rise of southern hemisphere sea debris. Cient. World J. 5, 915–921. https://doi.org/10.1100/tsw.2005.120

Barnes, D.K.A., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. Philos. Trans. R. Soc. B Biol. Sci. 364, 1985–1998. https://doi.org/10.1098/rstb.2008.0205

Barnes, D.K.A., Morley, S.A., Bell, J., Brewin, P., Brigden, K., Collins, M.A., Glass, T., Goodall-Copestake, W.P., Henry, L., Laptikhovsky, V., Piechaud, N., Richardson, A.J., Rose, P., Sands, C.J., Schofield, A., Shreeve, R.S., Small, A., Stamford, T., Taylor, B., 2018. Marine plastics threaten giant Atlantic marine protected areas. Curr. Biol. 28, 1137–1138. https://doi.org/10.1016/j.cub.2018.08.064.

Batistella, M., 1996. Espécies vegetais dominantes do Arquipélago de Fernando de Noronha: grupos ecológicos e repartição espacial. Acta Bot. Bras. 10, 223–235. http://doi.org/10.1590/S0102-33061996000200003

Chamas, A., Moon, H., Zheng, J., Qiu, J., Tabassum, T., Jang, H.J., Abu-Omar, M., Scott, S.L., Suh, S., 2020. Degradation rates of plastics in the environment. ACS Sustain.

Chem. Eng. 8, 3494–3511. https://pubs.acs.org/doi/10.1021/acssuschemeng.9b06635

Carson, H.S., Lamson, M.R., Nakashima, D., Toloumu, D., Hafner, J., Maximenko, N., McDermid, K.J., 2013. Tracking the sources and sinks of local marine debris in Hawai'i. Mar. Environ. Res. 84, 76–83. https://doi.org/10.1016/j.marenvres.201

Cheshire, A.C., Adler, E., Barbière, J., Cohen, Y., Evans, S., Jarayabhand, S., Jeftic, L., Jung, R.T., Kinsey, S., Kusui, E.T., Lavine, I., Manyara, P., Oosterbaan, L., Pereira, M.A., Sheavly, S., Tkalin, A., Varadarajan, S., Wenneker, B., Westphalen, G., 2009. UNEP/IOC Guidelines on Survey and Monitoring of Marine Litter. UNEP Regional Seas Reports and Studies, No. 186; IOC Technical Series No. 83: xii. 120 pp.

Cristiano, S.C., Rockett, G.C., Portz, L.C., Souza Filho, J.R., 2020. Beach landscape management as a sustainable tourism resource in Fernando de Noronha Island (Brazil). Mar. Pollut. Bull. 150, 110621. https://doi.org/10.1016/j.marpolbul.2019.110621

de Scisciolo, T., Mijts, E.N., Becker, T., Eppinga, M.B., 2016. Beach debris on Aruba, Southern Caribbean: Attribution to local land-based and distal marine-based sources. Mar. Pollut. Bull. 106, 49–57. https://doi.org/10.1016/j.marpolbul.2016.03.039

Dias, B.F. de S., 2016. Marine debris: understanding, preventing and mitigating the significant adverse impacts on marine and coastal biodiversity. CBD Technical Series 83.

Donohue, M.J., Boland, R.C., Sramek, C.M., Antonelis, G.A., 2001. Derelict fishing gear in the Northwestern Hawaiian Islands: diving surveys and debris removal in 1999 confirm threat to coral reef ecosystems. Mar. Pollut. Bull. 42, 1301–1312. https://doi.org/10.1016/S0025-326X(01)00139-4

Duhec, A. V., Jeanne, R.F., Maximenko, N., Hafner, J., 2015. Composition and potential origin of marine debris stranded in the Western Indian Ocean on remote Alphonse Island, Seychelles. Mar. Pollut. Bull. 96, 76–86. https://doi.org/10.1016/j.marpolbul.2015.05.042

Eriksson, C., Burton, H., Fitch, S., Schulz, M., van den Hoff, J., 2013. Daily accumulation rates of marine debris on sub-Antarctic island beaches. Mar. Pollut. Bull. 66, 199–208. https://doi.org/10.1016/j.marpolbul.2012.08.026

Fioravanso, A.G., 2017. Conflitos socioambientais em áreas marinhas protegidas: o contexto da gestão ambiental pública no Arquipélago de Fernando de Noronha, Brasil. Universidade Federal do Rio Grande. URL https://gerenciamentocosteiro.furg.br/images/dissertacoes/056-Aline-Guzenski-Fioravanso.pdf (accessed 12.10.20).

Gregory, M.R., 2009. Environmental implications of plastic debris in marine settings—entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. Philos. Trans. R. Soc. B Biol. Sci. 364, 2013–2025. https://doi.org/10.1098/rstb.2008.0265

Hall, N.M., Berry, K.L.E., Rintoul, L., Hoogenboom, M.O., 2015. Microplastic ingestion by scleractinian corals. Mar. Biol. 162, 725–732. https://doi.org/10.1007/s00227-015-2619-7

Halpern, B.S., Frazier, M., Potapenko, J., Casey, K.S., Koenig, K., Longo, C., Lowndes, J.S., Rockwood, R.C., Selig, E.R., Selkoe, K.A, Walbridge, S., 2015. Spatial and temporal changes in cumulative human impacts on the world's ocean. Nat. Commun. 6, 1–7. https://doi.org/10.1038/ncomms8615

IBGE – Instituto Brasileiro de Geografia e Estatística, 2020. URL https://www.ibge.gov.br/cidades-e-estados/pe/fernando-de-noronha.html (accessed 12.10.20).

ICMBio – Instituto Chico Mendes de Conservação da Biodiversidade, 2018. Management report ICMBio Noronha. Fernando de Noronha, PE.

Ivar do Sul, J.A., Costa, M.F., 2007. Marine debris review for Latin America and the wider Caribbean region: From the 1970s until now, and where do we go from here? Mar. Pollut. Bull. 54, 1087–1104. https://doi.org/10.1016/j.marpolbul.2007.05.004

Ivar do Sul, J.A., Costa, M.F., Fillmann, G., 2014. Microplastics in the pelagic environment around oceanic islands of the Western Tropical Atlantic Ocean. Water Air Soil Pollut. 225, 2004. https://doi.org/10.1007/s11270-014-2004-z

Ivar do Sul, J.A., Costa, M.F., Fillmann, G., 2017. Occurrence and characteristics of microplastics on insular beaches in the Western Tropical Atlantic Ocean. PeerJ Preprints. 5, e2901v1. https://doi.org/10.7287/peerj.preprints.2901v1

Ivar do Sul, J.A., Spengler, A., Costa, M.F., 2009. Here, there and everywhere. Small plastic fragments and pellets on beaches of Fernando de Noronha (Equatorial Western Atlantic). Mar. Pollut. Bull. 58, 1236–1238. https://doi.org/10.1016/j.marpolbul.2009.05.004

Jang, Y.C., Hong, S., Lee, J., Lee, M.J., Shim, W.J., 2014. Estimation of lost tourism revenue in Geoje Island from the 2011 marine debris pollution event in South Korea. Mar. Pollut. Bull. 81, 49–54. https://doi.org/10.1016/j.marpolbul.2014.02.021

Kiessling, T., Gutow, L., Thiel, M., 2015. Marine litter as habitat and dispersal vector, in: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer International Publishing, Cham, pp. 141–181. https://doi.org/10.1007/978-3-319-16510-3_6

Kühn, S., Bravo Rebolledo, E.L., van Franeker, J.A., 2015. Deleterious effects of litter on marine life, in: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer International Publishing, Cham, pp. 75–116. https://doi.org/10.1007/978-3-319-16510-3_4

Lavers, J.L., Bond, A.L., 2017. Exceptional and rapid accumulation of anthropogenic debris on one of the world’s most remote and pristine islands. Proc. Natl. Acad. Sci. U. S. A. 114, 6052–6055. https://doi.org/10.1073/pnas.1619818114

Lumpkin, R., Garzoli, S.L., 2005. Near-surface circulation in the Tropical Atlantic Ocean. Deep. Res. Part I Oceanogr. Res. Pap. 52, 495–518. https://doi.org/10.1016/j.dsr.2004.09.001

Luna-Jorquera, G., Thiel, M., Portflitt-Toro, M., Dewitte, B., 2019. Marine protected areas invaded by floating anthropogenic litter: An example from the South Pacific. Aquat. Conserv.: Mar. Freshw. Ecosyst. 29, 245–259. https://doi.org/10.1002/aqc.3095

Martinez-Ribes, L., Basterretxea, G., Palmer, M., Tintoré, J., 2007. Origin and abundance of beach debris in the Balearic Islands. Sci. Mar. 71, 305–314. https://doi.org/10.3989/scimar.2007.71n2305

Matheus, Z., Francini-Filho, R.B., Pereira-Filho, G.H., Moraes, F.C., de Moura, R.L., Brasileiro, P.S., Amado-Filho, G.M., 2019. Benthic reef assemblages of the Fernando de Noronha Archipelago, tropical South-west Atlantic: Effects of depth, wave exposure and cross-shelf positioning. PLoS One 14, 1–15. https://doi.org/10.1371/journal.pone.0210664

McDermid, K.J., McMullen, T.L., 2004. Quantitative analysis of small-plastic debris on beaches in the Hawaiian archipelago. Mar. Pollut. Bull. 48, 790–794. https://doi.org/10.1016/j.marpolbul.2003.10.017

Miranda-Urbina, D., Thiel, M., Luna-Jorquera, G., 2015. Litter and seabirds found across a longitudinal gradient in the South Pacific Ocean. Mar. Pollut. Bull. 96, 235–244. https://doi.org/10.1016/j.marpolbul.2015.05.021

Monteiro, R.C.P., Ivar do Sul, J.A., Costa, M.F., 2020. Small microplastics on beaches of Fernando de Noronha Island, Tropical Atlantic Ocean. Ocean Coast. Res. 68. http://dx.doi.org/10.1590/s2675-28242020068235

Nunes, J. de A.C.C., Sampaio, C.L.S., Barros, F., Leduc, A.O.H.C., 2018. Plastic debris collars: An underreported stressor in tropical reef fishes. Mar. Pollut. Bull. 129, 802–805. https://doi.org/10.1016/j.marpolbul.2017.10.076

Pérez-Venegas, D., Pavés, H., Pulgar, J., Ahrendt, C., Seguel, M., Galbán-Malagón, C.J., 2017. Coastal debris survey in a Remote Island of the Chilean Northern Patagonia. Mar. Pollut. Bull. 125, 530–534. https://doi.org/10.1016/j.marpolbul.2017.09.026

Pieper, C., Ventura, M.A., Martins, A., Cunha, R.T., 2015. Beach debris in the Azores (NE Atlantic): Faial Island as a first case study. Mar. Pollut. Bull. 101, 575–582. https://doi.org/10.1016/j.marpolbul.2015.10.056

Ribic, C.A., Sheavly, S.B., Klavitter, J., 2012. Baseline for beached marine debris on Sand Island, Midway Atoll. Mar. Pollut. Bull. 64, 1726–1729. https://doi.org/10.1016/j.marpolbul.2012.04.001

Quimbayo, J.P., Dias, M.S., Kulbicki, M., Mendes, T.C., Lamb, R.W., Johnson, A.F., Aburto-Oropeza, O., Alvarado, J.J., Bocos, A.A., Ferreira, C.E.L., Garcia, E., Luiz, O.J., Mascareñas-Osorio, I., Pinheiro, H.T., Rodriguez-Zaragoza, F., Salas, E., Zapata, F.A., Floeter, S.R., 2019. Determinants of reef fish assemblages in tropical Oceanic islands. Ecography 42, 77–87. https://doi.org/10.1111/ecog.03506

Rotjan, R.D., Sharp, K.H., Gauthier, A.E., Yelton, R., Lopez, E.M.B., Carilli, J., Kagan, J.C., Urban-Rich, J., 2019. Patterns, dynamics and consequences of microplastic ingestion by the temperate coral, *Astrangia poculata*. Proc. R. Soc. B Biol. Sci. 286, 20190726. https://doi.org/10.1098/rspb.2019.0726

Sazima, I., Gadig, O.B.F., Namora, R.C., Motta, F.S., 2002. Plastic debris collars on juvenile carcharhinid sharks (*Rhizoprionodon lalandii*) in southwest Atlantic. Mar. Pollut. Bull. 44, 1147–1149. https://doi.org/10.1016/S0025-326X(02)00141-8

Sheavly, S.B., Register, K.M., 2007. Marine debris and plastics: Environmental concerns, sources, impacts and solutions. J. Polym. Environ. 15, 301–305. http://doi.org/10.1007/s10924-007-0074-3

Wilson, S.P., Verlis, K.M., 2017. The ugly face of tourism: Marine debris pollution linked to visitation in the southern Great Barrier Reef, Australia. Mar. Pollut. Bull. 117, 239–246. https://doi.org/10.1016/j.marpolbul.2017.01.036

CAPÍTULO 2

Drivers of temporal variation in benthic cover and coral health of an oceanic intertidal reef in Southwestern Atlantic

Capítulo 2

Drivers of temporal variation in benthic cover and coral health of an oceanic intertidal reef in Southwestern Atlantic*

Thayná Jeremias Mello[a,b,✉], Edson Aparecido Vieira[a], Amana Guedes Garrido[c,d], Carla Zilberberg[c,e], Juliana Lopes De Lima[c], Lucas Penna Soares Santos[f], Guilherme Ortigara Longo[a]

[a]*Department of Oceanography and Limnology, Universidade Federal Do Rio Grande Do Norte, Natal, RN, 59014-002, Brazil*

[b] *Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio), Núcleo de Gestão Integrada Fernando de Noronha, PE, 53990-970, Brazil*

[c]*Instituto de Biodiversidade e Sustentabilidade, Universidade Federal do Rio de Janeiro (NUPEM-UFRJ), Macaé, RJ, 27965-045, Brazil*

[d]*Centro de Biologia Marinha, Universidade de São Paulo, São Sebastião, SP, 11612-109, Brazil*

[e] *Instituto Coral Vivo, Rio de Janeiro, RJ, 20940-040, Brazil*

[f] *Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio), Centro Nacional de Pesquisa e Conservação de Aves Silvestres / Núcleo de Gestão Integrada Fernando de Noronha, PE, 53990-970, Brazil*

✉*Corresponding author: thayna.mello@icmbio.gov.br*

*Manuscript published in *Regional Studies in Marine Science* Volume 60, June 2023, 102874

https://doi.org/10.1016/j.rsma.2023.102874

Abstract

Intertidal environments and shallow reefs are subject to natural disturbances and impacts by humans that influence their dynamics and therefore their conservation. These environments are among the most subject to anthropogenic impacts worldwide, ranging from global warming to local pollution and disordered tourism. Fernando de Noronha Marine National Park is an important Marine Protected Area that covers marine habitats up to 50m deep of an oceanic archipelago off North East Brazil. It is a no-take area, so the main potential impacts in intertidal and shallow environments there are related to tourism, pollution and climate change. One of the chief tourist attractions in this Park is a tide pool where snorkelers can dive under a series of regulations. We monitored corals and benthic organisms in this tide pool for two years and evaluated their response to different disturbances: burial by sand, temperature anomalies, and tourism. We observed high temporal variability in the benthic cover in the Atalaia pool, with an important role of the sand being naturally brought by waves coming from the southeast to south-southeast direction on a seasonal basis. Despite the highly variable environment of this pool leading to periodic visual coral cover decrease when sand accumulated, coral health was not significantly affected neither by burial nor by the number of visitors accessing the pool. However, sea temperature anomaly played a role, being the main environmental driver affecting coral health, leading to increased paleness, bleaching and unhealthy purple/pink color and causing changes on the associated symbionts. Regardless of the local scope of this monitoring, this is an interesting study case to be used by other marine parks, explicitly showing how biological monitoring can aid the evaluation and potential adjustments to management strategies.

Keywords: benthic ecology; coral bleaching; nature conservation; abiotic factors, anthropogenic factors; zooxanthellae; Brazil; Pernambuco; Fernando de Noronha Archipelago

1. Introduction

Intertidal reef ecosystems have high ecological and economic importance, providing feeding, resting, spawning, and nursery areas for many marine organisms (Thompson *et al.* 2002; Andrades *et al.* 2018) and several ecosystem services such as food, economic and recreational activities for humans (Andrades *et al.* 2018). These habitats are subject to natural disturbances and anthropogenic impacts that influence their dynamics and therefore their conservation (Araújo *et al.* 2012). Natural disturbances vary in magnitude and scale, ranging from catastrophic events, such as hurricanes, to cyclical and seasonal fluctuations like tidal cycles and swells. The energy of waves generated from distant storms is transferred to the seafloor of intertidal ecosystems, modifying the habitats and directly battering the organisms (Denny & Gaylord 2002). The movement of water masses due to waves, currents and tides is especially important in shallow reefs, because it influences depth, temperature, luminosity, salinity and the sediment loads requiring specialized physiology and behaviour for survival in this environment (Denny & Gaylord 2002). Sediment dynamics can influence the structure of benthic communities in intertidal habitats affecting organisms by burial, abrasion, and increased turbidity, interacting with the effects of waves (Araújo *et al.* 2012, Figurski *et al.* 2016). These conditions can be especially harmful to corals, causing their mortality through sediment smothering and increasing competition with algae (Rogers 1990, Nugues & Roberts 2003).

Intertidal environments and shallow reefs are among the most impacted by humans worldwide (Halpern *et al.* 2008), with different levels of impacts depending on the activity (Addessi 1994). These impacts can happen in global scales (e.g. ocean warming and acidification) and in local scales (e.g. pollution, over-collection of living resources and disordered tourism; Halpern *et al.* 2008). As a consequence, alterations of trophic interactions (Valentine & Heck Jr. 2005), phase shifts (Norström *et al.* 2009), species replacement (Murray *et al.* 1999) mass bleaching events, and proliferation of coral diseases (Hughes *et al.* 2017, Hughes *et al.* 2018, Lough *et al.* 2018) can occur. The occurrence of environmental extremes is likely to increase (Thompson *et al.* 2002) and climate change is expected to interfere in the patterns of natural (Michener *et al.* 1997, Vaselli *et al.* 2008) and anthropogenic disturbances locally (Donovan *et al.* 2020, França *et al.* 2020).

The establishment of Marine Protected Areas (MPAs) is one of the main strategies in place to prevent anthropogenic impacts (Agardy 1994). When effectively managed, MPAs can significantly reduce local impacts through the creation of no-take areas, and tourism management among other administrative tools. However, the effectiveness of MPAs in protecting their biota from the impacts of climate change is limited, since the mitigation of the causes of global climate change goes beyond the capacity of MPAs (Edgar *et al.* 2014; Bruno *et al.* 2018). Besides that, even within protected areas, the ease of access makes intertidal habitats more prone to human impacts. For example, recreational divers can impact benthic organisms through trampling and direct contact possibly affecting community dynamics (Giglio *et al.* 2019). On the other hand, they are easier to regulate than open sea habitats (Thompson *et al.* 2002). In environments subjected to large natural fluctuations, such as intertidal ecosystems, adequate management planning must account for natural disturbances. Therefore, monitoring programs are essential to evaluate whether management practices in place are effective in mitigating impacts and are crucial to guarantee reef biota conservation and tourism sustainability (Thompson *et al.* 2002, Giglio *et al.* 2019).

Fernando de Noronha Marine National Park (FN) is an important MPA that protects marine habitats up to 50 m deep of an oceanic archipelago off North East Brazil. It is a no-take area, so the main potential impacts in intertidal and shallow environments there are related to tourism (Cristiano *et al.* 2020), pollution (Grillo & Mello 2020) and climate change (Ferreira *et al.* 2013). Long-term coral reef monitoring of FN showed a gradual and steady decrease of coral cover, mainly in shallow areas (Ferreira *et al.* 2020). FN received around 100,000 visitors per year on average during the period of this study. One of the chief tourist attractions in this Park is the Atalaia tidepool where snorkelers can dive under a series of regulations. Visitation to this tidepool requires reservation, and available spaces usually fill up quickly, so only a portion of the visitors can visit Atalaia.

Nevertheless, park managers often report coral bleaching and mortality events, in addition to a generalized perception that this reef has suffered many changes. We monitored corals, their endosymbionts and the cover of benthic organisms in this pool for two years (2017-2019) and evaluated their response to different disturbances: burial by sand, temperature anomalies, and tourism. If the current regulations are effective, we expected that changes in the reef and abnormalities in coral health would be more related to environmental variables than to the number of visitors in the pool.

2. Materials and Methods

2.1 Study area

Fernando de Noronha Marine National Park is a no-take area where the only permitted activities are tourism and scientific research. The climate is tropical, and the archipelago is under influence of the constant south-east and east trade winds (Matheus *et al.* 2019) and the South Equatorial Current. Wind and current exposure characterize two distinct regions in the archipelago. The windward coast faces southeast and is exposed to the currents and winds (Ambrosio *et al.* 2022); therefore, the sea is rough all year round. The leeward coast faces northwest and is sheltered from the currents and winds, and from December to February it is subject to swells from the north. The study was conducted at Atalaia pool (-3.85° S, -32.41° W), located in the windward coast of FN island (Fig. 1). The pool is a biogenic vermetid reef, composed of a solid limestone formed by skeletal accretions mottled with fine calcarenite and bottomed in the volcanic boulder foundation (Jindrich 1983, Laborel-Deguen *et al.* 2019).

Snorkelling is permitted in the pool at low tide during park opening hours (8am-6pm). The number of visitors allowed is defined by the park managers according to the time of the tide and the amount of sand at the bottom of the pool. On days with ideal conditions (i.e. tides lower than 90cm within the park opening hours, no swell and low sand cover on the bottom), visitors are divided into 6 groups of 16 people and each group is allowed to stay for 30 minutes in the tidepool. Therefore, the maximum daily number of visitors per day is 96 people and cannot be exceeded. The tidepool forms when the tide is below 90cm, and the depth range of the snorkelling sites is between 50 cm and 90 cm. When sand cover is high, the pool area deep enough for snorkelling is reduced, so the number and size of groups may be reduced. Float vests, mask and snorkel are mandatory, the use of fins and sunscreen lotion is not allowed, while cameras are allowed, and camera sticks length is limited to 20 cm. In order to schedule the visit, visitors must watch an informative video about the tidepool and visitation rules. Upon arrival at the tidepool, visitors attend a 5-minute lecture given by a park monitor on minimum impact flotation techniques and the importance and fragility of the organisms that live in the pool. Standing, kneeling and leaning on the bottom in any way is not permitted. During the dive, the monitor supervises the visitors, alerting them if there is inappropriate behaviour and interfere whenever necessary.

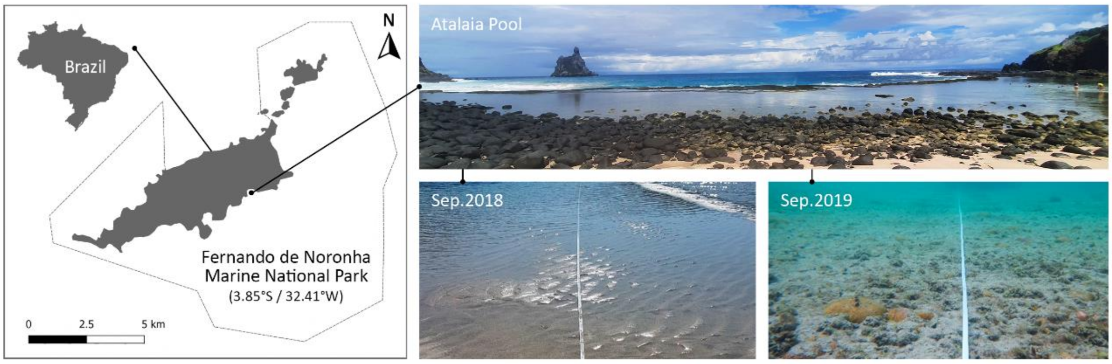


Figure 1. Fernando de Noronha Archipelago and location of the study site. Dotted lines indicate the boundaries of the National Park (left panel). General view of Atalaia pool (right panel above), and view from the same transect in September 2018 and September 2019 (right panel below).

2.2 Sampling design

Field sampling was conducted monthly between November 2017 and November 2019 (except for May and July 2018). Four 20 m transects were established in the pool, using aluminium nails to mark their starting and ending points. Benthic cover of major groups – turf, crustose coralline algae, macroalgae, cyanobacteria, corals, other invertebrates, sand, rock – was estimated visually in situ using 25 X 25 cm quadrats, placed every 2 meters along the transects.

When *Siderastrea stellata* Verrill 1868 coral colonies were present, their color was classified as normal or with color alterations: white bleached, pale, and pink/purple colonies. The percentage of coral tissue of each color was estimated.

Additionally, we marked 20 colonies of *S. stellata* to estimate the duration of burial events. The marking was conducted in April 2018, when sand cover was relatively low in the pool and the marks consisted of aluminium nails fixed to the bedrock near the colonies with plastic tags. The colonies were evenly distributed in the tidepool and were monitored until June 2019 to verify if they were uncovered or buried. The search for colonies was done with the help of a location sketch and putting away macroalgae that could be covering them. The colonies were classified as buried only when they were located again in subsequent samplings. When it was not possible to determine if the colony was buried, dead, or if the mark was lost, the colonies' status was classified as unknown.

2.3 Coral endosymbiont analyses

We collected a small tissue sample from 29 colonies of *S. stellata* in July (n = 3) and October (n = 12) 2018, and January (n = 4) and November (n = 10) 2019, to investigate the temporal changes of the dominant endosymbiont community associated with *S. stellata* in the Atalaia pool. Tissue samples were fixed in CHAOS saline solution buffer (4 M guanidine thiocyanate, 0.1% N-lauroylsarcosine sodium, 10 mM Tris pH 8.0, 0.1 M 2-mercaptoethanol) to start tissue lysis and DNA extraction. Total DNA was extracted with phenol:chloroform protocol (Fukami *et al.* 2004) and the endosymbiotic Symbiodiniaceae were identified as ITS2 (internal transcribed spacer 2 region) rDNA phylotypes (LaJeunesse 2001). The ITS2 region was amplified by PCR (Polymerase Chain Reaction), using the primers SYM_VAR_5.8S2 (sense, 5'-GAATTGCAGAACTCCGTGAACC-3') and SYM_VAR_REV (antisense, 5’-CGGGTTCWCTTGTYTGACTTCATGC-3’), and amplification conditions were: 2 min at 98 °C; 35 cycles of 10 s at 98 °C, 45 s at 56 °C, 45 s at 72 °C, and a final extension of 5 min at 72 °C (Hume *et al.* 2018). Amplification products were purified with ExoSAP-IT enzymes (UBS Corporation, Cleveland, OH) following the manufacturer's instructions and sequenced by the Sanger method (Sanger *et al.* 1977; sequencer ABI 3500, Applied Biosystems). Sequences were visualized and edited in GeneStudio software (GeneStudio, Inc) to remove low-quality regions. The sequencing dataset was deposited in the NCBI Sequence Read Archive database (under the accession numbers OQ398681-OQ398706).

Symbiodiniaceae genera were identified by heuristic searches, using BLAST (Altschul *et al.* 1990) in the GenBank nucleotide database (http://blast.ncbi.nlm.nih.gov/Blast.cgi). Phylogenetic analyses for each genus were conducted to identify dominant Symbiodiniaceae ITS2 phylotypes with ITS2 rDNA sequences available in GenBank, without outgroups. Sequence alignments were performed in MAFFT 7 software (Katoh & Standley 2013) with FFT-NS-2 strategy. The jModelTest 2 software (Darriba *et al.* 2018) selected the best-fit models of nucleotide substitution, according to the Akaike Information Criterion (AIC). The best-fitting models for the *Breviolum* spp. and the *Cladocopium* spp. datasets were HKY+G and GTR, respectively. Phylogenetic analyses were conducted by the Bayesian Inference reconstruction method (two independent runs of four Markov chains for a million generations, 10% posterior burn-in) using MrBayes software (Huelsenbeck & Ronquist 2001). Gaps were considered as the fifth state. The best trees obtained from the two methods were compared, rooted at the midpoint, and edited with FigTree 1.4.2 software (Rambaut 2009).

2.4 Environmental and tourism data

In order to describe environmental conditions and the level of touristic use of Atalaia pool during the study period, which could help explaining the changes in benthic cover and coral health, we obtained data on wave direction, water temperature and number of visitors per month. Wave direction for Fernando de Noronha Archipelago was obtained from WaveWatch III Global Wave Model (https://coastwatch.pfeg.noaa.gov/erddap/index.html). The dimension Tdir, that represents peak wave direction for the total wave conditions was used in the analysis.

Sea surface temperature (SST) data was obtained from Fernando de Noronha virtual station in Coral Reef Watch (https://coralreefwatch.noaa.gov/product/vs/gauges/fernando_de_noronha.php). Daily maximum and minimum SST were averaged by month to obtain the mean monthly values used in the analysis. Sea Surface Temperature (SST) anomaly, that represents the difference between the observed daily SST and the long-term average was also averaged by month. The daily number of visitors in the pool was provided by Econoronha (https://grupocataratas.com/econoronha/), which is the company that schedules tourist access to the location.

2.5 Data analysis

As we were not interested in the variation of benthic cover inside the Atalaia pool, we grouped the quadrats within the respective transect and used the four transects as replicates to evaluate benthic cover variation across time. After a square-root transformation, we used the percentage of the different benthic categories to build a resemblance matrix for the sampling events using Bray-Curtis distance, and then tested the variation across time using a PERMANOVA test with 999 unrestricted permutations of raw data (Anderson 2001). In case the effect of time was significant, we performed a post-hoc test and then a SIMPER analysis for the significant pair-wise comparisons to obtain the most important benthic categories for the observed temporal differences (Clark 1993). This analysis was performed using Primer 6 (Clark & Warwick 2001).

To examine the influence of wave direction on the amount of sand covering the pool, we performed a circular-linear correlation among mean wave directions and mean sand cover in the subsequent month. The mean wave direction for each sampling month was calculated after applying a Rayleigh Test to wave direction data to discard uniformity. Circular statistics were calculated using Oriana v. 4 (Kovach 2011).

We tested the influence of sand cover on coral cover using a linear regression (sand cover independent, coral cover dependent variable). To understand the effects of predictors (SST, SST anomaly, sand cover and number of visitors) on coral health we performed a multiple regression. All color anomalies were grouped under the classification "unhealthy", which was used as the response variable in this analysis. The regressions were performed using R (R Core Team 2022). The prevalence of coral disease was too low to enable any further exploration.

3. Results

The live cover in the Atalaia tidepool was mostly represented by turf algae (mean percentage and standard deviation 30.22 ±14.36), followed by macroalgae (10.18 ± 4.92), the coral *Siderastrea stellata* (6.46 ± 1.46), and crustose calcareous algae (3.46 ± 2.54) which varied throughout the entire period of the study (Fig. 2). We observed strong seasonality on the benthic cover (PERMANOVA: Pseudo-F22,69 = 2.59; p = 0.001), with great variation in the sand cover (Fig. 3), that ranged from 6.3% to 84.6% (mean and standard deviation 46.93 ± 19.67%). Sand cover causing burial events were influenced by wave direction (R2 = 0.62; p = 0.001), with sand being removed when waves came from North to East-southeast direction and deposited when waves came from Southeast to South-southeast direction (Fig. 3).

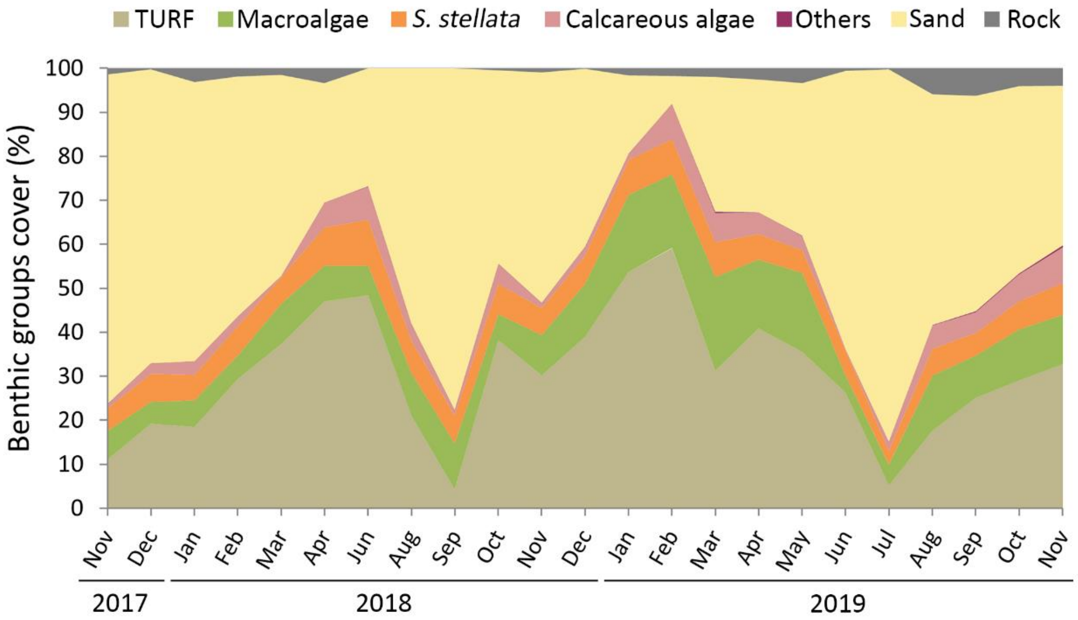


Figure 2. Benthic cover on Atalaia pool along the study period. Months are indicated by their initial three letters (*e.g.* Jan = January).

The natural accumulation of sand in the tidepool temporarily reduced coral cover ($R^2 = 0.45$; $p < 0.001$; Fig. 4). For instance, about 70% of the monitored colonies of *S. stellata* were buried in at least one of the monitoring events and 55% were buried in two or more monitoring events (Fig. 5). Some colonies (#12 to #20) were buried in the sampling conducted in August and September 2018, suggesting that they may have remained buried for at least 33 days, while colonies #19 and #20 remained buried in the sampling conducted in October 2018, indicated the burial event for these corals could have lasted up to 64 days. Colony #4 was also buried in three subsequent samplings, from January to March 2019 (80 days), as did colony #14, buried from December 2018 to February 2019 (81 days). These burial events coincide with the period when the sand cover was higher in the tidepool (Fig. 5).

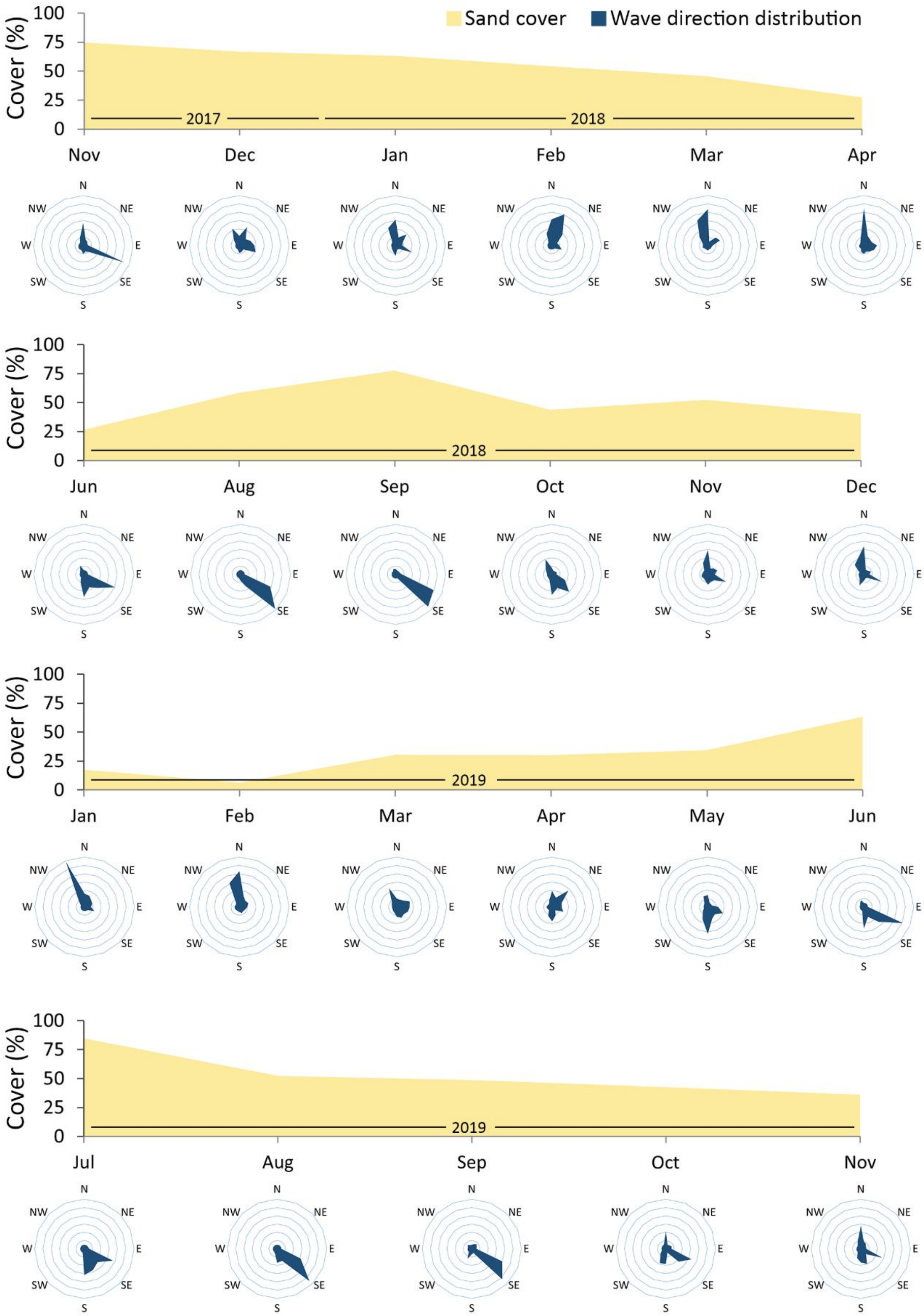


Figure 3. Sand cover (%) and wave direction along the study period. Months are abbreviated using the three first letters (*e.g.* Jan = January). N= north; NE = northeast; E = east; SE = southeast; SW = southwest; W = west; NW = northwest.

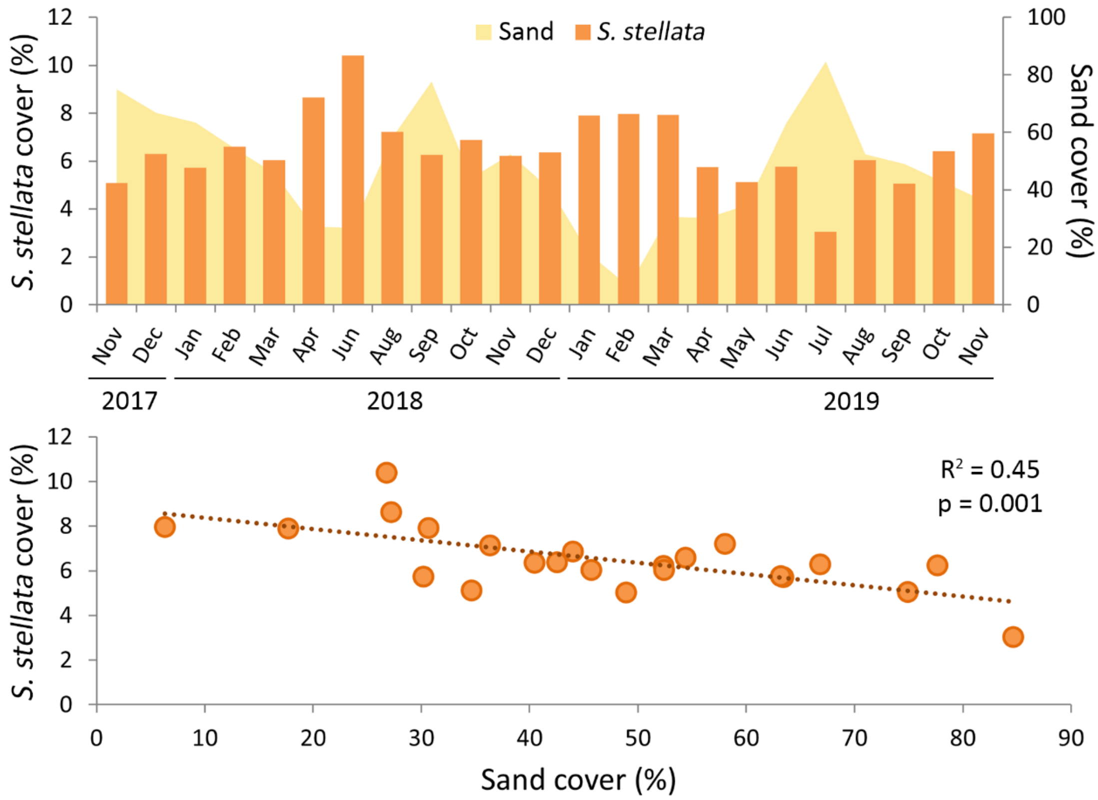


Figure 4. (a) *Siderastrea stellata* and sand cover (%) along the study period. (b) Relationship between sand cover and *Siderastrea stellata* cover. Coefficient of regression (R^2) is indicated in the graph.

Despite the burial events, thermal anomalies were the most important driver of coral health throughout the monitoring period (Multiple regression health~anomaly+burial+visitors; Fig. 6). During thermal anomalies, corals got pale two times more often than they got completely bleached (16% and 8% of all samples respectively), and rarely assumed pink or purple shades (~4%; Fig. 6). The number of visitors did not affect coral health and had minimum variation throughout the entire monitoring period (min=990, max=1678, average=1420.13 visitors/month). Monthly number of visitors during the study period is shown in the supplementary material (Table S1).

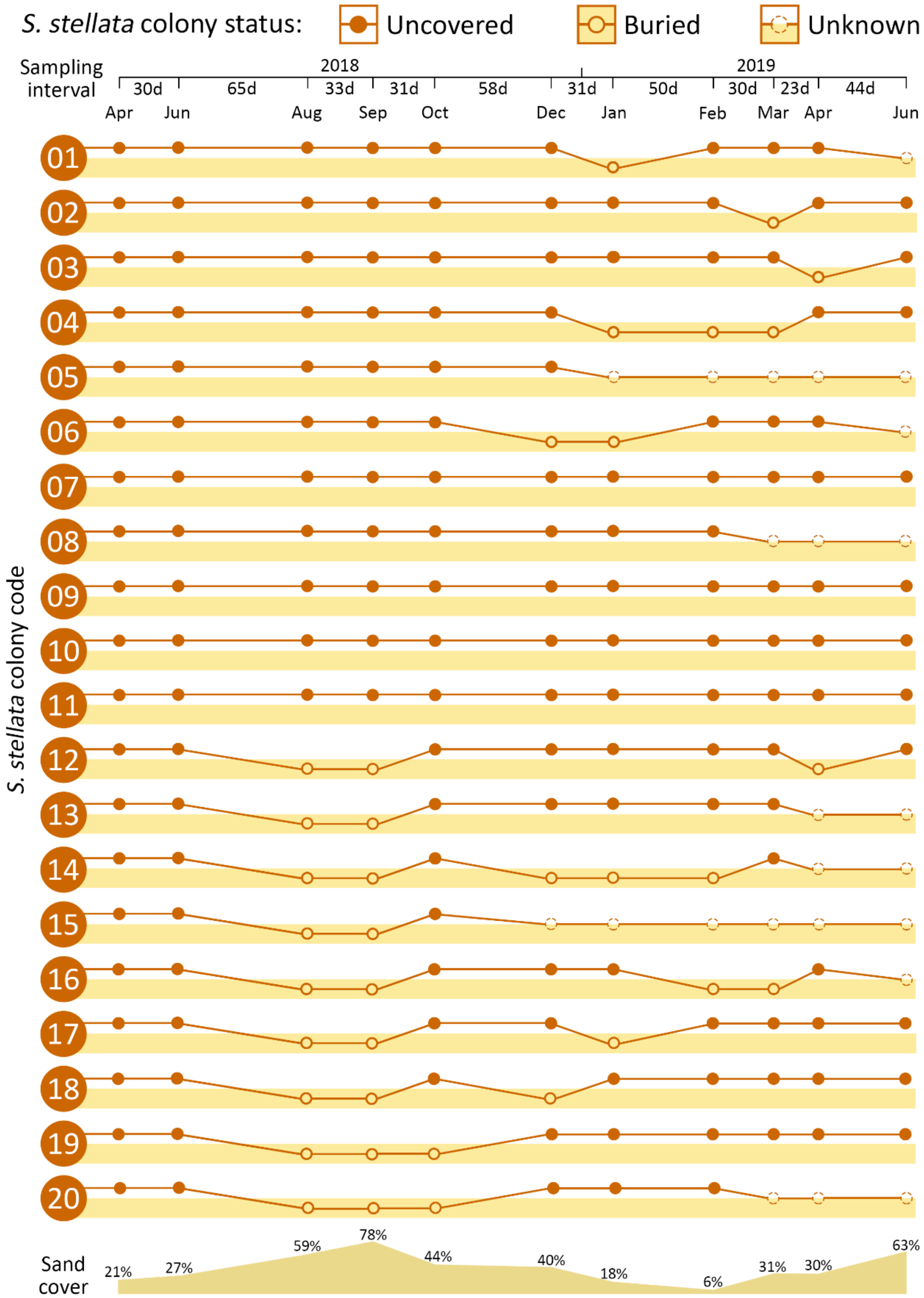


Figure 5. Status of the monitored *Siderastrea stellata* colonies from April 2018 to June 2019, medium sand cover (%), and interval between samplings (days).

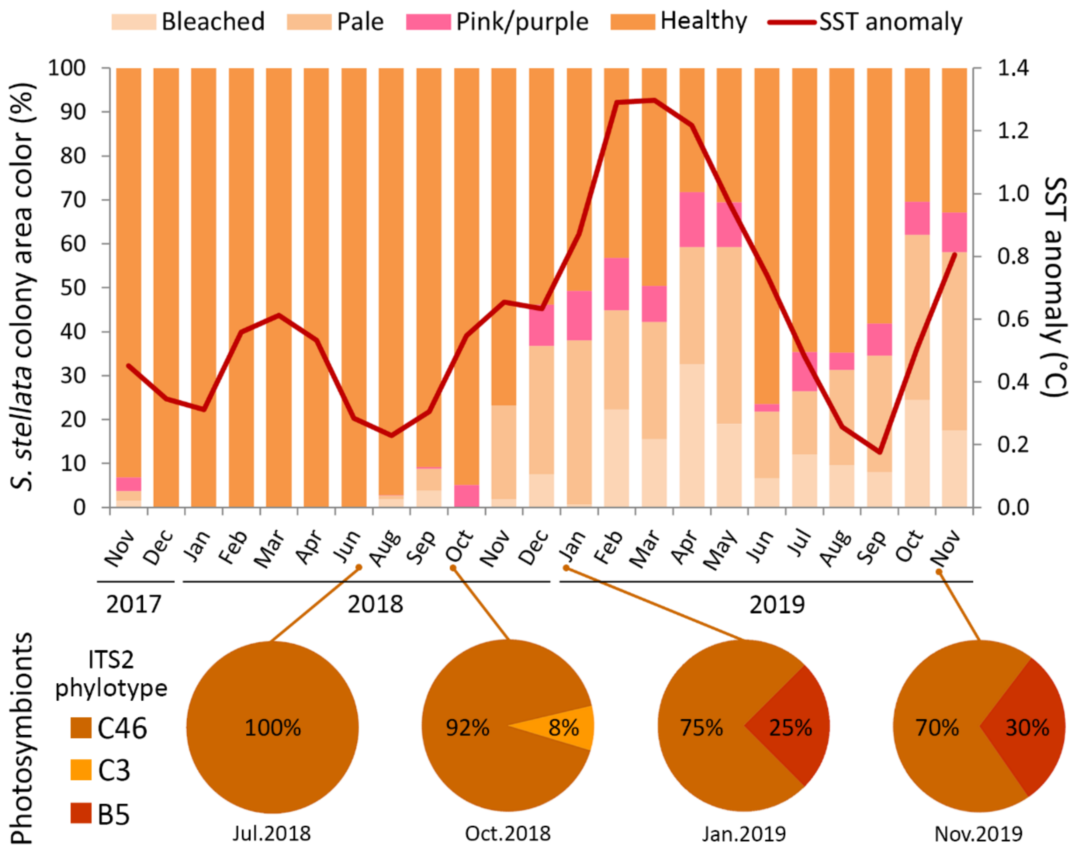


Figure 6. Coral health, SST anomaly along the study period, and proportion of endosymbiont phylotypes within *Siderastrea stellata*.

The endosymbiont community of *S. stellata* in the tidepool was dominated by the phylotype *Cladocopium* C46 (70-100% of sampled colonies) throughout the monitored period, with one record of *Cladocopium thermophilum* (B.Hume, D'Angelo, E.G.Smith, J.R.Stevens, J.Burt & Wiedenmann, 2018) (phylotype C3). During the period of thermal anomalies, the phylotype Breviolum B5, which was not found before, emerged and became dominant in 25-30% of the corals (Fig. 6; phylogenetic reconstructions are in Supplementary Material S1).

4. Discussion

We observed high temporal variability in the benthic cover in the Atalaia pool, with an important role of the sand being naturally brought by waves coming from the southeast to south-southeast direction on a seasonal basis. Despite the highly variable environment of this pool leading to periodic visual coral cover decrease when sand accumulated, coral health was not significantly affected neither by burial nor by the number of visitors accessing the pool. However, sea temperature anomaly played a role, being the main environmental driver affecting coral health, leading to increased paleness, bleaching and unhealthy purple/pink color and causing changes on the associated symbionts. This is an interesting study case that demonstrates the importance of understanding natural fluctuations to guide proper management of intertidal ecosystems.

Seasonal burial and exhumation of intertidal reefs have been related to seabed morphology and wave forces (Storlazzi *et al.* 2011) and highly variable tidepools have also been reported in different locations (Bertocci *et al.* 2012). Wave seasonality and consequent variation in the amount of sand influences the dynamics of all benthic organisms in the pool, potentially causing a temporary decrease in live cover due to burial, with subsequent recovery when sand is withdrawn by waves coming from the north to east-southeast direction. In Atalaia pool, macroalgae, turf, corals and CCA decreased in cover when sand cover increases but recovered when sand was withdrawn from the pool. Algae are usually intolerant to sedimentation because it limits nutrient uptake and hinders photosynthesis (Airoldi 2003). However, some species such as coralline algae are able to deal with sediment abrasion and anoxia, enabling them to survive seasonal burial (Figurski *et al.* 2016). On the other hand, sponges, ascidians and other filtering or suspension feeding organisms are rare or absent in the pool, despite being common in other reefs of the archipelago not subject to burial (Matheus *et al.* 2019), possibly due to anoxia and clogging of feeding structures when buried (Figurski *et al.* 2016).

We observed that burial events could last more than 80 days and reached most of colonies in the tidepool. When sand was removed, coral cover returned to previous values, demonstrating that there was no significant coral mortality or effects on coral health due to burial. Laboratory experiments have shown that most resistant and resilient coral species can survive up to 15 days of complete burial (Rice and Hunter 1992) and *Siderastrea radians* colonies can recover even when damage is extensive, growing new tissue 2–3 weeks after ambient conditions were restored and fully recovering after 6 weeks (Lirman *et al.* 2002). A field experiment demonstrated that *Siderastrea stellata* exhibits great recovery capacity when damaged by sediment burial, showing signs of recovery as fast as 2 days after being completely bleached due to a burial event (Longo *et al.* 2020). The high survival of the colonies after very long-lasting burial events can be related to the high hydrodynamics in the tidepool, promoting water circulation to the colonies even when buried. It is also possible that sediment movements briefly exposed parts of these colonies in the interval between samplings, and the region free of sediment was able to mitigate for the loss of productivity due to limited light uptake (Tunala *et al.* 2019).

When different stressors were tested to explain coral health, only temperature anomaly had an effect, increasing the pale, bleached and/or pink/purple colony area, but corals were recovered on subsequent monitoring events after the anomaly ceased. This indicates the high resilience of *S. stellata* that can thrive under stressful conditions that do not allow the survival of other species (Longo *et al.* 2020), alike its congener *S. radians* in the Caribbean (Lirman *et al.* 2002). Although some colonies got completely bleached, color alterations (pale, pink and purple colonies) were more frequent than white bleaching, indicating that *S. stellata* seems to be resistant too, with thermal stress not being sufficient to result in the loss of symbionts for several colonies. Paleness and pink pigmentation are considered early states of bleaching, representing signals of debilitated coral health (Sassi *et al.* 2021). Colorful bleaching possibly provides a photobiological protective system with fluorescent pigments that regulate the amount of light that enter the coral tissues, and is believed to occur when corals are exposed to brief or mild heat (Bollati *et al.* 2020).

Shifts on the symbiont community may help corals to resist or recover after extreme temperature events, by increasing the proportion of thermal tolerant symbionts (Baker 2003, Oliver & Palumbi 2011). After bleaching, corals can recover their endosymbiotic algae communities (Symbiodiniaceae sensu LaJeunesse *et al.* 2018) mostly through the remaining endosymbionts (Jones & Yellowlees 1997) that can cause shifts in the community composition towards more stress-tolerant lineages (Bay *et al.* 2016). We documented intense thermal anomalies during our study, particularly from October 2018 to March 2019 and in October-November 2019 (maximum of 1.30°C in March 2019, Fig. 6), when only 30-60% of the corals remained healthy. Our data shows that while the photosymbiont assemblage is normally dominated by Cladocopium C46, during thermal anomalies the proportion of *S. Stellata* colonies dominated by Breviolum B5 increases.

Physiological traits are still poorly understood for these lineages. Although B5 and C3 are suggested as phylotypes with intermediate thermotolerance, pairwise comparisons have suggested that B5 is more thermotolerant than C3 (LaJeunesse *et al.* 2009, Swain *et al.* 2017). Breviolum B5 was already identified in association with *S. radians* in well-lit habitats in Caribbean reefs (Monteiro *et al.* 2013) and its presence may play a role in coral adaptation to high temperatures and irradiances (LaJeunesse 2002). Cladocopium C46 is a specific symbiont of *Siderastrea* spp. common along the Atlantic (Monteiro *et al.* 2013) but little is known about its thermotolerance. We cannot affirm if the shift occurs by B5 internal proliferation or new acquirement from the water column (switch or shuffling; Jones *et al.* 2008), however changing the photosymbiont assemblage may be linked to the success of *S. stellata* in enduring sea temperature anomaly events in the Atalaia pool. Further long-term investigations are needed to clarify whether there is a natural seasonality in the fluctuation of symbiont composition and whether the observed changes are lasting or temporary.

Coral health was not affected by the number of visitors in the tidepool, indicating that the tourism management measures in place (limiting the number of visitors, periodically assessing the conditions of the pool, educational videos, pre-dive briefing and supervision of visitors) are being effective to avoid major damage to the benthic organisms, considering current environmental conditions. Tourism is an important source of impacts on coral reefs, but when properly managed and supervised, the negative effects can be substantially reduced. Development of carrying capacity studies and consequent limitation of the number of visitors (Luiz 2009, Naranjo-Arriola 2021), use of mooring buoys instead of anchors (Forrester 2020, Shokri & Mohammadi 2021), educating tourists about the effects of pollutants including sunscreen (Levine 2020), training guides and divers on minimal impact diving (Giglio *et al.* 2018, 2022) are some examples of management measures with positive effects.

It is important to integrate knowledge about how natural stressors influence benthic communities, their drivers and resilience when planning tourism and evaluating its impacts in coastal ecosystems. Differentiating natural dynamics from consequences of human impacts is critical to evaluate and adjust management strategies, notably in the context of increasing human and climate derived stressors in these environments (Figurski *et al.* 2016). The negative effects of burial, for instance, could be exacerbated by higher temperatures (Tunala *et al.* 2019) decreasing the resistance and resilience of corals. Documenting such natural variation in a highly visited tide pool is critical to understand how local impacts interact and how they may be modulated by global changes (França *et al.* 2020). Our study demonstrates that implementing long-term biological monitoring programs in marine parks is critical to separate natural and impact-driven community dynamic in ecosystems prompting informed management strategies.

Acknowledgements

This work was also supported by Serrapilheira Institute (Grant No. Serra-1708-15364) awarded to GOL, the National Council for Scientific and Technological Development (CNPq; Grant 443329/2019-2, awarded to GOL) and the research program PELD/ILOC ("Pesquisa Ecológica de Longa Duração das Comunidades Recifais das Ilhas Oceânicas"; financed by CNPq through the Grant 441241/2016-6 awarded to Carlos E. L. Ferreira). GOL is also grateful to a research productivity scholarship provided by CNPq (Grant 310517/2019-2). This study was part of the research and monitoring program of ICMBio Noronha, with the support of ICMBio Noronha Volunteer Program. We are grateful to Francielly Furlani, Daniel Rovira, Rebeca Assis, Jessika Santana, Luísa Fagundes and Ricardo Araújo for their help with data collection. We thank Vanessa G. Staggemeier for the help with circular statistics. This study was conducted under the research permit SISBIO # 41327-12.

References

Addessi, L. (1994). Human Disturbance and Long-Term Changes on a Rocky Intertidal Community. *Ecological Applications*, *4*(4):786–797. https://doi.org/10.2307/1942008

Agardy, M. T. (1994). Advances in marine conservation: the role of marine protected areas. *Trends in Ecology and Evolution*,*9*(7): 267–270. https://doi.org/10.1016/0169-5347(94)90297-6

Airoldi, L. (2003). the Effects of Sedimentation on Rocky Coast Assemblages. *Oceanography and Marine Biology, An Annual Review, Volume 41*:161–236. https://doi.org/10.1201/9780203180570-23

Altschul, S.F., Gish, W., Miller, W., Myers, E.W., Lipman, D.J. (1990). Basic Local Alignment Search Tool. *Journal of Molecular Biology*. 403–410.

Ambrosio, B. G., Takase, L. S., Stein, L. P., Costa, M. B., Siegle, E. (2022). Wave-induced sediment and rhodolith mobility on a narrow insular shelf dominated by wave variability (Fernando de Noronha Archipelago, Brazil). *Continental Shelf Research*, *235*(November 2021): 104662. https://doi.org/10.1016/j.csr.2022.104662

Anderson, M. J. (2001). A new method for non-parametric multivariate analysis of variance. *Austral Ecology*, *26*(1), 32–46. https://doi.org/10.1046/j.1442-9993.2001.01070.x

Andrades, R., Machado, F. S., Reis-Filho, J. A., Macieira, R. M., Giarrizzo, T. (2018). Intertidal biogeographic subprovinces: Local and regional factors shaping fish assemblages. *Frontiers in Marine Science*, *9*(NOV): 1–14. https://doi.org/10.3389/fmars.2018.00412

Araujo, R., Arenas, F., Åberg, P., Sousa-Pinto, I., Serrão, E. A. (2012). The role of disturbance in differential regulation of co-occurring brown algae species: Interactive effects of sediment deposition, abrasion and grazing on algae recruits. *Journal of Experimental Marine Biology and Ecology*, *422–423*: 1–8. https://doi.org/10.1016/j.jembe.2012.04.006

Baker, A.C. (2003). Flexibility and specificity in coral-algal symbiosis: Diversity, ecology, and biogeography of Symbiodinium. *Annual Review of Ecology, Evolution, and Systematics 34:*661–689. https://doi.org/10.1146/annurev.ecolsys.34.011802.132417

Bay, L.K., Doyle, J., Logan, M., Berkelmans, R. (2016) Recovery from bleaching is mediated by threshold densities of background thermo-tolerant symbiont types in a reef-building coral. *Royal Society Open Science 3:*160322

Bertocci, I., Araújo, R., Incera, M., Arenas, F., Pereira, R., Abreu, H., … Sousa-Pinto, I. (2012). Benthic assemblages of rock pools in northern Portugal: seasonal and between-pool variability. *Scientia Marina 76*(4): 781–789. https://doi.org/10.3989/scimar.03669.21A

Bollati, E., D'Angelo, C., Alderdice, R., Pratchett, M., Ziegler, M., Wiedenmann, J. (2020). Optical Feedback Loop Involving Dinoflagellate Symbiont and Scleractinian Host Drives Colorful Coral Bleaching. *Current Biology*, *30*(13), 2433-2445.e3. https://doi.org/10.1016/j.cub.2020.04.055

Bruno, J. F., Bates, A. E., Cacciapaglia, C., Pike, E. P., Amstrup, S. C., Van Hooidonk, R., … Aronson, R. B. (2018). Climate change threatens the world's marine protected areas. *Nature Climate Change*, *8*(6), 499–503. https://doi.org/10.1038/s41558-018-0149-2

Clarke, K. R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology*, *18*(1), 117–143. https://doi.org/10.1111/j.1442-9993.1993.tb00438.x

Clarke, K.R., Warwick, R.M. (2001) Change in marine communities: an approach to statistical analysis and interpretation. PRIMER-E Ltd., Plymouth

Cristiano, C., Camboim, G., Carla, L. (2020). Beach landscape management as a sustainable tourism resource in Fernando de Noronha Island (Brazil). *Marine Pollution Bulletin 150*(September 2019):110621. https://doi.org/10.1016/j.marpolbul.2019.110621

Darriba, D., Flouri, T., Stamatakis, A. (2018) The state of software for Evolutionary Biology. *Molecular Biology and Evolution 35*:1037–1046.

Denny, M., Gaylord, B. (2002). The mechanics of wave-swept algae. *Journal of Experimental Biology 205*(10):1355–1362. https://doi.org/10.1242/jeb.205.10.1355

Donovan, M. K., Adam, T. C., Shantz, A. A., Speare, K. E., Munsterman, K. S., Rice, M. M., … Burkepile, D. E. (2020). Nitrogen pollution interacts with heat stress to increase coral bleaching across the seascape. *Proceedings of the National Academy of Sciences of the United States of America*, *117*(10): 5351–5357. https://doi.org/10.1073/pnas.1915395117

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., … Thomson, R. J. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature*, *506*(7487): 216–220. https://doi.org/10.1038/nature13022

Ferreira, B. P., Costa, M. B. S. F., Coxey, M. S., Gaspar, A. L. B., Veleda, D., Araujo, M. (2013). The effects of sea surface temperature anomalies on oceanic coral reef systems in the southwestern tropical Atlantic. *Coral Reefs*, *32*(2):441–454. https://doi.org/10.1007/s00338-012-0992-y

Ferreira, B. P., Coxey, M. S., Gaspar, A. L., Silveira, C. B. L. da, Souza, F. N. R. da, Matheus, Z., … Messias, L. T. (2020). Status and trends of coral reefs of the Brazil region. In D. Souter, S. Planes, J. Wicquart, M. Loga, D. Obura, F. Staub (Eds.), *Status of Coral Reefs of the World: 2020* (pp. 1–13). Retrieved from https://gcrmn.net/2020-report/

Figurski, J. D., Freiwald, J., Lonhart, S. I., Storlazzi, C. D. (2016). Seasonal sediment dynamics shape temperate bedrock reef communities. *Marine Ecology Progress Series 552*: 19–29. https://doi.org/10.3354/meps11763

Forrester, G. E. (2020). The influence of boat moorings on anchoring and potential anchor damage to coral reefs. *Ocean & Coastal Management*, *198:*105354. https://doi.org/10.1016/j.ocecoaman.2020.105354

França, F. M., Benkwitt, C. E., Peralta, G., Robinson, J. P. W., Graham, N. A. J., Tylianakis, J. M., … Barlow, J. (2020). Climatic and local stressor interactions threaten tropical forests and coral reefs. *Philosophical Transactions of the Royal Society B: Biological Sciences 375*. https://doi.org/10.1098/rstb.2019.0116

Fukami, H., Budd, A.F., Levitan, D.R., Jara, J., Kersanach, R., Knowlton, N. (2004) Geographic differences in species boundaries among members of the *Montastraea annularis* complex based on molecular and morphological markers. *Evolution (NY) 58:*324–337.

Giglio, V. J., Luiz, O. J., Chadwick, N. E., Ferreira, C. E. L (2018) Using an educational video-briefing to mitigate the ecological impacts of scuba diving. Journal of Sustainable Tourism, 26:5, 782-797, DOI: 10.1080/09669582.2017.1408636

Giglio, V. J., Luiz, O. J., Ferreira, C. E. L. (2019). Ecological impacts and management strategies for recreational diving: A review. *Journal of Environmental Management*, *256*(December 2019), 109949. https://doi.org/10.1016/j.jenvman.2019.109949

Giglio, V. J., Marconi, M., Pereira-Filho, G. H., Leite, K. L., Figueroa, A. C., Motta, F. S. (2022). Scuba divers' behavior and satisfaction in a new marine protected area: Lessons from the implementation of a best practices program. *Ocean & Coastal Management 220*:106091. https://doi.org/10.1016/j.ocecoaman.2022.106091

Grillo, A. C., Mello, T. J. (2021). Marine debris in the Fernando de Noronha Archipelago, a remote oceanic marine protected area in tropical SW Atlantic, *Marine Pollution Bulletin 164*(October 2020). https://doi.org/10.1016/j.marpolbul.2021.112021

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., … Watson, R. (2008). A Global Map of Human Impact on Marine Ecosystems. *Science*, *319*(5865): 948–952. https://doi.org/10.1126/science.1149345

Huelsenbeck, J.P., Ronquist, F. (2001) MrBayes: Bayesian inference of phylogenetic trees. *Bioinformatics 17*:754–755.

Hughes, T. P., Kerry, J. T., Álvarez-Noriega, M., Álvarez-Romero, J. G., Anderson, K. D., Baird, A. H., … Wilson, S. K. (2017). Global warming and recurrent mass bleaching of corals. *Nature*, *543*(7645), 373–377. https://doi.org/10.1038/nature21707

Hughes, T. P., Anderson, K. D., Connolly, S. R., Heron, S. F., Kerry, J. T., Lough, J. M., … Wilson, S. K. (2018). Spatial and temporal patterns of mass bleaching of corals in the Anthropocene. *Science*, *359*(6371), 80–83. https://doi.org/10.1126/science.aan8048

Hume, B.C.C., Ziegler, M., Poulain, J., Pochon, X., Romac, S., Boissin, E., de Vargas, C., Planes, S., Wincker, P., Voolstra, C.R. (2018) An improved primer set and amplification protocol with increased specificity and sensitivity targeting the *Symbiodinium* ITS2 region. *PeerJ 6*:e4816; DOI 10.7717/peerj.4816.

ICMBio 2021. Painel dinâmico de informações. Accessed in 02/03/2023. http://qv.icmbio.gov.br/QvAJAXZfc/opendoc2.htm?document=painel_corporativo_6476.qvw&host=Local&anonymous=true

Jindrich, V. (1983). Structure and diagenesis of recent algal- foraminifer reefs, Fernando de Noronha, Brazil. *Journal of Sedimentary Petrology*, *53*(2):449–459. https://doi.org/10.1306/212F8206-2B24-11D7-8648000102C1865D

Jones, A., Berkelmans, R., van Oppen, M., Mieog, J., and Sinclair, W. (2008) A community change in the algal endosymbionts of a scleractinian coral following a natural bleaching event: field evidence of acclimatization. *Proceedings of the Royal Society B: Biological Sciences 275:*1359–1365.

Jones, R. J., Yellowlees, D. (1997). Regulation and control of intracellular algae (= zooxanthellae) in hard corals Regulation and control of intracellular algae (l zooxanthellae) in hard corals. *Philosophical Transactions of the Royal Society B: Biological Sciences 352*:457–468. https://doi.org/10.1098/rstb.1997.0033

Katoh, K., Standley, D.M. (2013) MAFFT multiple sequence alignment software version 7: Improvements in performance and usability. *Molecular Biology and Evolution* 30: 772–780.

Kovach, W. L. (2013). Oriana for Windows, version 4.02. Kovach Computing Services URL https://www.kovcomp.co.uk/oriana/index.html.

Laborel-Deguen, F., Castro, C. B., Nunes, F. L. D., Pires, D. O. (2019). *Recifes brasileiros: o legado de Laborel*. Rio de Janeiro: Museu Nacional. Retrieved from https://coralvivo.org.br/o-que-fazemos/

LaJeunesse, T.C. (2001) Investigating the biodiversity, ecology, and phylogeny of endosymbiotic dinoflagellates in the genus *Symbiodinium* using the ITS region: in search of a "species" level marker. *Journal of Phycology 37:*866–880.

LaJeunesse, .T. (2002) Diversity and community structure of symbiotic dinoflagellates from Caribbean coral reefs. *Marine Biology 141***:** 387–400. https://doi.org/10.1007/s00227-002-0829-2

LaJeunesse, T.C., Smith, R.T., Finney, J. Oxenford, H. (2009) Outbreak and persistence of opportunistic symbiotic dinoflagellates during the 2005 Caribbean mass coral 'bleaching' event. *Proceedings of the Royal Society B-Biological Sciences 276:* 4139– 4148.

LaJeunesse, T.C., Parkinson, J.E., Gabrielson, P.W., Jeong, H.J., Reimer, J.D., Voolstra, C.R., Santos, S.R. (2018). Systematic revision of Symbiodiniaceae highlights the antiquity and diversity of coral endosymbionts. *Current Biology 28:*2570-2580.e6. https://doi.org/10.1016/j.cub.2018.07.008

Levine, A. (2020). Sunscreen use and awareness of chemical toxicity among beach goers in Hawaii prior to a ban on the sale of sunscreens containing ingredients found to be toxic to coral reef ecosystems. *Marine Policy 117:*103875. https://doi.org/10.1016/j.marpol.2020.103875

Lirman, D., Manzello, D., Maciá, S. (2002). Back from the dead: The resilience of *Siderastrea radians* to severe stress. *Coral Reefs 21*(3): 291–292. https://doi.org/10.1007/s00338-002-0244-7

Longo, G. O., Correia, L. F. C., Mello, T. J. (2020). Coral recovery after a burial event: insights on coral resilience in a marginal reef. *Marine Biodiversity 50*(6). https://doi.org/10.1007/s12526-020-01121-4

Lough, J. M., Anderson, K. D., Hughes, T. P. (2018). Increasing thermal stress for tropical coral reefs: 1871-2017. *Scientific Reports, 8*(1):1–8. https://doi.org/10.1038/s41598-018-24530-9

Luiz, O.J. 2009. Estudo de capacidade de carga e de operacionalização das atividades de turismo náutico no Parque Nacional Marinho de Fernando de Noronha. Technical Report. Brasília, 2009. 87pp.

Matheus, Z., Francini-Filho, R. B., Pereira-Filho, G. H., Moraes, F. C., Moura, R. L., Brasileiro, P. S., Amado-Filho, G. M. (2019). Benthic reef assemblages of the Fernando de Noronha Archipelago, tropical South-west Atlantic: Effects of depth, wave exposure and cross-shelf positioning. *PLoS ONE*, *14*(1):1–15. https://doi.org/https://doi.org/10.1371/journal.pone.0210664

Michener, W. K., Blood, E. R., Bildstein, K. L., Brinson, M. M., Gardner, L. R. (1997). Climate Change, Hurricanes and Tropical Storms, and Rising Sea Level in Coastal Wetlands. *Ecological Applications*, *7*(3):770. https://doi.org/10.2307/2269434

Monteiro, J.G., Costa, C.F., Gorlach-Lira, K., Fitt, W.K., Stefanni, S.S., Sassi, R., Santos, R.S., LaJeunesse, T.C. (2013). Ecological and biogeographic implications of *Siderastrea* symbiotic relationship with Symbiodinium sp. C46 in Sal Island (Cape Verde, East Atlantic Ocean). *Marine Biodiversity 43*:261-272

Murray, S. N., Denis, T. G., Kido, J. S., Smith, J. R. (1999). Human visitation and the frequency and potential effects of collecting on rocky intertidal populations in southern California marine reserves. *California Cooperative Oceanic Fisheries Investigations Reports*, *40*(August 2014):100–106.

Naranjo-Arriola, A. (2021). Tourist carrying capacity as a sustainability management tool for coral reefs in Caño Island Biological Reserve, Costa Rica. *Ocean & Coastal Management, 212:* 105857. https://doi.org/10.1016/j.ocecoaman.2021.105857

Norström, A. V., Nyström, M., Lokrantz, J., Folke, C. (2009). Alternative states on coral reefs: Beyond coral-macroalgal phase shifts. *Marine Ecology Progress Series*, *376*:293–306. https://doi.org/10.3354/meps07815

Nugues, M. M., Roberts, C. M. (2003). Coral mortality and interaction with algae in relation to sedimentation. *Coral Reefs*, 22(4):507–516. https://doi.org/10.1007/s00338-003-0338-x

Oliver, T. A., Palumbi, S.R. (2011). Many corals host thermally resistant symbionts in high-temperature habitat. *Coral Reefs* 30:241–250. https://doi.org/10.1007/s00338-010-0696-

Rambaut, A. (2009) FigTree: Tree figure drawing software for Apple Macintosh and Microsoft Windows. http://taxonomy.zoology.gla.ac.uk/rod/treeview.html

R Core Team (2022). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Rice, S. A., Hunter, C. L. (1992). Effects of suspended sediment and burial on scleractinian corals from west central Florida patch reefs. *Bulletin of Marine Science, 51*(3): 429–442.

Sanger F., Nicklen S., Coulson A. (1977) DNA sequencing with chain-terminating. *Proceedings of the National Academy of Sciences of the United States of America 74:*5463–5467.

Sassi, C. F. C., Farias, G. M. de, Vasconcelos, A. de S., Macedo, R. S. de, França, J. P. de S., Sassi, R. (2021). Histopatological effects of bleaching and disease on the coral Siderastrea stellata from coastal reefs of Brazil. *Iheringia. Série Zoologia 111*:1–8. https://doi.org/10.1590/1678-4766e2021007

Shokri, M., Mohammadi, M. (2021). Effects of recreational SCUBA diving on coral reefs with an emphasis on tourism suitability index and carrying capacity of reefs in Kish Island, the northern Persian Gulf. *Regional Studies in Marine Science, 45:*101813. https://doi.org/10.1016/j.rsma.2021.101813

Storlazzi, C. D., Fregoso, T. A., Golden, N. E., Finlayson, D. P. (2011). Sediment dynamics and the burial and exhumation of bedrock reefs along an emergent coastline as elucidated by repetitive sonar surveys: Northern Monterey Bay, CA. *Marine Geology*, *289*(1–4): 46–59. https://doi.org/10.1016/j.margeo.2011.09.010

Swain, T.D., Chandler, J., Backman, V., Marcelino, L. (2017) Consensus thermotolerance ranking for 110 Symbiodinium phylotypes: an exemplar utilization of a novel iterative partial-rank aggregation tool with broad application potential. *Functional Ecology 31:*172–183.

Tunala, L. P., Tâmega, F. T. S., Duarte, H. M., Coutinho, R. (2019). Stress factors in the photobiology of the reef coral Siderastrea stellata. *Journal of Experimental Marine Biology and Ecology*, *519*(July): 151188. https://doi.org/10.1016/j.jembe.2019.151188

Valentine, J. F., Heck, K. L. (2005). Perspective review of the impacts of overfishing on coral reef food web linkages. *Coral Reefs*, *24*(2):209–213. https://doi.org/10.1007/s00338-004-0468-9

Vaselli, S., Bertocci, I., Maggi, E., Benedetti-Cecchi, L. (2008). Effects of mean intensity and temporal variance of sediment scouring events on assemblages of rocky shores. *Marine Ecology Progress Series*, *364*:57–66. https://doi.org/10.3354/meps07469

Thompson, R. C., Crowe, T. P., Hawkins, S. J. (2002). Rocky intertidal communities: Past environmental changes, present status and predictions for the next 25 years. *Environmental Conservation* *29*(2):168–191. https://doi.org/10.1017/S0376892902000115

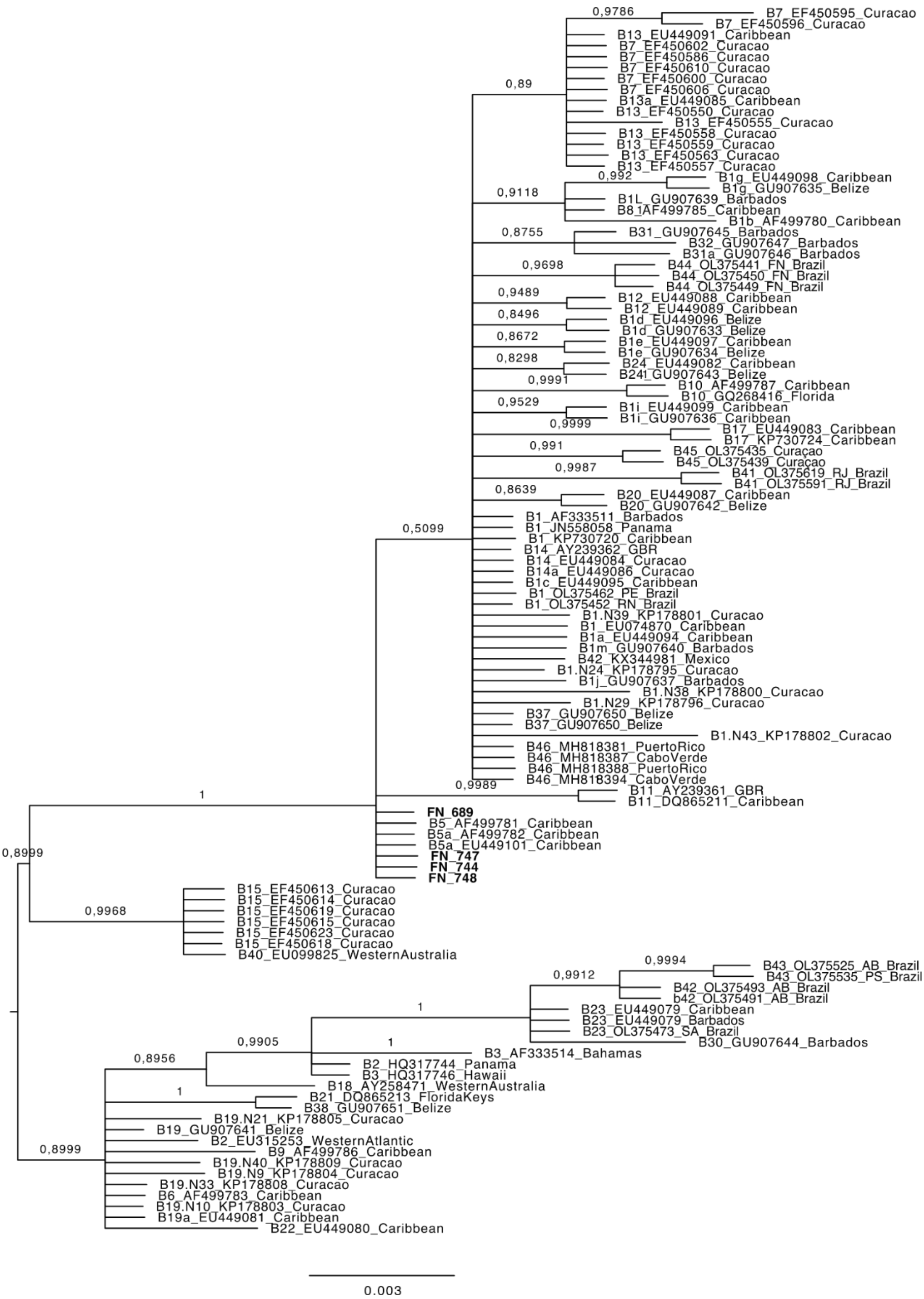
Supplementary Material

Table S1. Monthly number of visitors to the Atalaia tidepool during the study period.

Month	Visitors	Month	Visitors
nov-17	1485	jan-19	1576
dez-17	1418	fev-19	1526
jan-18	1534	mar-19	1539
fev-18	1397	abr-19	1461
mar-18	1678	mai-19	1498
abr-18	1546	jun-19	1257
jun-18	1457	jul-19	1588
ago-18	1084	ago-19	1612
set-18	990	set-19	1514
out-18	1139	out-19	1581
nov-18	1049	nov-19	1383
dez-18	1351	**mean ± SD**	**1420 ± 191**

Figure S1 Bayesian phylogenetic trees based on ITS2 rDNA from the genera (a) *Breviolum* and *Cladocopium* (b: with strains from Atlantic ocean; c: with only C1 and C3 representatives to resolve close relationships). Branch support from posterior probability is indicated. Reference sequence names include phylotype identification, GenBank code and collection location. Sequences from this work are bold.

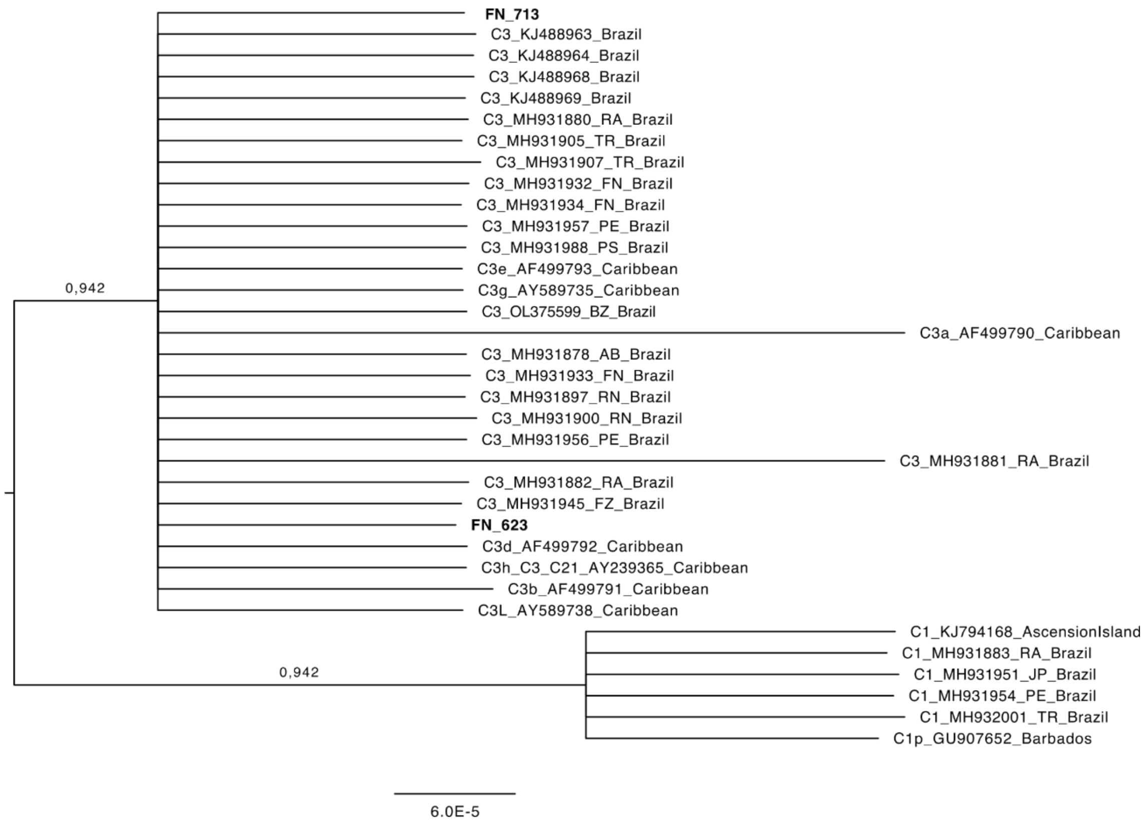
a.


0,9786
B7_EF450595_Curacao
B7_EF450596_Curacao
B13_EU449091_Caribbean
B7_EF450602_Curacao
B7_EF450586_Curacao
B7_EF450610_Curacao
B7_EF450600_Curacao
0,89
B7_EF450606_Curacao
B13a_EU449085_Caribbean
B13_EF450550_Curacao
B13_EF450555_Curacao
B13_EF450558_Curacao
B13_EF450559_Curacao
B13_EF450563_Curacao
B13_EF450557_Curacao
0,992
B1g_EU449098_Caribbean
0,9118
B1g_GU907635_Belize
B1L_GU907639_Barbados
B8_AF499785_Caribbean
B1b_AF499780_Caribbean
0,8755
B31_GU907645_Barbados
B32_GU907647_Barbados
B31a_GU907646_Barbados
0,9698
B44_OL375441_FN_Brazil
B44_OL375450_FN_Brazil
B44_OL375449_FN_Brazil
0,9489
B12_EU449088_Caribbean
B12_EU449089_Caribbean
0,8496
B1d_EU449096_Belize
B1d_GU907633_Belize
0,8672
B1e_EU449097_Caribbean
B1e_GU907634_Belize
0,8298
B24_EU449082_Caribbean
B24_GU907643_Belize
0,9991
B10_AF499787_Caribbean
B10_GQ268416_Florida
0,9529
B1i_EU449099_Caribbean
B1i_GU907636_Caribbean
0,9999
B17_EU449083_Caribbean
B17_KP730724_Caribbean
0,991
B45_OL375435_Curaçao
B45_OL375439_Curaçao
0,9987
B41_OL375619_RJ_Brazil
B41_OL375591_RJ_Brazil
0,8639
B20_EU449087_Caribbean
B20_GU907642_Belize
B1_AF333511_Barbados
0,5099
B1_JN558058_Panama
B1_KP730720_Caribbean
B14_AY239362_GBR
B14_EU449084_Curacao
B14a_EU449086_Curacao
B1c_EU449095_Caribbean
B1_OL375462_PE_Brazil
B1_OL375452_RN_Brazil
B1.N39_KP178801_Curacao
B1_EU074870_Caribbean
B1a_EU449094_Caribbean
B1m_GU907640_Barbados
B42_KX344981_Mexico
B1.N24_KP178795_Curacao
B1j_GU907637_Barbados
B1.N38_KP178800_Curacao
B1.N29_KP178796_Curacao
B37_GU907650_Belize
B37_GU907650_Belize
B1.N43_KP178802_Curacao
B46_MH818381_PuertoRico
B46_MH818387_CaboVerde
B46_MH818388_PuertoRico
B46_MH818394_CaboVerde
0,9989
B11_AY239361_GBR
1
B11_DQ865211_Caribbean
FN_689
B5_AF499781_Caribbean
B5a_AF499782_Caribbean
B5a_EU449101_Caribbean
0,8999
FN_747
FN_744
FN_748
B15_EF450613_Curacao
B15_EF450614_Curacao
0,9968
B15_EF450619_Curacao
B15_EF450615_Curacao
B15_EF450623_Curacao
B15_EF450618_Curacao
B40_EU099825_WesternAustralia
0,9994
B43_OL375525_AB_Brazil
0,9912
B43_OL375535_PS_Brazil
B42_OL375493_AB_Brazil
b42_OL375491_AB_Brazil
1
B23_EU449079_Caribbean
B23_EU449079_Barbados
B23_OL375473_SA_Brazil
0,9905
1
B30_GU907644_Barbados
0,8956
B3_AF333514_Bahamas
B2_HQ317744_Panama
B3_HQ317746_Hawaii
B18_AY258471_WesternAustralia
1
B21_DQ865213_FloridaKeys
B38_GU907651_Belize
B19.N21_KP178805_Curacao
B19_GU907641_Belize
0,8999
B2_EU315253_WesternAtlantic
B9_AF499786_Caribbean
B19.N40_KP178809_Curacao
B19.N9_KP178804_Curacao
B19.N33_KP178808_Curacao
B6_AF499783_Caribbean
B19.N10_KP178803_Curacao
B19a_EU449081_Caribbean
B22_EU449080_Caribbean
0.003

b.


C12_AF499801_Caribbean
C12_EU446120_Bahamas
C12_KP178785_Curacao
C38a_AY258483_Caribbean
C7_DQ838548_Panama
C7.N22_KP178784_Curacao
C7.N32_KP178792_Curacao
C81_GU907657_Barbados
C90_AJ621128_Panama
C90_JN558045_Panama
C90_JN558047_Panama
C168_OL375587_ES_Brazil
C168_OL375589_ES_Brazil
C11N10_KP109697_PuertoRico
C11N12_KP109699_VirginIsland
C3f_AY589734_Caribbean
C3p_GU907654_Barbados
C76_AY765411_Caribbean
C77_AY765412_Caribbean
C1_KJ794168_AscensionIsland
C1_MH931879_RA_Brazil
C1_MH931883_RA_Brazil
C1_MH931885_RA_Brazil
C1_MH931895_ASC_Brazil
C1_MH931951_JP_Brazil
C1_MH931954_PE_Brazil
C1_MH932001_TR_Brazil
C130_MH931953_JP_Brazil
C16.N16_KP178786_Curacao
C167_MH818390_CapeVerde
C167_MH818392_CapeVerde
C167_MH818393_Spain
C1a_AF49978_Caribbean
C1p_GU907652_Barbados
C21_EF428341_Mexico
C3_KJ189562_Brazil
C3_KJ189563_Brazil
C3_KJ189564_Brazil
C3_KJ488963_Brazil
C3_KJ488964_Brazil
C3_KJ488968_Brazil
C3_KJ488969_Brazil
C3_OL375516_AB_Brazil
C3_OL375604_RJ_Brazil
C3_MH931878_AB_Brazil
C3_MH931880_RA_Brazil
C3_MH931881_RA_Brazil
C3_MH931882_RA_Brazil
C3_MH931884_RA_Brazil
C3_MH931897_RN_Brazil
C3_MH931900_RN_Brazil
C3_MH931905_TR_Brazil
C3_MH931907_TR_Brazil
C3_MH931932_FN_Brazil
C3_MH931933_FN_Brazil
C3_MH931934_FN_Brazil
C3_MH931945_FZ_Brazil
C3_MH931956_PE_Brazil
C3_MH931957_PE_Brazil
C3_MH931988_PS_Brazil
C3.N14_KP178779_Curacao
C3.N25_KP178781_Curacao
C38_AY258482_Caribbean
C3a_AF499790_Caribbean
C3b_AF499791_Caribbean
C3bN1_KF551185_Curacao
C3bN9_KP109696_VirginIslands
C3d_AF499792_Caribbean
C3e_AF499793_Caribbean
C3g_AY589735_Caribbean
C3h_C3C21_AY239365_Caribbean
C3L_AY589738_Caribbean
C48_AY589753_Caribbean
C50_KP109700_PuertoRico
C52_AY589757_Caribbean
C54a_AY765406_Caribbean
C6_AF499796_Caribbean
C72_KP109698_PuertoRico
C83_GU907661_Barbados
C93_EU446108_Bermuda
FN_566
0,7211
0,8821
0,9831
0,9344
0,8797
0,8296
0,9783
C46_EU449108_Caribbean
C46a_AY765405_Caribbean
FN_561
FN_683
FN_710
FN_691
FN_749
FN_750
FN_751
FN_801
FN_802
FNS_10
FNS_21
FNS_22R
FNS_6
FNS_7R
FNS_9
RN_269F
RN_274
RN_378
RN_379
C44.N17_KP178787_Curacao
C44.N18_KP178788_Curacao
C44.N19_KP178789_Curacao
C10_AF499799_Caribbean
C1c_MN876163_China
C4_AF499794_Caribbean
C42_MN876144_China
C43_AY589749_Caribbean
C9_AF499798_Caribbean
0,9882
0,9637
0,9783
6.0E-4

c.


FN_713
C3_KJ488963_Brazil
C3_KJ488964_Brazil
C3_KJ488968_Brazil
C3_KJ488969_Brazil
C3_MH931880_RA_Brazil
C3_MH931905_TR_Brazil
C3_MH931907_TR_Brazil
C3_MH931932_FN_Brazil
C3_MH931934_FN_Brazil
C3_MH931957_PE_Brazil
C3_MH931988_PS_Brazil
C3e_AF499793_Caribbean
C3g_AY589735_Caribbean
C3_OL375599_BZ_Brazil
C3a_AF499790_Caribbean
C3_MH931878_AB_Brazil
C3_MH931933_FN_Brazil
C3_MH931897_RN_Brazil
C3_MH931900_RN_Brazil
C3_MH931956_PE_Brazil
C3_MH931881_RA_Brazil
C3_MH931882_RA_Brazil
C3_MH931945_FZ_Brazil
FN_623
C3d_AF499792_Caribbean
C3h_C3_C21_AY239365_Caribbean
C3b_AF499791_Caribbean
C3L_AY589738_Caribbean
C1_KJ794168_AscensionIsland
C1_MH931883_RA_Brazil
C1_MH931951_JP_Brazil
C1_MH931954_PE_Brazil
C1_MH932001_TR_Brazil
C1p_GU907652_Barbados
0,942
0,942
6.0E-5

CAPÍTULO 3

Marine Pollution in Fernando de Noronha Archipelago, an oceanic marine protected area in tropical SW Atlantic

Capítulo 3

Marine Pollution in Fernando de Noronha Archipelago, an oceanic marine protected area in tropical SW Atlantic

Thayná J. Mello, Cybelle M. Longhini, Bruno Mattos Silva Wanderley, Cesar Alexandro da Silva, Bethânia Dal'Col Lehrback, Fábio Cavalca Bom, Renato Rodrigues Neto, Fabian Sá, Guilherme O. Longo

*Manuscript in preparation, to be submitted to the periodic *Science of the Total Environment*

Abstract

Reefs worldwide are facing a decline, with sewage pollution recognized as a major threat, impacting the biota and interacting with other anthropogenic impacts in ways that are still poorly understood. Wastewater pollution affects coastal waters globally, often due to inadequate wastewater management infrastructure resulting from disorderly urbanization. Fernando de Noronha Archipelago (FN), a set of oceanic marine protected areas in southwestern Atlantic with outstanding importance to reef conservation, has experienced great population growth driven by tourism development. However, the sewage infrastructure has not kept pace with this growth. To understand the impacts of sewage disposal and other pollutants on the reefs, we conducted a comprehensive study mapping and quantifying marine pollution in FN. We assessed 13 sites across FN, characterizing pollution sources and evaluating their effects on reef benthic and fish communities. We quantified pollutants (nutrients, metals, microplastics, fecal sterols, and Polycyclic Aromatic Hydrocarbons - PAHs) in water and sediment samples obtained around the archipelago. Isotopic analysis of macroalgae samples helped determine the origin of organic matter incorporated into the reefs. We also characterized benthic and fish communities, and algae biomass at the collection sites. While pollution was more pronounced in the multiple-use area, it was also present in the no-take area. This suggests that pollutants discharged into the multiple-use area are transported by currents towards the no-take area along the leeward coast, and from distant sources to the windward coast. Although nutrient concentrations in the water were generally low, the effluents from wastewater treatment plants did not meet legislation standards. The isotopic signature of nitrogen in macroalgae indicated the presence of sewage-derived nitrogen throughout the multiple-use area. Sites subject to nutrient enrichment exhibited distinct benthic communities, characterized by higher abundances of fast-growing green macroalgae, higher biomass of brown macroalgae, and the presence of opportunistic macroalgae. The sediment analysis revealed general low concentrations of PAHs and coprostanol, and metal concentrations below the Probable Effect Level (PEL) for all elements and sites. However, the multiple-use area, particularly Porto, showed higher levels of contaminants, indicating potential untreated sewage and chemical pollution. Golfinhos, a site within the no-take area presented a much higher concentration of metals (Fe, Al, Ni, Cr, Cu) and metalloid (As) than that observed in the other sampled areas, reaching the Threshold Effect Level (TEL) for Ni, Cr, and As. Microplastics were detected ubiquitously in both the sediments and waters of Fernando de Noronha, irrespective of their protection

category. This study provides baseline data and management recommendations. We highlight the need to improve wastewater and sewage management infrastructure to protect the marine ecosystem in the archipelago.

Introduction

Reefs currently experience multiple impacts derived from human activities, on local and global scales, and are in decline worldwide (Eddy *et al.* 2021). The establishment of marine protected areas (MPAs) is one of the main strategies adopted globally for marine conservation (Roberts and Hawkins 2000; Dalongeville *et al*. 2022). Recent climate-related warming causing coral bleaching had drastic impacts on reefs around the world, putting the effectiveness of MPAs in protecting the biota into question (Edgar *et al.* 2014; Bruno *et al.* 2018). If on the one hand conserving coral reefs into the future must rely on global political action (Anthony et al. 2020; Baumann *et al*. 2022), local management practices can increase the resistance and resilience of reef ecosystems to climate impacts (Wooldridge & Done 2009; Bruno et al. 2018), especially in reefs that have not suffered major disturbances (Baum et al. 2023). On local scales, overfishing and pollution are recognized as major threats to coral reefs, but conservation and recovery of water quality have received significantly less attention and investments by managers than fishing regulations (Burk et al. 2011; Wear *et al*. 2021).

Environmental quality is affected by the disposal of pollutants in marine systems, both from point and non-point sources, through water bodies, terrestrial runoff, subsurface groundwater, vessels, and wastewater, affecting both human and ecological health (van Dam 2011, Tuholske *et al*. 2021). Inadequately treated sewage and other sources of marine pollution add nutrients, pathogens, sediments, organic matter, suspended solids, pharmaceuticals, household chemicals, soaps, petrochemicals, heavy metals, endocrine disruptors, and micro and macro plastics to natural systems (Tilley et al. 2014, Wear and Thurber 2015).

The effect of untreated sewage discharge on reef ecosystems has been evidenced in many regions of the world (Pastorok and Bilyard 1985; Hunter and Evans 1995; Duprey et al. 2017). Nutrient enrichment and the presence of pathogens in sewage can result in destabilization of the local microbiome, increased disease occurrence and decreased coral cover, macroalgae proliferation, decline in fish abundance, and changes in the balance of interactions between organisms (Pastorok and Billard 1985; Zeneveld *et al*. 2016, Shaver *et al*. 2018). Nutrient enrichment can be especially detrimental in tropical reefs, because they are generally oligotrophic, which makes them more sensitive to the input of limiting macronutrients (Adam et al. 2021, Wear and Thurber 2015).

Plastic pollution is also a major concern to biodiversity conservation (Santos *et al*. 2021). Plastics can be ingested by organisms at the base of food webs, suggesting that they can be transferred further across trophic levels, and cause entanglement of several groups of marine species, like seabirds, sea turtles, mammals, and fish (Ostle *et* al. 2019). Plastic pollution has also been linked to higher disease prevalence in corals by promoting colonization by pathogens (Lamb *et al*. 2018). Inorganic nutrients and plastic pollution are considered important contaminants affecting coral reefs, but each type of anthropogenic contaminant can impact the biota and interact with each other, and with other environmental impacts in poorly understood ways (van Dam 2011, Wear and Thurber 2015, Tuholske et al. 2021)**.** Contamination by pesticides, heavy metals, hydrocarbons, and other pollutants (e.g. organometallic substances and industrial organochlorines) can affect the health, growth, reproduction, and survival of reef organisms (Fabricius *et al*. 2005, van Dam 2011, Wear and Thurber 2015).

Despite the importance of chemical pollution to human and ecosystem health, few field studies have investigated its impacts on reef ecosystems (Wear and Thurber 2015, Tuholske et al. 2021). Wastewater pollution affects coastal waters all around the world and Brazil is the fourth country in the world with the highest total nitrogen input into coastal waters, with ~30% untreated wastewater inputs (Tuholske et al. 2021). Contamination by domestic sewage in Brazil is linked to disorderly urbanization associated with a lack of wastewater management infrastructure and threatens not only large urban centers but also remote areas which experience recent population growth due to the development of tourism and industries (Araújo et al. 2021).

This may be the case of Fernando de Noronha archipelago (FN), consisting of two marine protected areas in the South Atlantic, including a designated Natural World Heritage Site (UNESCO 2001), that faces a challenging situation. The rapid growth of tourism on the island since 2010 has resulted in a substantial increase in both visitors and residents, outpacing the development of local infrastructure (Cristiano et al. 2020, Araújo et al. 2023). This concerning trend of escalating tourism and urban expansion has caught the attention of UNESCO, which has classified the World Heritage Site as being of "significant concern" in its latest assessment (IUCN 2020). The archipelago is in a region of the globe where cumulative anthropogenic impacts (thermal anomalies, nutrient pollution, shipping and fishing) have also increased recently (Halpern et al. 2015, Magris et al. 2021, Bastos et al. 2022). Studies conducted before the recent upsurge of tourism in FN already detected signs of marine pollution. Biomarker analyses in foraminiferal assemblages found high concentrations of zinc, copper, lead, and dissolved organic carbon (Prazeres et al. 2012), and an evaluation of sediments also revealed higher contents of organic matter in sites that receive wastewater and direct sewage outflow (Barcelos *et al*. 2018), and a high concentration of surfactants has been observed in water samples (Araújo et al. 2023). However, it is still not clear to what extent pollution from the island affects marine ecosystems around FN and whether current and cumulative impacts are detectable in the marine community structure.

Water properties in FN were investigated from samples obtained about 3 km off the archipelago in 2013 (Bertini and Braga 2022), and for samples obtained on the leeward coast in 2013-2014 (Assunção et al. 2016). Reference values for the concentrations of heavy metals and trace elements available for the benthic seaweeds (Ferreira et al. 2012) and soils reflect the volcanic origin of the archipelago, tending to be higher than those of mainland soils (Oliveira et al. 2011, Fabricio Neta et al. 2018). The lack of long-term reference data on environmental quality and marine community structure is a widespread problem, but information on the magnitude and spatial distribution of pollution is essential to prompt informed mitigation measures and monitoring (Risk et al. 2009). We investigated the spatial patterns of marine pollution in FN aiming to map and quantify the main sources of impact and their effects on reef communities. For that, we characterized marine communities and evaluated water and sediment to quantify multiple pollutants around the archipelago. Additionally, we used stable isotopes to evaluate the sources of the nutrients absorbed by the macroalgae in the study sites.

Materials and Methods

Study site

Fernando de Noronha (FN) is a volcanic archipelago (3°50′S, 32°24′W) located ~360 km off the coast of Brazil, in the Western Atlantic, formed by 21 islands and islets with a total area of ~ 27km^2. The climate is tropical oceanic with an average annual rainfall of 1,400mm. Rain is unevenly distributed throughout the year, with a rainy season between March and July and a dry season between August and February. The archipelago is under the influence of the South Equatorial Current, which flows from East to West and is characterized by warm (28°C) and oligotrophic waters. Exposure to currents and winds affects the island's leeward and windward coasts differently, influencing each coast's biological, oceanographic, and management aspects (Matheus et al. 2019, Grillo and Mello 2021).

Rocky reefs and isolated outcrops covered mainly by algal turf and macroalgae predominate up to 60 m, with low coral cover (Zamoner et al. 2021). However, coral cover can reach 20% in some sites on the leeward coast (Krajewsky and Floeter 2011). Carbonate fringes dominated by coralline algae occur up to 20 m, mainly on the windward coast (Matheus et al. 2019). Fish assemblages are characterized by a high abundance of a few species (Krajewsky and Floeter 2011), but FN has the greatest fish richness among South Atlantic oceanic islands (Pimentel et al. 2020).

The archipelago is listed as Natural World Heritage Site because of its importance as breeding and feeding area for tuna, sharks, sea turtles, tropical seabirds, and marine mammals, with an exceptional population of resident dolphins in the Baía dos Golfinhos (UNESCO 2001). Fernando de Noronha comprises two protected areas: a no-take Marine National Park (MNP) and a multiple-use Environmental Protected Area (EPA), corresponding to IUCN categories II and VI respectively (Dudley 2008). The no-take area covers the secondary islands, 70% of the terrestrial area of the main island, plus the marine area to approximately 50 m depth on the secondary islands and the entire windward coast and part of the leeward coast of the main island. The rest of the archipelago is covered by the multiple-use area (Fig. 1). The only permitted activity within the no-take area is sustainable tourism, subject to strict regulations to minimize the impact of visitors on the park’s ecosystem. All the urban infrastructure and human occupation are located on the main island, in the multiple-use area where regulated fishing is allowed, and human activities must occur in a way that minimizes environmental impacts.

The resident population in FN is estimated at 4,700 people. Tourism is the basis of the local economy, and the number of visitors has been increasing consistently in recent years. In 2011, FN received 62,000 visitors; in 2021, there were more than 110,000 (ICMBio 2021). Because there are no perennial watercourses, there are artificial water reservoirs that supply the population demand. Only around 66% of the buildings are connected to the sewage system that sends the products to two treatment stations. The sewage system is overburdened, with cracks and clogs, and leakages and overflows are frequent. Septic tanks are used in 31% of the buildings, and the remaining 3% release sewage in the open (Cristiano et al. 2020). FN has no significant agricultural activities, and the use of fertilizers is not allowed. Livestock is also limited to a small number, with no more than 400 animals present in the area.

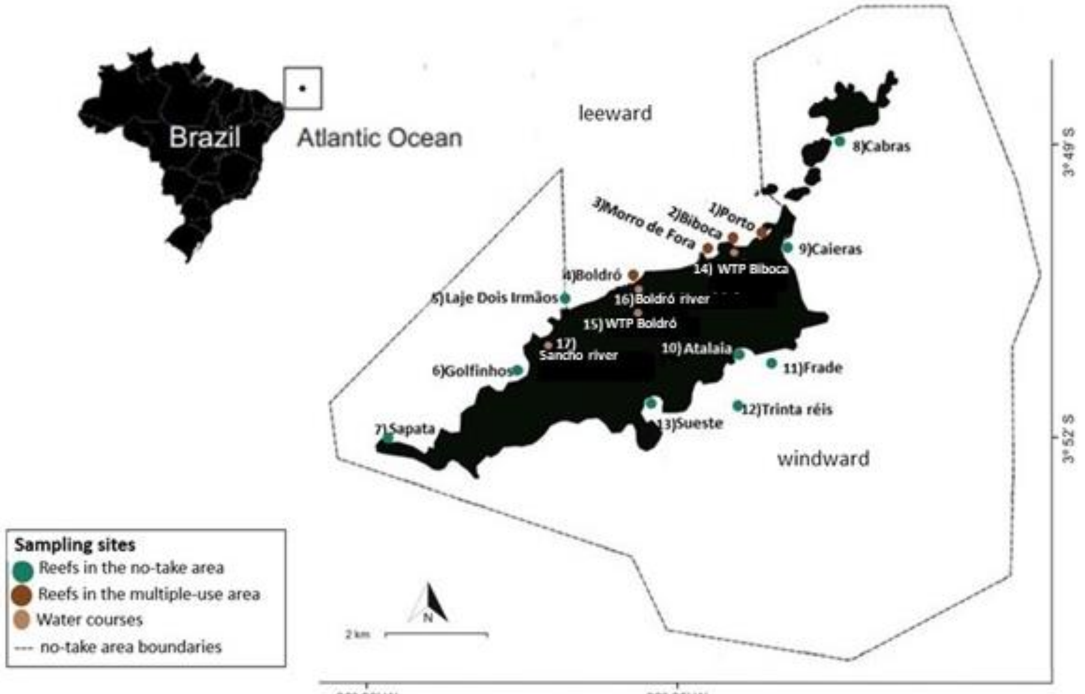


Figure 1: Location and sampling sites around the Fernando de Noronha archipelago and freshwater courses to evaluate multiple pollution sources and marine communities. WTP: wastewater treatment plants.

Environmental quality assessment

Multiple geochemical tracers were selected to evaluate the environmental quality and the main sources of pollutants in the FN. We determined dissolved inorganic nutrients (orthophosphate, nitrite, nitrate, and N-ammoniacal) and microplastics concentrations in the water; stable isotopic and elemental signatures of macroalgae; and concentration of metal (Al, Cd, Cr, Cu, Fe, Ni, Pb, Zn) and metalloid (As), total PAHs, fecal sterols and microplastics (MPs) in the sediments. Target contaminants, their criteria to assess environmental quality and their importance to the study area are summarised in Table 1.

To determine contamination, water quality parameters were compared to values established by the Brazilian legislation (Resolução CONAMA N° 357/2005, Resolução CONAMA N° 397/2008, Resolução CONAMA N° 430/2011), which do not establish threshold values for dissolved orthophosphate, but for total phosphorus (P-Total). Orthophosphate serves as an indicator of sewage effluents, which can stem from sources such as human waste and phosphate-based detergents and soaps. Since phosphorus is a nutrient that limits algal growth, elevated orthophosphate levels can be associated with the occurrence of harmful algal blooms, underscoring its importance in water quality monitoring (Showkat and Najar 2019). Wastewater treatment processes can significantly reduce the concentrations of nitrogen and phosphorous in the effluents, so the presence of high levels of these nutrients indicates untreated or poorly treated sewage (Preisner 2020).

Stable isotopic ratios are widely used as indicators of nutrient source, with $\delta^{15}N$ being especially useful as a signal of sewage pollution (Risk 2009, Alldred et al. 2023). There are no reference values for isotopic ratios at the study locality, so the obtained results were compared to the literature (Constanzo et al. 2001, Michener and Lajtha 2007, Risk 2009, Roth et al. 2016, Alldred et al. 2023). We considered $\delta^{15}N$ values up to 3‰ as background levels, above 3‰ indicating the presence of anthropogenic input (humaanimal sewage) and above 6‰ indicating strong exposure to sewage discharge. When sewage treatment process is efficient in removing nitrogen the $\delta^{15}N$ isotope signature exhibits higher values (>10‰) and can reach more than 23‰ (Onodera *et al* 2021).

Table 1: Target contaminants, criteria to assess environmental quality and relevance to the study area.

Environmental compartment	Target contaminants or Geochemical tracers	Criteria for environmental quality	Relevance to the study area
Water	Dissolved inorganic nutrients (orthophosphate, nitrite, nitrate, N-ammoniacal)	Delivered by sewage effluents and wastewater; nutrients to primary producers and algae blooms can occur under enrichment conditions	Identify sources of pollution by sewage effluents
	Stable isotopic and elemental signatures in macroalgae ($\delta^{15}N$ and $\delta^{13}C$)	Indicator of nutrient and anthropogenic organic matter sources; $\delta^{15}N$ is a sewage tracer and $\delta^{13}C$ distinguishes marine and terrestrial inputs	Identify sources of pollution by sewage effluents
	Microplastics	Widespread plastic pollution	Map areas with higher plastic pollution by human uses and marine transport
Sediment	Metals (Al, Cd, Cr, Cu, Fe, Ni, Pb, Zn) and metalloid (As)	Indicator of pollution by sewage, ship traffic and maintenance	Map areas with metal/metalloid contamination and their relation with anthropogenic sources
	PAHs	Indicators of oil and fossil fuel pollution	Identify pollution by ship traffic
	Fecal sterols	Tracer of fecal contamination by human feces	Identify sources of pollution by sewage effluents
	Microplastics	Widespread plastic pollution	Map areas with higher plastic pollution by human uses and marine transport

In general, $\delta^{13}C$ indicates the position of the organism and its habitat in the gradient of terrestrial to marine-influenced environments, with more negative values indicating terrestrial influence, whilst less negative values indicating marine influence (Lapointe et al. 2021, Michener and Lajtha 2007, Roth et al. 2016). Additionally, the molar ratio of C:N is used to assess the availability of N for macroalgae growth, with values above 13:1 indicating that N is a growth-limiting nutrient (Lapointe et al. 2021). Conversely, lower C:N ratios indicate reduced N-limitation to algal growth (Lapointe et al. 2010).

The presence of coprostanol in the environment, a fecal sterol produced in the intestines of warm-blooded animals, is widely used as an indicator of fecal contamination, alongside other fecal sterols (epicoprostanol, cholesterol, cholestanol, and coprostanone). Coprostanol can represent up to 60% of all fecal sterols in human feces, making it a commonly used marker to evaluate the presence of domestic sewage in the environment (Araújo et al. 2021).

Polycyclic aromatic hydrocarbons (PAHs) are hazardous to marine environments as they can be toxic to marine organisms, causing a range of effects such as reduced growth, developmental abnormalities, reproductive problems, and even death. PAHs can also persist in sediments, where they can accumulate and pose a risk to marine life (Sachaniya et al. 2019).

Sampling procedures

We sampled a total of 17 sites, including 4 reefs in the multiple-use area, 9 reefs in the no-take area, two freshwater courses where the effluents from the wastewater treatment plants are discharged (WTP Biboca and WTP Boldró), one river in the no-take area, and one brackish river (Boldró river, salinity around 22) in the multiple-use area (Fig. 1). All fieldwork was conducted during the rainy season in April 2022. Boldró river is located at the Boldró beach, downstream of the sewage treatment plant disposal site (about 300 m) and also receives disposal from a desalination plant. In the reef sites sampled depths ranged from 1-19 m, in addition to a tide pool (Atalaia 0.6 cm). Reef structural complexity was estimated by assigning scores to habitat attributes (adapted from Roos et al. 2019): topography, with 1= flat, 2= scattered boulders, 3= abundant boulders; height, with 1=

small, 2= medium, 3= high; and amount of refuges, with 0= none or very few, 1= few, 2= medium, 3= many. The final score was obtained by the sum of the scores of each attribute, with equal weights. We also recorded the depth of each transect using dive computers and measured the distance from the shore using GPS (Table 2). We conducted environmental quality and biological sampling in each reef using SCUBA diving.

Table 2: Sampling in Fernando de Noronha Archipelago. Type - R: reef, E: effluent, B: brackish, F: freshwater. MPA: marine protected area category MU: multiple-use, NT: no-take. Coast- LW: leeward, WW: windward. Dist: distance from coast. Score: complexity score. T: bottom water temperature. Transects- F: fish census, PQ: photoquadrats, BQ: biomass quadrats. Water - TN: total nutrients, DN: dissolved nutrients, MP: microplastics. Sediments- N & M: nutrients & metals, OR & MP: organic compounds & microplastics. ISO: Number of macroalgae species sampled for isotopic analysis. NA: not applicable, NC: not collected, WTP: wastewater treatment plant.

#	**Site**	**type**	**MPA**	**coast**	**coordinates**	**dist (m)**	**score**	**depth (m)**			**pH**	**T (°C)**	**transects**			**water (l)**			**sediment (samples)**		**ISO**
					latitude S longitude W			**min**	**mean**	**max**			**F**	**PQ**	**BQ**	**TN**	**DN**	**MP**	**N&M**	**OR&MP**	
1	Porto	**R**	MU	LW	03°50'02.0" 32°24'06.4"	230	4	3.7	4.5	5.5	8.4	29.9	5	50	3	0.25	3.3	1.2	3	3	6
2	Biboca	**R**	MU	LW	03°50'08.1" 32°24'24.4"	205	2	5.3	5.5	6.6	8.4	28.7	5	50	3	0.25	3.3	1.2	3	3	6
3	Morro de Fora	**R**	MU	LW	03°50'15.5" 32°25'03.3"	240	6	9.3	11.6	13.3	8.4	27.4	4	40	3	0.25	3.3	1.2	3	3	7
4	Boldró	**R**	MU	LW	03°50'37.3" 32°25'48.2"	170	2	6.6	7	7.3	8.5	28.6	5	50	3	0.25	3.3	1.2	3	3	5
5	Laje Dois Irmãos	**R**	NT	LW	03°50'.39.9" 32°26'27.9"	535	8	17.8	18.4	19.4	8.4	27.4	4	40	3	0.25	3.3	1.2	1	1	3
6	Golfinhos	**R**	NT	LW	03°51'31.6" 32°27'02.7"	303	7	8	8.8	10.3	8.4	28.9	5	50	3	0.25	3.3	1.2	1	1	3
7	Sapata	**R**	NT	LW	03°52'28.0" 32°28'28.4"	56	8	11.2	14	17.4	NC	NC	4	40	3	0.25	3.3	1.2	1	1	2
8	Cabras	**R**	NT	WW	03°49'03.8" 32°23'13.3"	230	6	8.8	8.8	8.8	8.7	29	5	50	3	0.25	3.1	1.3	1	1	2
9	Caieras	**R**	NT	WW	03°50'27.9" 32°23'33.6"	430	5	11.5	12.3	12.9	8.7	29	5	50	3	0.25	3.3	1.3	1	1	4
10	Atalaia	**R**	NT	WW	03°51"25.6" 32°24'33.1"	0	3	0.5	0.6	0.7	NC	NC	5	50	3	0.25	3.4	1.2	1	1	6
11	Frade	**R**	NT	WW	03°51'39.8" 32°24'06.3"	416	7	13.1	15.7	18	8.4	29	4	40	3	0.25	3.3	1.2	1	1	4
12	Trinta Réis	**R**	NT	WW	03°52'01.9" 32°24'45.5"	340	5	11.4	13.3	17.2	8.4	29	4	40	3	0.25	2.2	1.2	1	1	7
13	Sueste	**R**	NT	WW	03°51'59.7" 32°25'34.4"	120	3	1	1	1.2	8.4	31	10	100	3	0.25	3.3	1.2	2	2	7
14	WTP Biboca	**E**	MU	LW	03°50'21.7" 32°24'28.8"	NA	NA	NA	NA	NA	NA	NA	NA	NA	3	0.25	0.3	0.3	NA	NA	NA
15	WTP Boldró	**E**	MU	LW	03°50'50.6" 32°25'46.2"	NA	NA	NA	NA	NA	7.8	28	NA	NA	3	0.25	0.3	NC	NA	NA	NA
16	Boldró river	**B**	MU	LW	03°50'44.4" 32°25'47.0"	NA	NA	NA	NA	NA	8.2	29.4	NA	NA	3	0.25	0.5	0.5	1	1	2
17	Sancho river	**F**	NT	LW	03°51'17.7" 32°26'33.3"	NA	NA	NA	NA	NA	7.2	28.3	NA	NA	3	0.25	1	0.6	NA	NA	NA

Water samples collection

Water samples were collected in polyethylene bottles to determine water quality parameters: dissolved (orthophosphate, nitrite, nitrate, and N-ammoniacal), while a one-liter amber glass bottle was used to evaluate microplastics in suspended particulate matter (SPM). The containers were submerged to the collection depth using SCUBA gear and water was collected 1 m above the maximum depth of the site. The samples were kept under refrigeration in the vessel (using thermal boxes with ice) until landing. Sample processing started no later than 4 hours after collection. All material for nutrient analysis underwent a cleaning and decontamination process with 10 % hydrochloric acid, and ultrapure water before collection.

Macroalgae collection

We collected three replicates of green and brown macroalgae species found on each reef to determine the stable isotopic and elemental signatures ($\delta^{15}N$, $\delta^{13}C$, %N, %C, and C: N ratio). The samples were stowed in Ziplock bags. In the laboratory, they were dried in a heating chamber set at a temperature of 50° for 48 hours and approximately 0.5 g were kept in 5ml Eppendorf tubes until analysis.

Sediment collection

Sediment samples (approximately 2 cm surface layer) to measure N and C percentages, metal and metalloid concentrations (Al, As, Cd, Cr, Cu, Fe, Ni, Pb, Zn), organic contaminants (polycyclic aromatic hydrocarbons – PAHs and lipid biomarkers - sterols), and microplastics. Aliquots for organic and microplastic analysis were collected with glass containers (100 ml) previously calcined at 450 °C in a ceramic muffle furnace. For metal and metalloid determination, samples were collected and stored in acid-cleaned polyethylene bottles (50 mL). To avoid contamination, samples were always handled using nitrile gloves. All samples were kept frozen until sample processing.

Biological sampling consisted of benthic community and fish community assessments, and algae biomass estimates. We assessed benthic cover using photoquadrats (50 x 50 cm) placed every 2 m on 20 m transects. We used 4 or 5 transects on each site, totalling 65 transects. The percent cover of each substratum was estimated using 25 random points on each photoquadrat, with the semi-automatic tool of CoralNet (Beijbom et al. 2012). Each point was identified to the lowest taxonomic level possible, but due to morphological similarity, the brown macroalgae belonging to the genera *Canistrocarpus, Dictyota* and *Dictyopteris* were grouped as "brown macroalgae" for the analysis.

Fish assemblages and algae sampling

We assessed the fish assemblages from belt transects (20 x 2 m) in the same transects used for benthic cover assessment. The fish recorded in the transect were counted and their size was estimated. We used allometric equations from FishBase database to estimate fish biomass (Froese and Pauly 2023).

To estimate algae biomass, we scrapped all the algae in 25 x 25 cm frames and placed them inside ziplock bags at three points along the first transect previously used for community assessment (0 m, 10 m, and 20 m), totalling 42 biomass quadrats across all sites. In the laboratory, the samples were rinsed with seawater, the excess of water was removed using a manual centrifuge, the algae were sorted and identified to the lowest possible taxonomic level and weighted (wet weight).

Sample processing and geochemical analyses

Water samples

For dissolved inorganic nutrients, seawater samples were filtered through 0.45 μm nitrocellulose membrane (MF-Millipore, 47 mm) and analysed for orthophosphate, nitrite, nitrate, and N-ammoniacal using the colorimetric method according to Grasshoff *et al*. (1999). Absorbance was measured by UV/VIS spectrophotometer (Biospectro SP-220) and calibration curves (from 0.10 to 10 μM) were used to determine each nutrient concentration by the linear regression ($y = ax + b$). When nutrient levels were higher than the calibration curve interval, dilution was conducted using ultrapure water and the final concentration was corrected by the dilution factor. The Limits of Detection (LOD) were 0.001 mg L^{-1} for orthophosphate and nitrite, 0.002 mg L^{-1} for nitrate and 0.004 mg L^{-1} for N-ammoniacal. Suspended Particulate Material (SPM) was determined using the gravimetric method after filtration through a dried and pre-weighted membrane.

Macroalgae isotope as organic matter tracer

Isotope ratio of $\delta^{15}N$, $\delta^{13}C$ and C:N ratio analysis were performed by continuous flow isotope ratio mass spectrometry (Flash EA – Delta V). The normalization of the results was carried out with at least two certified international standards from the IAEA or USGS. The standard uncertainty of the CF-IRMS analysis for $\delta^{13}C$ is ± 0.10‰ and for $\delta^{15}N$ is ±0.15‰.

Sediment samples

Sediment samples were freeze-dried and ground to pounder for metals, metalloid, and organic compounds analyses (PAHs and sterols). The method EPA 3051A (U.S. EPA 2007) was used for partial digestion of metals and metalloid from sediment. For this, approximately 0.50 g of sediment was placed into sealed Teflon® tubes with 10 mL of distilled HNO_3^- 65% and heated in a microwave digester Mars 6 X-press, 900-1050 W (©CEM Corporation, Matthews, NC, USA) (5.5 °C/min ramp time for 5:30 min; kept at 175 °C for 4:30 min). After cooling, each extract was filtered through a quantitative filter and stored. Aluminum, As, Cd, Cr, Cu, Fe, Ni, Pb, and Zn were quantified by Inductively Coupled Plasma Optical Emission Spectrometry (ICP – OES model iCAP 6300 Duo; Thermo Fisher Scientific, Bremen, Germany) according to the method EPA 6010D (U.S. EPA 2018). The analytical quantification was determined from the calibration curve (from 10.0 to 2560.0 $\mu g.L^{-1}$ for minor elements; and from 500.0 to 128,000.0 $\mu g.L^{-1}$ for major metals) using a multi-element standard solution (AccuStandard®, New Haven, CT USA). Internal and external standards were used to assess precision and accuracy, with errors lower than 5 % and accuracy higher than 95 %. The Limit of Detection (LOD) and Limit of Quantification (LOQ) for metals and metalloid analysed by ICP-OES are shown in Table S1.

Metal concentrations were compared to Threshold Effect Limit (TEL - the concentration at which toxicity started to be observed in benthic organisms) and Probable Effect Limit (PEL - the concentration at which a large percentage of the benthic community shows a toxic response) values in reference tables developed by National Oceanic and Atmospheric Administration (NOAA) (Buchman 2008). These reference values are used for screening only, and understanding their effects requires accounting for factors that modify sediment toxicity depending on the organisms and exposure pathways (Simpson and Batley 2007).

For PAHs and sterols determination in sediments, samples were extracted with dichloromethane/methanol (9:1) according to the EPA 3540C Soxhlet method (U.S. EPA, 1996). The cleanup and fractionation steps were performed in concentrated extracts by elution them in a silica/alumina chromatography column. The fraction F2 rich in PAHs was eluted with dichloromethane: hexane (1:1), while the fraction F3 rich in sterols was eluted with ethyl acetate. The compounds were analysed in Agilent Technologies 7890 gas chromatography coupled with a mass spectrometer 5975C model (GC-MS). A DB-5ms capillary column (5 % methyl-phenyl siloxane, 30 m in length × 0.250 mm i.d. × 0.25 µm film) was used with helium as carrier gas (1.0 mL /min^{-1}). The mass spectrometer system operated with 70 eV of energy and full scan mode (m/z 50-550). For PAHs quantification, a standard mixture containing 16 priority PAHs was used to prepare calibration solutions in the concentration range of 5 to 1000 ng mL^{-1}. Sterols quantification was performed using 6-level analytical curve (0.5 to 50 µg mL^{-1}) with calibration standards including 5β-cholestan-3β-ol (coprostanol); 5α-cholestan-3β-ol (epi-coprostanol); 5α-cholestan-3-one (coprostanone); cholest-5-en-3β-ol (cholesterol); 5α-androstan-3β-ol; (24S)-24-ethylcholesta-5,22-dien-3β-ol (stigmasterol); 24-α-ethylcholestan-3β-ol (stigmastanol); Cholesta-5,24-dien-3β-ol (desmosterol) and lanosterol. All compounds were identified by comparing their mass spectra, retention time, and ion fragmentation to lipids found in the NIST library and literature. The sediment was considered contaminated by sewage when coprostanol concentrations were > 0.3 µg g^{-1} (Araújo et al. 2021).

Quality assurance and control tests (QA/QC) procedures were achieved in all steps. One blank sample was included for each batch of analyses. Surrogates (100 ng p-terphenyl d-14 and 5 µg androstanol) were added to the samples previously to extraction procedures to verify the extraction efficiency. Recovery values obtained in the range between 70 and 12 0% were considered accepted as indexes of good analytical performance for the method. The limits of detection (LOD) were calculated based on the signal-to-noise ratio (S/N = 3) by dilution of the lowest level of the analytical curve. The limits of quantification (LOQ) were the lowest level of the analytical curve.

We measured the 16 priority pollutant PAHs recommended by the U.S. Environmental Protection Agency by their potential to cause harmful effect on human health and the environment (Naphthalene, Acenaphthylene, Acenaphthene, Fluorene, Phenanthrene, Anthracene, Fluoranthene, Pyrene, Benz(a)anthracene, Chrysene, Benz(b)fluoranthene, Benzo(k)fluoranthene, Benz(a)pyrene, Indeno(1,2,3-cd)pyrene, Dibenz(ah)anthracene, and Benz(ghi)perylene). We determined the pollution levels of PAHs by following the classification proposed by Baumard et al. (1998), for the total 16 PAH: low: 0 – 100 ng/g; moderate: 100 – 1,100 ng/g; high: 1,000 –5,000 ng/g; and very high > 5,000 ng/g.

Microplastics (MPs) in seawater and sediments

Seawater samples for analysing microplastic particles (MPs) were filtered through 1.2 μm glass microfiber membranes (©Whatman GF/C, 47 mm) previously calcinated at 450 °C for 2 hours. Each membrane was placed into glass Petri dishes, covered and dried at 40 °C for 24 hours and then stored in a desiccator until MPs analyses. The filters were then observed in a stereoscopic microscope equipped with a digital camera (Moticam Pro 252 A) and the MPs up to 5mm were counted by a complete sweep on the filters, which were expressed as microplastics per liter (MPs L^{-1}).

Microplastics were extracted from sediment samples using a methodology adapted fromZhang et al. (2019). To achieve a constant weight, the samples were transferred to aluminum containers and dried in an oven at a controlled temperature of 70°C for a duration ranging from 24 to 72 hours. To ensure precision in the measurements, approximately 100 g of each sample was weighed using a calibrated precision scale (Bel Engineering®) and then transferred to properly labeled beakers. Each beaker was then filled with 700 ml of concentrated saline solution (NaCl: 140 $g.L^{-1}$). The mixture was manually stirred with a glass rod for 2 minutes to ensure homogenization of the solution. After 24 hours at room temperature, the overlying water was filtered using a vacuum pump and fiberglass membrane filters with 1.2 μm pores to retain the microplastics (MPs). Each filter was carefully placed in glass Petri dishes and covered with aluminum foil to prevent contamination. The filters were then dried in an oven at 40°C for 24 hours to facilitate subsequent analysis of the MPs.

Identification of the MPs was performed using a stereoscopic microscope (Bel Photonics®) equipped with a digital camera. The results are presented as $MPs.g^{-1}$ (microplastic particles per gram of dry sediment), considering the weight of the sampled material at each sampling station. To ensure data accuracy, blank tests were carried out for all procedures. Moreover, the distilled water used during the experiments was previously filtered through a 0.8 μm membrane filter, and all equipment was rinsed three times before use. Throughout the experiments, 100% cotton lab coats were worn to prevent potential contaminations.

Statistical analyses

Differences among sites, protection categories (multiple-use and no-take MPAs), and coasts (leeward and windward) were tested for multiple descriptors: (1) Isotopic ratio, macroalgae biomass, and fish richness and abundance (as response variables), using permutational ANOVA followed by post hoc Tukey test; (2) Community similarity using analysis of similarity and nonmetric multidimensional scaling (NMDS) for benthic cover (ANOSIM on arcsin square-root transformed percent cover data; Euclidean distance and 999 permutations) and fish assemblages (ANOSIM on square-root transformed biomass data; Bray-Curtis distance and 999 permutations); (3) Species as indicator of sites, protection categories and coasts with an Indicator Species Analysis for the benthic community, macroalgae biomass data and fish assemblages. To perform the analyses for fish communities, we excluded extreme values that could introduce bias to the results, corresponding to two specific cases: the highly abundant *Harengula* sp. and large *Negaprion bevirostris*.

The availability of macroalgae species varied among the sites, but the isotopic signatures were congruent within the sites regardless of the species (Figures S1 and S2). For this reason, samples from all species were pooled as replicates for statistical analyses of isotopic and C:N ratios. We did not perform statistical tests to the metals, microplastic, sterols and HPA data due to the insufficient number of replicates.

To assess the importance of different pollution parameters and site structure on each site (complexity, depth, nutrients, water microplastics, sediment microplastics, HPAs, sterols, $\delta^{15}N$), we conducted a Principal Component Analysis (PCA). The data were standardized using mean normalization, and variables with a correlation coefficient exceeding 0.6 were excluded from the analysis (depth, nitrite, and coprostanol).

To assess the influence of habitat structure and pollution parameters, we employed multiple regression analysis and developed generalized linear models. Model selection was conducted using the Akaike Information Criterion (AIC). To summarize the benthic community data, we conducted a Principal Coordinates Analysis (PCO) on benthic cover data (square root and Euclidean distance) and fish community abundance data (log(x+1)-transformed and Bray-curtis distance) (Figures S3 and S4). The first two axes, which accounted for a substantial amount of the variation (~64%; Benthos axis 1: 36.06%, Benthos axis 2: 27.97%; Fish axis 1: 32.2%, Fish axis 2: 15.91%), were included as dependent variables.

Additionally, we developed models for various parameters, including total macroalgae biomass, individual benthic groups and species, fish abundance, total fish abundance, total fish richness, and individual fish species. The independent variables considered in these models were habitat complexity, ammonia concentration (mg L^{-1}), orthophosphate concentration (mg L^{-1}), *Caulerpa verticilata* $\delta^{15}N$ (‰), coprostanol (µg g^{-1}), plastic in the water (items L^{-1}) and metals. Regarding the metal data, we also performed a PCO analysis to summarize the information. The first axis from the PCO analysis, which explained a substantial portion of the variation (46.94%), was included in the model as an independent variable. We excluded the parameters distance from the coast, depth, total HPAs, and plastic in the sediments from the models due to collinearity. We assessed collinearity using variance inflation factor (VIF) testing, excluding coefficient values above 3. All statistical analysis were performed using R 2023.0606+421 (R Core Team 2022).

Results

1. Environmental quality

1.1. Nutrients (water)

Water quality was satisfactory in most sampled stations, according to the Brazilian regulations, but there are some issues in the multiple-use area. In marine sites (Table 3), mean concentrations reached 0.036 ± 0.012 mg L^{-1} (mean ± SE) for orthophosphate, and 0.038 ± 0.012 mg L^{-1} for N-ammoniacal. Considering that dissolved orthophosphate concentration was 0.086 mg L^{-1} in Porto and 0.078 mg L^{-1} in Morro de Fora. These concentrations surpass the limits established by the legislation for P-Total. This suggests that if these observations were made for the filtered water, the concentration of P-Total would be even higher.

Table 3: Water quality parameters of the marine sites (saline water) in Fernando de Noronha Archipelago. All concentrations are in mg L^{-1}. When available, standards established by legislation are indicated in parentheses. *Limit available to P-Total. Samples outside the standards are in bold. < LOD: Below the limit of detection.

MPA	Site	Orthophosphate (0.062*)	Nitrite (0.07)	Nitrate (0.40)	N-Amonniacal
multiple-use	Porto	**0.086**	< LOD	< LOD	0.070
	Biboca	0.035	0.001	< LOD	0.050
	Morro de Fora	**0.078**	< LOD	< LOD	0.030
	Boldró	0.025	< LOD	< LOD	0.040
no-take	Laje Dois Irmãos	0.029	< LOD	< LOD	0.080
	Golfinhos	0.025	< LOD	< LOD	0.080
	Sapata	0.021	< LOD	< LOD	0.020
	Cabras	0.027	< LOD	< LOD	0.010
	Caieras	0.021	< LOD	< LOD	0.020
	Atalaia	0.058	< LOD	< LOD	0.020
	Frade	0.021	< LOD	< LOD	0.020
	Trinta reis	0.016	< LOD	< LOD	0.030
	Sueste	0.029	< LOD	< LOD	0.020
	Total mean ± SE	**0.036 ± 0.012**	**< LOD**	**< LOD**	**0.038 ± 0.012**

Sancho river presented very high orthophosphate concentrations that surpassed the limits established by the legislation for P-Total (6.77 mg L^{-1}), but the concentrations of nitrite, nitrate and N-ammoniacal in this river were low. The concentrations of dissolved nutrients in Boldró river were above the limits established by the legislation for orthophosphate (1.65 mg L^{-1}) and N-ammoniacal (2.26 mg L^{-1}). Additionally, nitrite concentrations in this river were close to the legal limits (0.18 mg L^{-1}). The effluent from wastewater treatment plants (WTP) Biboca presented N-ammoniacal concentration (64.4 mg L^{-1}) more than three times above the limit established by the legislation. Orthophosphate concentrations in the effluents of WTP Biboca and WTP Boldró were high, although there is no limit established by the legislation (Table 4).

Table 4: Water quality parameters of the watercourses in Fernando de Noronha Archipelago. All concentrations are in mg L^{-1}. Watercourse classes are defined according to CONAMA 357/2005. When available, standards established by legislation are indicated in parentheses. *Limit available to P-Total. Samples outside the standards are in bold. WTP: wastewater treatment plant. < LOD: Below the limit of detection.

Site	Class	Orthophosphate	Nitrite	Nitrate	N-Amonniacal
Sancho River	freshwater, class 1	**6.77** (0.15*)	< LOD (1)	< LOD (10)	0.13 (3.7)
Boldró River	brackish, class 2	**1.65** (0.186*)	0.18 (0.2)	0.33 (0.7)	**2.26 (0.7)**
WTP Biboca	effluent	22.82	0.002	< LQ	**64.40 (20)**
WTP Boldró	effluent	8.27	0.019	0.020	16.70 (20)

1.2. Macroalgae isotopic and elemental signature

We collected a total of 65 samples of 12 green and brown macroalgae species (Table 1). The lowest value of $\delta^{15}N$ was found in *Stypopodium zonale* in Caieras (0.75 ‰) and the highest value was found in *Dictyopteris delicatula* in Biboca (9.96 ‰). The isotopic signature of $\delta^{15}N$ in the macroalgae differed between sites (Figure 2a, ANOVA F= 276.34, p< 0.0001), protection categories (multiple-use area = 6.15 ± 2.23 ‰, no-take area = 2.24 ± 0.76 ‰; F= 444.84, p< 0.0001) and coasts (leeward coast = 5.38 ± 0.20, windward coast = 1.95 ± 0.05 ‰; F= 254.98, p < 0.0001). The effect of protection categories on $\delta^{15}N$ was preeminent. Values were all above the background levels in the multiple-use area sites, indicating anthropogenic input of N, and reached a mean of 9.05 ‰ at Biboca, demonstrating strong exposure to sewage discharge. The no-take sites located on the leeward coast presented higher $\delta^{15}N$ than the no-take sites on the windward coast, with values indicative of nutrient enrichment at Golfinhos.

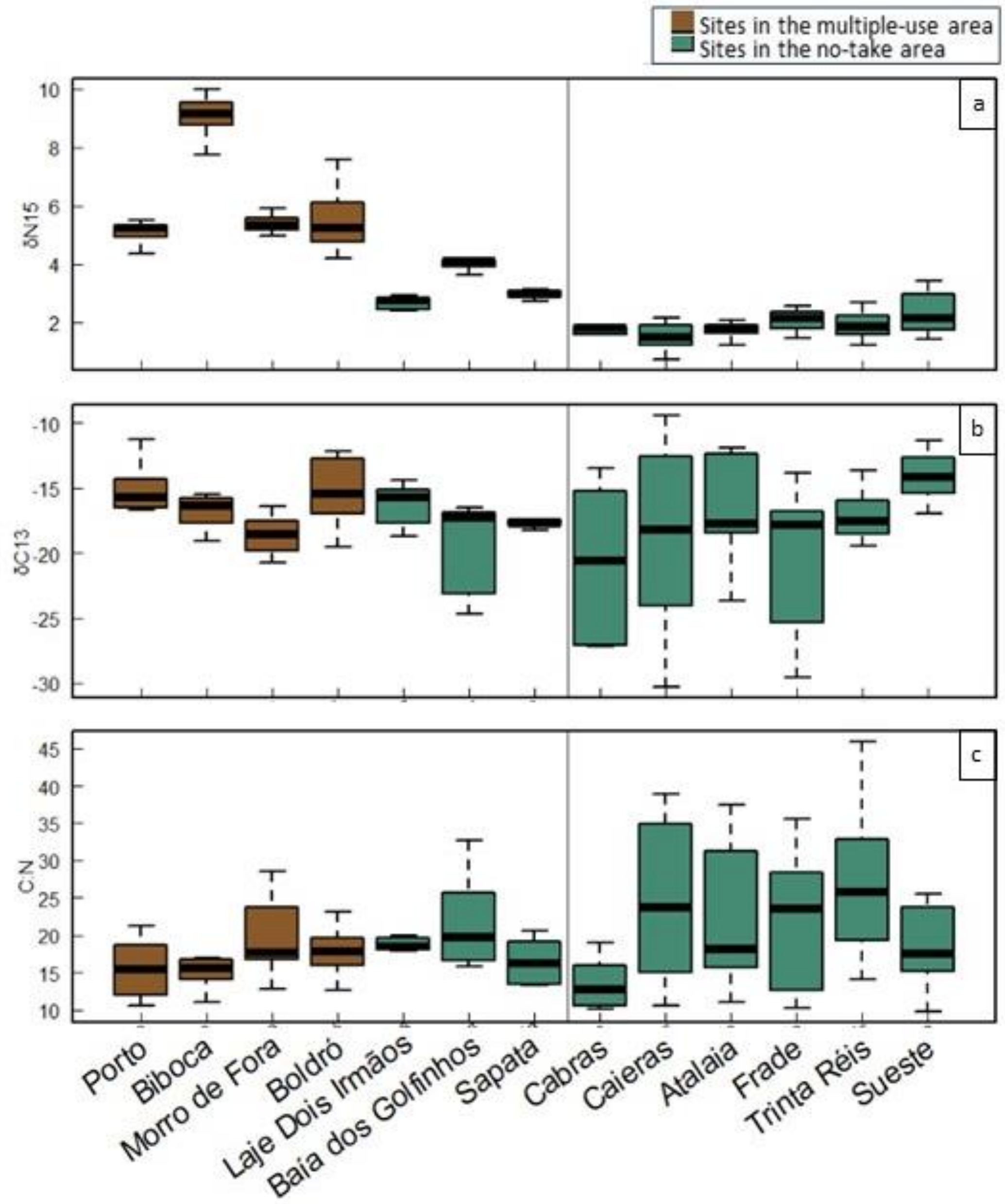


Fig. 2: Elemental composition of macroalgae in Fernando de Noronha Archipelago, (a) δ^{15}N, (b) δ^{13}C, (c) C: N. Left panels: leeward coast, right panels: windward coast. The middle thick lines represent the medians, the boxes represent the 1st and 3rd quartiles, and the whiskers represent the maximum and minimum values.

The isotopic signature of δ^{13}C differed among sites (Figure 2b, $F= 3.15$, $p< 0.001$), but did not differ between protection categories (multiple-use area = -16,28 ± 0,43 ‰, no-take area = -17,69± 0,45 ‰; $F = 4.50$, $p = 0.03$) and coasts (leeward coast = -16.72± 0.35, windward coast = -17.71 ± 0.56 ‰; $F = 1.92$, $p =0.16$). C:N ratio differed among sites (Figure 2c, $F= 5.22$, $p< 0.0001$), protection categories (multiple-use area = 17.26 ± 0.45, no-take area = 21.66 ± 0.85; $F=15.60$, $p< 0.001$) and coasts; (leeward coast = 17.72±

0.41, windward coast = 22.45 ± 1.05, F= 18.94, $p < 0.001$). The values in general indicate oligotrophic environments, but the sites in the multiple-use area and the leeward coast exhibit reduced N-limitation to macroalgae growth compared to the no-take area and the windward coast.

1.3. Metals and metalloid (sediment)

Among the quantified metals and metalloid (As), mean concentrations for major and minor elements in sediments followed the order Fe > Al > Cr > Ni > Zn > As > Cu (Fig. 3). Cadmium and Pb concentrations were below the limit of quantification in all the samples. Baía dos Golfinhos (no-take area) presented the highest concentrations for most detected metals and metalloid (Fe, Al, Ni, Cr, As, Cu), and some of them occurred above the threshold effect level TEL conditions (e.g., 61.3 $mg.kg^{-1}$ for Ni, 59.5 $mg.kg^{-1}$ for Cr and 12.1 $mg.kg^{-1}$ for As; Fig. 3). Arsenic concentrations were also higher than the TEL in the multiple-use sites Biboca (8.3 $mg.kg^{-1}$), Morro de Fora (9.5 $mg.kg^{-1}$) and Boldró River (8.4 $mg.kg^{-1}$). Observed metal concentrations were below the probable effect level (PEL) for all metals, metalloid, and sites.

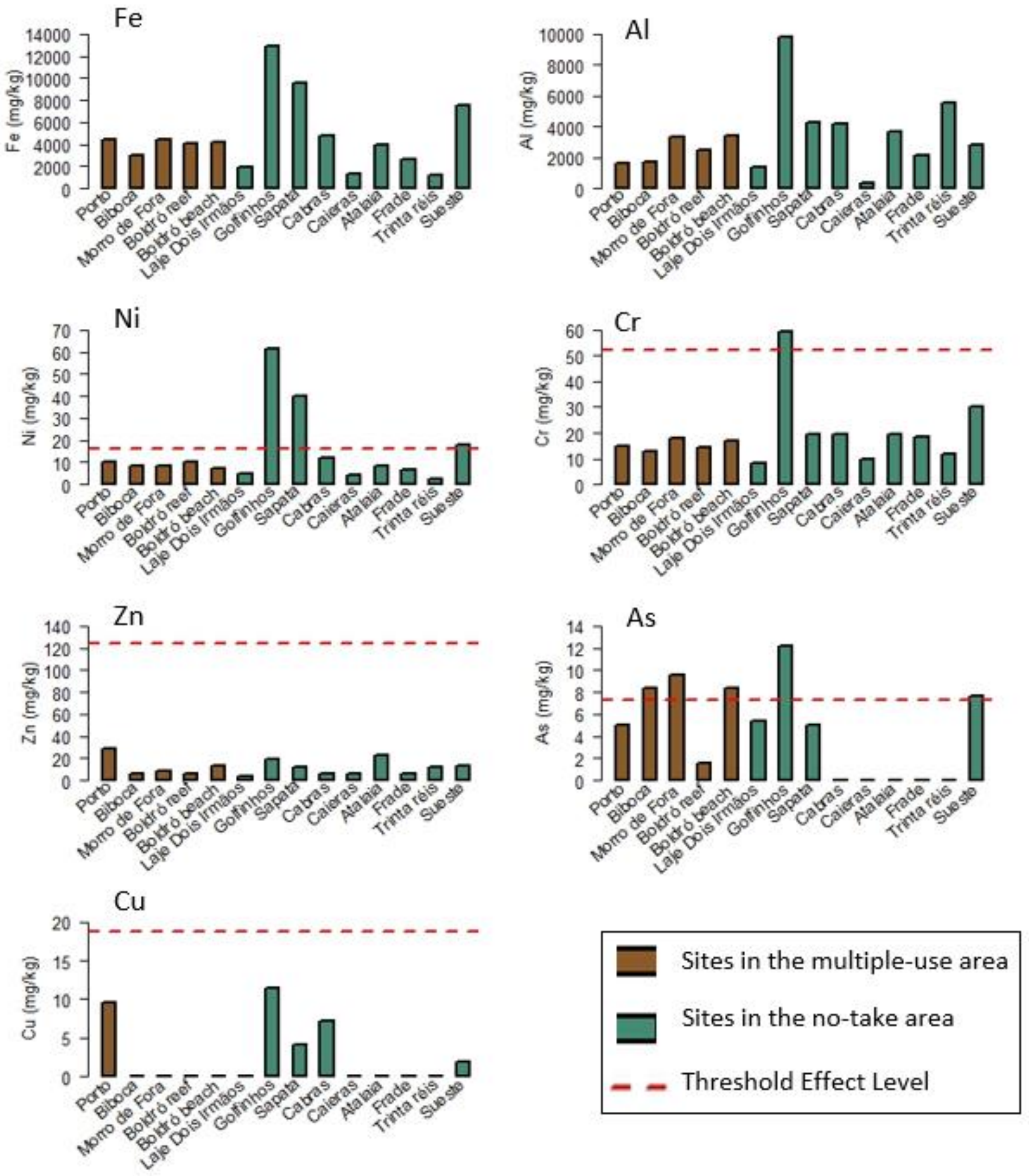


Figure 3: Concentration of Fe, Al, Ni, Cr, Zn, As, and Cu (mg/kg) in sediments of the multiple-use and no-take areas in Fernando de Noronha Archipelago. The scales vary according to the maximum concentration observed for each metal. For the metals with established Threshold Effect Limit (TEL) guidelines, the TEL is indicated by the red dotted line. For samples below the minimum limit of quantification, the values are indicated as zero.

1.4. Fecal sterols and PAHs

The total fecal sterol concentrations in marine sediments from Fernando de Noronha ranged from 0.238 to 1.128 μg g^{-1}, while coprostanol concentrations ranged from 0.103 to 0.288 μg g^{-1}, indicating no sewage contamination in the sediment of most sites. However, the concentration of coprostanol in Porto was two times higher than the maximum value observed at the other sites and closer to values indicative of contamination (0.3 µg g^{-1}). Boldró River, on the other hand, had a higher concentration of coprostanol (0.436 µg g^{-1}), indicating contamination by domestic sewage (Table 5).

The mean of the sum of the concentrations of the 16 PAHs ranged from 16.51 to 1113.58 ng g^{-1}, with the highest value recorded at Porto. However, total PAHs in Porto exhibited great variation between samples (total PAHs in the replicates of Porto were 35.18, 99.67, and 3205.89 ng g^{-1}), attesting a point contamination which will be discussed further. Moderate levels of pollution were observed at Sueste and Boldró River, while all other sites had low levels of PAHs pollution (Table 5).

Table 5: Mean values of fecal sterols (coprostanol, epi-coprostanol, cholesterol, cholestanol, and coprostanone) and total polycyclic aromatic hydrocarbons (PAHs) concentrations in marine sediments in Fernando de Noronha Archipelago. Sterol concentrations are in µg g^{-1} and PAH concentrations are in ng g^{-1}. *River sample.

MPA	Site	Coprostanol	Total fecal sterols	Total PAH
multiple-use	Porto	0.288	1.128	1113.58
	Biboca	0.106	0.301	22.14
	Morro de Fora	0.112	0.601	49.46
	Boldró	0.104	0.238	26.57
no-take	Laje Dois Irmãos	0.107	0.420	19.82
	Golfinhos	0.107	0.552	22.01
	Sapata	0.117	0.632	23.19
	Cabras	0.107	1.187	17.34
	Caieras	0.103	0.321	16.75
	Atalaia	0.114	0.505	57.48
	Frade	0.108	0.534	16.51
	Trinta reis	0.106	0.618	22.63
	Sueste	0.114	1.027	126.28
	Total mean ± SE	**0.133 ± 0.017**	**0.632 ± 0.109**	**185.67 ±144.22**
	*Boldró river	0.436	0.701	107.46

1.5. Microplastics

Microplastics were present in all the samples analysed (Fig. 4). The highest concentration of microplastics in the sediment was found in Golfinhos (0.39 items/g) and the highest concentration in the water was found in Cabras (26 items/L) (Fig. 4). The mean concentration of microplastics in each coast and MPA is shown on Table 6.

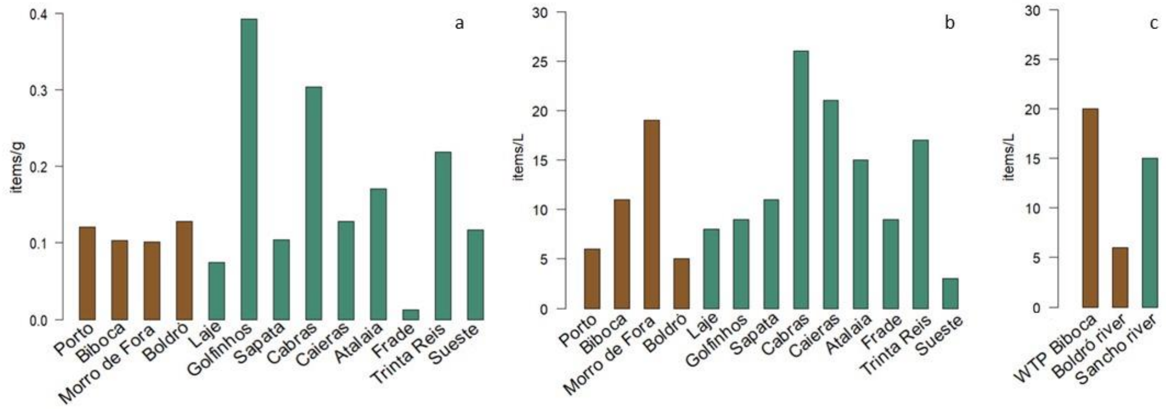


Figure 4: Microplastic concentration in (a) sediment, (b) seawater and (c) terrestrial watercourses in Fernando de Noronha.

Table 6: Mean concentrations of microplastics in the sediment (items/g) and water (items/L) of Fernando de Noronha. MPA = marine protected area.

	Sediment (items/g)	***Water (items/L)***
Coast		
leeward	0.13	9.9
windward	0.15	15.2
MPA		
multiple-use	0.13	9.8
no-take	0.14	13.9

1.6. Summary of the environmental quality parameters

The principal component analysis (PCA) (Fig. 5) revealed distinct patterns of pollution and isotopic signature of macroalgae among different sites in the study area. Specifically, the sites Porto, Biboca, Boldró, and Morro de Fora within the multiple-use area exhibited higher levels of coprostanol, orthophosphate and greater enrichment of $\delta^{15}N$. Porto stood out as the most polluted site, particularly in terms of coprostanol.

The sites Laje Dois Irmãos and Golfinhos had higher concentrations of N-ammoniacal, and Golfinhos showed more contamination in terms of microplastic in sediments. Within the windward sites of the no-take area, microplastics in the water emerged as the primary pollutant, particularly in Caieras, Frade, Cabras, and Trinta-Réis.

In comparison, the sites Atalaia and Sueste exhibited higher influence from orthophosphate compared to other windward coast sites. Sapata, located in the leeward coast of the no-take area, displayed similarities to the windward coast sites of the no-take area in terms of pollution characteristics. Despite complexity and depth being different among the sites, these factors are secondary or correlated in the determination of groupings in relation to water and sediment quality measurements.

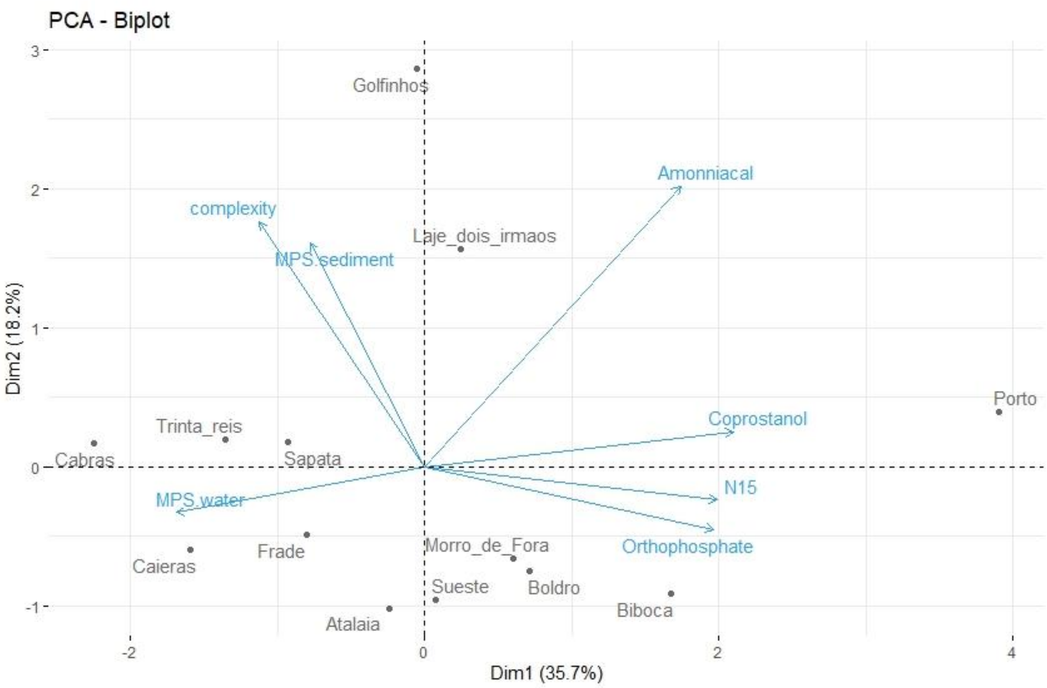


Figure 5: Principal component analysis (PCA) on the pollution and structure parameters of the sampling sites. MPS: microplastics.

2. *Community structure*

2.1. Benthic cover

We recorded 34 taxonomic groups of benthic organisms, including 4 coral and 13 macroalgae species. The benthic communities differed among sites (ANOSIM $R = 0.6853$, $p < 0.0001$), between protection categories ($R=0.4689$, $p < 0.0001$, Fig. 6) and coasts ($R= 0.0843$, $p < 0.01$. Fig. 6). In the NMDS the samples from the multiple-use sites separated from the samples within the no-take area. Specifically, the sites Porto and Biboca formed a cohesive group, while Boldró formed a distinct group on its own.

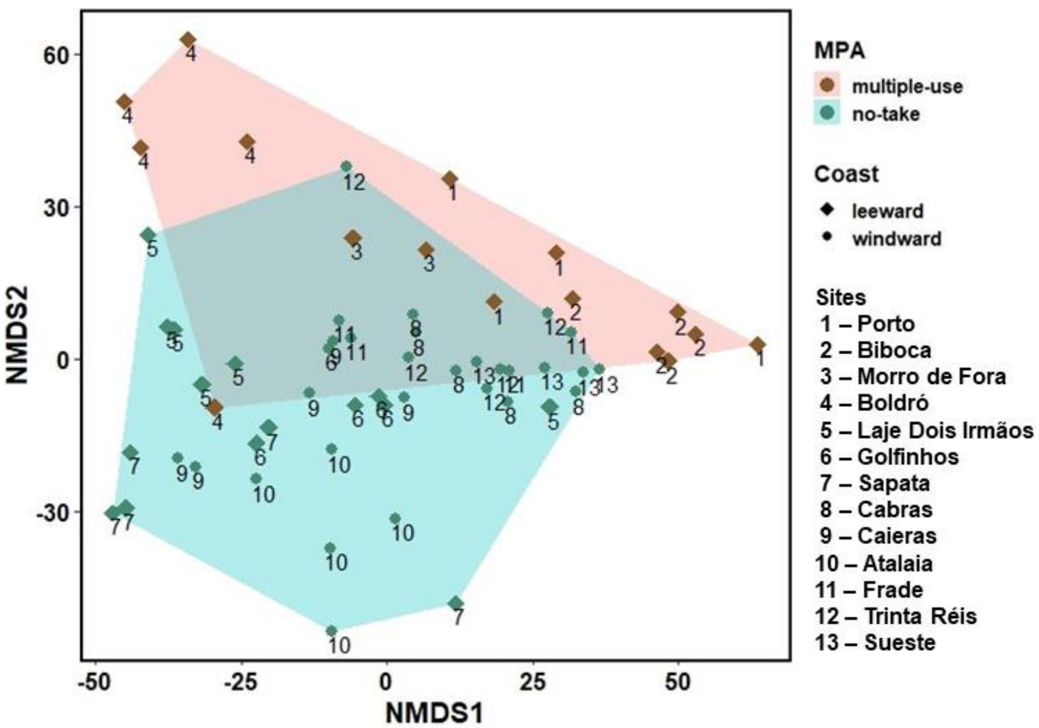


Figure 6: Multidimensional Scaling (MDS) of the benthic cover among sites in the Fernando de Noronha archipelago. Stress= 0.09.

Brown macroalgae (i.e. *Canistrocarpus* spp., *Dyctiota* spp., and *Dyctiopteris* spp.), algal turf and the green algae *Caulerpa verticilata* were the most important benthic groups in the archipelago (Fig. 7), with brown macroalgae having outstanding importance at Biboca, *C. verticilata* at Boldró, and algal turf at Sapata (~75%, 65% and 62% cover respectively). The brown algae *Spatoglossum schoroederi* was exclusive to Porto and Biboca (both ~14% cover), and *Bryopsis* sp. to Biboca (~2% cover). The coral *Montastrea cavernosa* was associated with Laje and Sapata (~12% and 14%), filamentous red algae were associated with Atalaia (~25%), and *Padina gymnospora* was associated with Trinta Réis (~3% cover) (indicator species analysis, $p < 0.001$) (Fig. 7).

Regards protection categories, in both multiple-use and no-take areas the most abundant groups were brown macroalgae (43% multiple-use area, 38% no-take area), *C. verticilata* (29% and 9%) and algal turf (10% and 33%). The species selected as an indicator for the multiple-use area were *S. schoroederi* ($p < 0.001$), *C. verticilata* ($p < 0.021$) and *Bryopsis* sp. ($p < 0.05$). For the no-take area, the indicators were algal turf, cyanobacteria (p < 0.001), *P. gymnospora*, *Montastrea cavernosa*, *Sargassum* sp., *Siderastrea stellata* and *Stypopodium zonale* (p < 0.05).

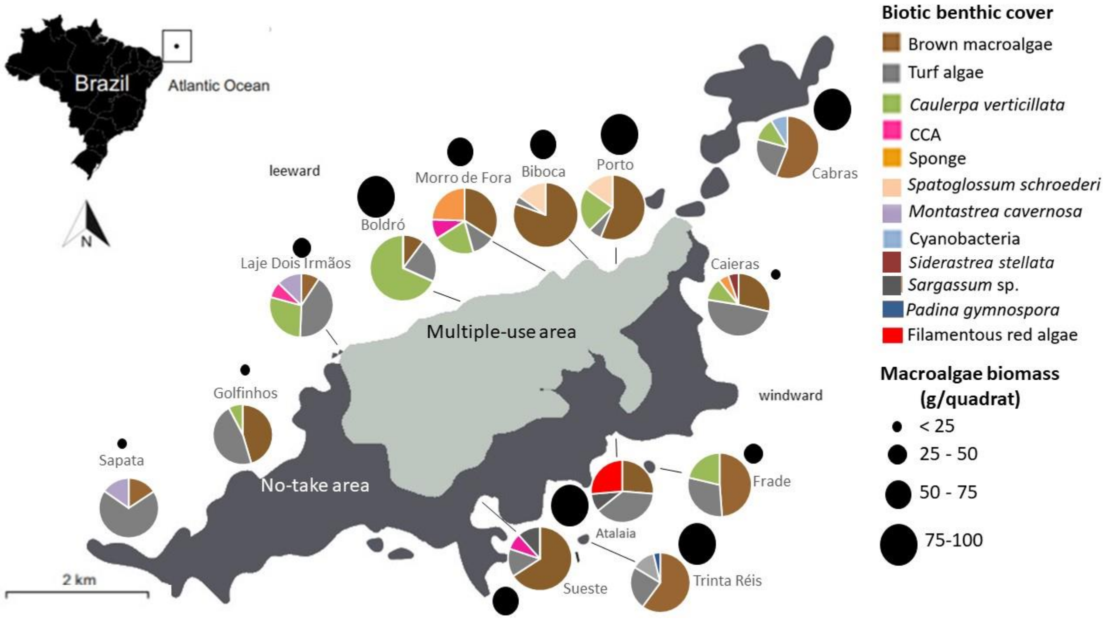


Figure 7: Proportion of cover of the main benthic groups (groups that account for more than 90% of coverage) and macroalgae biomass in the reefs of Fernando de Noronha archipelago.

2.2. Macroalgae biomass

A total of 33 macroalgae taxa were recorded in biomass surveys, 24 of which were identified at the species level. Mean macroalgae biomass per quadrat was 57.04 ± 7.5 g, and both biomass and macroalgae community differed among sites (Fig. 7; ANOVA F = 2.72, p = 0.014; ANOSIM, R = 0.7192, p < 0.001). Biomass variation among sites was mainly due to Atalaia, which had a significantly higher biomass than Golfinhos, Sapata and Caieras (Tukey test, p < 0.05). Biomass was not significantly different between protection categories (multiple-use area 71.20 ± 12.93 g, no-take area 46.42 ± 8.48 g,; F = 0.59, p = 0.445), but the composition of the algae communities differed between them (R = 0.1842, p < 0.001). There was also no biomass difference between coasts, but communities were significantly different (Leeward coast 56.94 ± 10.99 g, windward coast 57.18 ± 9.97 g, F= 2.98, p = 0.092; R = 0.2434, p < 0.001). In the indicator species analysis, the green algae *Caulerpa verticilata* was associated with Boldró and the brown algae *Dictyopteris ciliolata* with Baía dos Golfinhos. The coralline red algae *Amphiroa* sp. was also strongly associated with Atalaia (Table 7).

Table 7: List of macroalgae taxa associated with each site or group of sites in indicator species analysis of macroalgae biomass in Fernando de Noronha Archipelago. Significance codes: '***' 0.001, '**' 0.01, '*' 0.05.

Sites or groups of sites	**Indicator species**	**p-value**
Boldró	*Caulerpa verticilata*	*
Golfinhos	*Dictyopteris ciliolata*	**
Atalaia	*Amphiroa* sp.	**
	Dictyosphaeria versluysii	*
	Laurencia sp.	*
Porto + Biboca	*Spatoglossum schroederi*	***
	Dictyopteris delicatula	***
Laje Dois Irmãos + Golfinhos	Articulated calcareous	**
Cabras + Sueste	*Canistrocarpus cervicornis*	**
Trinta Réis + Sueste	*Dictyopteris jamaicensis*	***
Leeward coast	*Dictyopteris delicatula*	**
	Articulated calcareous	*
Windward coast	*Padina gymnospora*	***
	Canistrocarpus cervicornis	**
	Dictyopteris jamaicensis	**
Multiple-use area	*Dictyopteris delicatula*	**
	Caulerpa verticilata	**
	Filamentous green	**
	Dictyopteris jolyana	**
	Spatoglossum schroederi	*
	Sargassum vulgare	*
	Filamentous red	*
No-take area	*Canistrocarpus cervicornis*	*
	Padina gymnospora	*

2.3. Fish communities

We recorded 56 fish species of 31 families and a total of 5,405 individuals, with an average of 83 ± 6 individuals per census. The average species richness and fish biomass per census were 9.49 ± 0.35 and 9.17 ± 1.74 kg, respectively. The 20 most abundant species and the 20 with the highest biomass recorded are listed in Table 8.

Table 8: Abundance and biomass per census of the main fish species recorded in 13 sites (65 transects) at Fernando de Noronha Archipelago, ranked in decreasing order of abundance.

Abundance rank	Species	Abundance (mean ± SE)	Biomass (g) (mean ± SE)	Biomass rank
1	*Thalassoma noronhanum*	29.98±2.93	95.88±13.04	19
2	*Stegastes rocasensis*	9.88±1.37	120.11±19.12	17
3	*Azurina multilineata*	9.68±1.91	145.06±27.66	15
4	*Harengula spp*	5.38±3.82	2.58±2.05	43
5	*Acanthurus chirurgus*	4.49±0.94	401.14±102.08	9
6	*Cephalopholis fulva*	4.42±1.12	851.99±231.72	3
7	*Haemulon chrysargyreum*	3.65±1.83	485±323.42	7
8	*Halichoeres radiatus*	2.48±0.50	96.33±21.73	18
9	*Malacoctenus lianae*	2.20±0.44	1.51±0.31	46
10	*Melichthys niger*	1.77±0.54	251.03±67.62	11
11	*Abudefduf saxatilis*	1.29±0.43	94.29±33.78	20
12	*Haemulon parra*	1.17±0.48	187.19±89.52	13
13	*Holocentrus adscensionis*	0.97±0.38	379.37±142.98	10
14	*Sparisoma axillare*	0.65±0.15	504.98±201.83	6
15	*Sparisoma frondosum*	0.65±0.15	464.32±105.01	8
16	*Sparisoma amplum*	0.43±0.11	1242.06±350.22	2
17	*Stegastes pictus*	0.42±0.25	3.00±1.86	42
18	*Hemiramphus brasiliensis*	0.38±0.24	1.51±1.03	46
19	*Myripristis jacobus*	0.31±0.15	96.93±52.82	22
20	*Mulloidichthys martinicus*	0.28±0.16	61.24±33.47	24
22	*Caranx bartholomaei*	0.22±0.08	185.06±72.33	14
26	*Kyphosus sectatrix*	0.17±0.08	542.38±350.11	5
29	*Lutjanus jocu*	0.12±0.05	202.38±97.62	12
31	*Sphyraena barracuda*	0.12±0.06	745.22±354.47	4
43	*Negaprion brevirostris*	0.03±0.02	1428.11±1227.31	1
51	*Dermatolepis inermis*	0.02±0.02	127.58±127.58	16

Fish richness varied among sites (Fig. 8a; $F = 5.64$, $p < 0.001$, Table 9), but not between protection categories (multiple-use area 8.74 ± 0.59, no-take area 9.8 ± 0.43; $F = 1.93$, $p = 0.17$) or coasts (leeward coast 10.06 ± 0.69, windward coast 8.93 ± 0.38; $F = 2.60$, $p = 0.112$).

Table 9: Tukey test results for differences in fish richness between sites in Fernando de Noronha Archipelago. The listed sites are significantly different, p-value indicates significance level. Significance codes: '***' 0.001, '**' 0.01, '*' 0.05.

Sites	p-value
Leeward coast	
Laje Dois Irmãos - Porto	**
Laje Dois Irmãos - Biboca	***
Sapata - Biboca	**
Laje Dois Irmãos - Boldró	*
Windward coast	
Sueste - Atalaia	**
Windward x Leeward	
Atalaia - Biboca	*
Frade - Laje Dois Irmãos	**
Sueste - Laje Dois Irmãos	***
Sueste - Sapata	**

Fish abundance varied among sites (Fig. 8b; $F=3.48$, $p<0.001$), mainly because the site Golfinhos was different from Boldró, Caieras and Frade, and because Frade was different from Atalaia and Laje (Tukey test, $p < 0.05$). The fish abundance was significantly higher in the no-take area when compared to the multiple-use area (83.85 ± 5.78 and 62.95 ± 4.76, respectively; $F = 4.80$, $p < 0.05$), and there was no difference between coasts (leeward coast 81.03 ± 5.82, windward coast 74.57 ± 6.79; $F = 0.51$, $p = 0.4$). Fish biomass was different among sites (Fig. 8c.; $F = 2.30$, $p < 0.05$) and coasts (leeward coast 9.59 ± 1.35 kg, windward coast 5.90 ± 1.00; $F = 4.80$, $p < 0.05$), but not between protection categories (multiple-use area 7.01 ± 1.70 kg, no-take area 8.01 ± 1.0 kg; $F = 0.27$, $p = 0.60$).

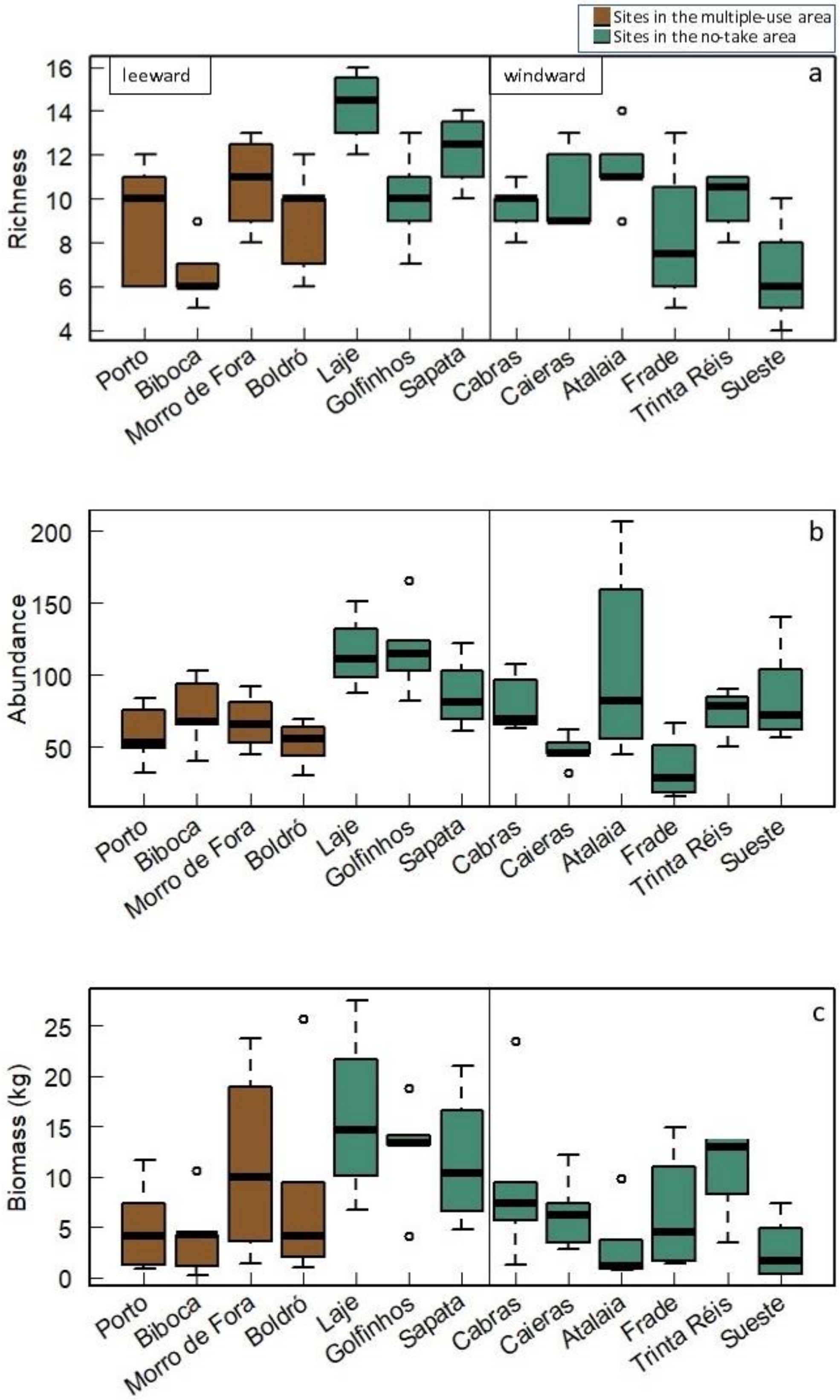


Figure 8: Fish species richness (a), abundance (b), and biomass (c) in Fernando de Noronha Archipelago. Left panels: leeward coast, right panels: windward coast. The middle thick lines represent the medians, the boxes represent the 1st and 3rd quartiles, and the whiskers represent the maximum and minimum values.

The structure of fish communities exhibited significant differences across sites (ANOSIM $R = 0.475$, $p < 0.0001$), protection categories ($R = 0.19$, $p < 0.01$, Fig. 9), and coasts ($R = 0.15$, $p < 0.001$, Fig. 9). Samples from the multiple-use areas formed a distinct group, except for some samples from Morro de Fora, while the samples from the shallow reefs within the no-take area (Atalaia and Sueste) clustered closer to the multiple-use group. Comparing the sites within the no-take area, those on the leeward coast (Laje Dois Irmãos, Golfinhos, and Sapata) were more similar to the no-take sites on the windward coast than to the multiple-use sites on the leeward coast. This suggests that the no-take areas on different coasts share certain similarities in fish community structure. The Indicator Species Analysis showed *Malacoctenus lianae* as the main indicator of the multiple-use area, and of *Azurina multilineata* to the no-take area. The complete results of indicator species analysis are shown in Table 10.

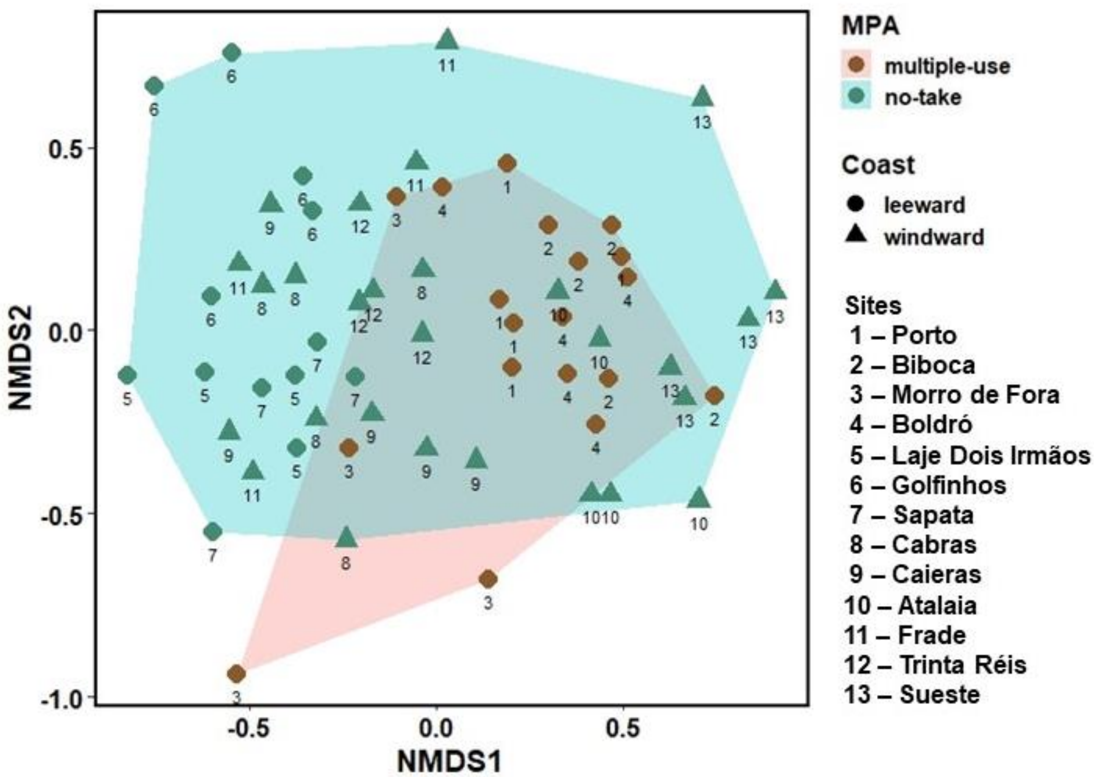


Figure 9: Multidimensional Scaling (MDS) of the fish community among sites in the Fernando de Noronha archipelago. Stress= 0.2243.

Table 10: List of fish taxa associated with each site or group of sites in indicator species analysis of fish communities in Fernando de Noronha Archipelago. Significance codes: '***' 0.001, '**' 0.01, '*' 0.05.

Sites or groups of sites	Indicator species	p-value
Morro de Fora	*Caranx crysos*	*
Laje	*Myripristis jacobus*	***
Golfinhos	*Cephalopholis fulva*	***
	Stegastes pictus	***
Sapata	*Pempheris schomburgkii*	***
	Sparisoma amplum	**
Atalaia	*Caranx latus*	***
Boldró + Morro de Fora	*Pseudopeneus maculatus*	*
Boldró + Atalaia	*Sparisoma axillare*	*
Laje + Sapata	*Elacatinus phthirophagus*	*
Leeward coast	*Pseudopeneus maculatus*	*
	Azurina multilineata	*
	Myripristis jacobus	**
Windward coast	*Abudefduf saxatilis*	*
	Acanthurus chirurgus	*
	Caranx latus	*
Multiple-use area	*Malacoctenus lianae*	***
	Halichoeres radiatus	*
	Pseudupeneus maculatus	*
No-take area	*Azurina multilineata*	**
	Melichthys niger	**
	Ophioblennius trinitatis	*
	Cephalopholis fulva	*

3. Relationship between abiotic and biological parameters

The results of the multiple regression models investigating the relationship between environmental parameters and biological variables are summarized in Table 11. Fish community was influenced by several variables. Fish_PCO1 was primarily determined by positive correlation with *S. amplum, A. multilineata and S. frondosum* and a negative correlation with *T. noronhanum, Lactophrys trigonus,* while *M. lianae*, had a positive correlation with habitat complexity, $\delta^{15}N$, and microplastics in the water, and a negative correlation with orthophosphate and coprostanol concentrations. Fish_PCO2 was primarily determined by negative correlation with *H. chrysargyreum,* and *S. rocasensis* which was positively correlated to N-ammoniacal concentrations.

Habitat complexity was also related to a few fish species, including *M. lianae* and *S. frondosum*. There was a positive correlation between N-ammoniacal concentrations and fish abundance, and to certain fish species like *S. axillare*. Orthophosphate exhibited a positive association with sponge cover and the abundance of *K. sectatrix.* The isotope ^{15}N showed a positive relationship with *Bryopsis* sp. and *S. schroederi,* cover, and was negatively related to *Sargassum* cover and fish richness. Coprostanol concentration exhibited a positive association with *S. schroederi* cover and a negative association with sponge cover. Notably, a strong positive relationship was observed between coprostanol and the abundance of *H. parra.* Metal concentration demonstrated a positive relationship with the abundance of *C. fulva, S. rocasensis* and *S. pictus*, while displaying a negative association with the abundance of *D. inermis.* The concentration of microplastics in the water showed a negative correlation with the abundance of *H. chrysargyreum* and *S. rocasensis*.

Table 11: Multiple regressions evaluating the relationship between environmental parameters and biological variables. Dependent variables include Benthos_PCO1 and Benthos_PCO 2: PCO analysis axis of % benthic cover; total macroalgae biomass; benthic groups and species; Fish_PCO1 and Fish_PCO2: PCO analysis of fish abundance; total fish abundance; total fish richness; fish species. Independent variables: complexity, ammonia, orthophosphate, $\delta^{15}N$, coprostanol, metals, microplastic in the water. R^2 and coefficient t-value (Estimate/Std. error) are shown for each model. Significance codes for the variables included in the models: '***' 0.001, '**' 0.01, '*' 0.05, '.' 0.1, NS – non-significant.

Dependent variable	**R^2**	**Independent variables included in the model**						
		Complexity	**Ammonia**	**Orthophosphate**	**$\delta^{15}N$**	**Coprostanol**	**Metals**	**Plastic**
Benthos								
Benthos_PCO1	0.53	-	-	-	-2.03 .	-	-	-
Benthos_PCO2	-	-	-	-	-	-	-	-
Macrolgae biomass	-	-	-	-	-	-	-	-
Brown macroalgae	0.39	-	-	-	1.94 .	-	-	-
Bryopsis sp.	0.77	-2.11 .	1.16 NS	-	3.08*	-1.63 NS	-	1.62 NS
CCA	-	-	-	-	-	-	-	-
C. verticilata	-	-	-	-	-	-	-	-
Cyanobacteria	-	-	-	-	-	-	-	-
M. cavernosa	0.79	-	2.31 .	-	-	-	-2.19 .	-
P. gymnospora	-	-	-	-	-	-	-	-
Sargassum sp.	0.74	-	-1.14 NS	-	-2.53 *	-	1.29 NS	-1.79 NS
S. schroederi	0.88	-2.21 .	1.30 NS	-	3.36 *	3.04 *	-	1.80 NS
S. stellata	-	-	-	-	-	-	-	-
Sponge	0.66	-	-	3.66 **	-	-2.88 *	-	-
S. zonale	0.59	-	-	-	-	-	-2.20 .	-
Algal turf	-	-	-	-	-	-	-	-
Fish								
Fish_PCO1	0.95	6.87 ***	-	-3.71 **	2.53 *	-8.72 **	2.19 .	6.45 ***
Fish_PCO2	0.53	-	3.21 *	-	-	-	-	-
Fish richness	0.82	-	2.42 .	1.44 NS	-2.73 *	-1.33 NS	-2.06 .	-

Dependent variable	R^2	Independent variables included in the model						
		Complexity	Ammonia	Orthophosphate	$\delta^{15}N$	Coprostanol	Metals	Plastic
Fish								
Fish abundance	0.70	-	2.97 *	-	-2.22 .	-1.43 NS	1.54 NS	-
A. multilineata	0.89	2.33 NS	3.70 *	-2.60 *	-2.10 .	-	-	1.46 NS
A. chirurgus	-	-	-	-	-	-	-	-
C. bartholomaei	-	-	-	-	-	-	-	-
C. fulva	0.78	-	3.34 *	-2.43 *	-	-	3.31 *	1.67 NS
D. inermes	0.43	-	-	-	-	-	-2.73 *	-
M. jacobus	0.79	-	2.28 .	-	-	-	-2.19 .	-
M. lianae	0.81	-4.59 **	-	-	3.09 *	-	-	-
M. martinicus	0.54	-	2.83 *	-1.73 NS	-	-	-	1.53 NS
M. niger	-	-	-	-	-	-	-	-
H. adscensionis	0.78	-	3.37 *	-2.58 *	-	-	3.32 *	1.73 NS
H. chrysargyreum	0.36	-	-	-	-	-	-	2.39 *
H. parra	0.98	-	-2.14 .	2.50 .	-1.79 NS	12.52 ***	-	-
H. radiatus	-	-	-	-	-	-	-	-
K. sectatrix	0.76	-2.65 .	-1.39 NS	5.01 **	-	-	-	-
L. jocu	-	-	-	-	-	-	-	-
S. amplum	0.77	2.05 .	2.16 .	-2.35 .	-	-	-	-
S. axillare	0.69	-	3.89 **	-	-	-1.42 NS	-2.21 .	-
S. frondosum	0.69	2.71 *	-	2.61 *	1.07 NS	-2.16 .	-1.18 NS	-
S. barracuda	-	-	-	-	-	-	-	-
S. rocasensis	0.77	-	1.65 NS	-	-	2.73*	2.70 *	3.52 **
S. pictus	0.72	-	2.73 *	-1.72 NS	-	-	3.13 *	1.22 NS
T. noronhanum	0.77	-2.79 *	-1.25 NS	-	-1.94 .	-	-	-1.94 .

Discussion

Sources, distribution, and levels of pollution

We detected the presence of marine pollution in Fernando de Noronha Archipelago and documented ecological responses through changes in the structure of biological communities (*i.e.* composition and abundance). Pollution was more pronounced in the multiple-use areas but it was also observed in the no-take area. Our observations are consistent with the coastal circulation models for the archipelago (Oliveira 2018), suggesting that on the leeward coast, pollutants discharged into the multiple-use area are transported by currents towards the no-take area, and eventually accumulate in calmer waters like those of Baía dos Golfinhos. On the windward coast, pollutants primarily result from long-range transport facilitated by superficial marine currents. On the leeward coast of the archipelago, the wastewater discharged from WTP Biboca and Boldró did not meet the standards established by Brazilian legislation, indicating potential inefficiencies in the treatment processes. The nutrient levels in the reef water were within acceptable limits in most reefs, which may be due to the role of coastal hydrodynamics in dispersing pollutants or to biological assimilation, transferring these compounds from the environmental to the biotic compartment.

A previous study (Bertini and Braga 2022) documented the nutrient concentrations 3 km off FN in July 2013 and found lower concentrations in comparison to what we found in the present study. The maximum concentration of dissolved orthophosphate found by those authors was 0.08 µmol/L, whereas our maximum point (Porto) was about 10 times higher with a concentration of 0.90 µmol/L. The concentrations of orthophosphate and N-Ammoniacal measured in this study were also much higher than those measured in 2013-2014 (Assunção et al. 2016) in four sites we also sampled (orthophosphate at Porto was 0.32 µmol/L and is now 0.95 µmol/L; N-Ammoniacal at Morro de Fora was 0.08 µmol/L and is now 2.35 µmol/L; and at Laje Dois Irmãos it was 0.01 µmol/L and is now 5.70 µmol/L; data from Assunção et al. 2016 and this study). These results suggest an increase in nutrient concentrations over time, particularly near the islands possibly due to increased terrestrial input. The sites Porto and Morro de Fora present evidences of contamination by phosphate that surpasses legal limits. The low levels of nitrate and the presence of N-ammoniacal indicate recent contamination by effluentes.

Terrestrial nutrient input into marine environments can originate from both anthropogenic and natural sources. Among the natural sources, bird guano stands out as a rich nitrogen contributor due to its uric acid content which is converted into ammonia by bacteria before it enters marine water (Justel-Díez et al. 2023). In areas with abundant seabird populations, the ammonia derived from guano can lead to localized increases in N-ammoniacal levels. The seabird colonies on the cliffs near Laje Dois Irmãos and Golfinhos (Santos and Serafini 2020) can be a source of nutrients to these sites. If this was the case, because guano is richer in phosphorus compared to anthropogenic sources of nutrients (Graham et al. 2018) we would expect to see higher concentrations of orthophosphate at these sites when compared to neighbouring sites, which did not happen. This observation suggests that the nutrients detected at these sites are primarily originating from anthropogenic nitrogen sources, thereby distinguishing them from the phosphorus-rich bird guano. On the other hand, the high orthophosphate concentration observed in the freshwater of Sancho River is probably related to the input of seabird guano from the surrounding nesting area.

On the windward coast, the nutrient levels were generally lower compared to those of the leeward coast, except for orthophosphate in Atalaia, an intertidal reef considered to be more susceptible to human influence compared to deeper reefs (Mello et al. 2023). A small creek originating from one of the most densely populated areas of the island flows into this location, and where the presence of cattle is also frequent, potentially acting as a source of nutrients (Fig. S5).

While water samples offer a snapshot at the time of sampling, the isotopic signature of macroalgae provides a more accurate record of nutrient availability and transfer to the biotic compartment over an extended period ranging from days to weeks (Risk et al. 2009, Alldred et al. 2023). Biological assimilation of the nutrients can occur at different trophic levels, including absorption of nutrients by phytoplankton in the water column, as well as accumulation in benthic organisms in the case of sediment-associated elements (Elser and Urabe 1999). Although the nutrient concentration in the reef water samples was within the legal limits for most samples, the isotopic signature of nitrogen in the macroalgae revealed the biological assimilation of nitrogen from anthropogenic sources throughout the multiple-use area. Biboca displayed the most pronounced enrichment, which matches those reported for strong exposure to sewage discharge (Lapointe et al. 2005, Lapointe et al. 2021, Alldred et al. 2023). The existing wastewater treatment at WTP Biboca proves to be inefficient in removing nitrogen from the effluents, as evident by the substantial nitrogen loading at this site and in the water collected straight from the WTP output over the sea. Lower, but still elevated $\delta^{15}N$ values observed for the sites Porto, Morro de Fora and Boldró indicate the discharge of untreated sewage at these sites, result from stormwater runoff associated with leakages from an overburdened and inefficient sewage system. The dilution and coastal transport of the effluents from Biboca can also contribute to the nitrogen enrichment observed at the downstream sites of the multiple-use and the no-take area. The macroalgae in the no-take area of the leeward coast exhibited enrichment compared to those in the no-take area of the windward coast, particularly in Baía dos Golfinhos.

Analysis of the sediments revealed that the levels of PAHs and fecal sterol coprostanol were generally low, with coprostanol being comparable to other tropical locations with similar population densities (Araújo et al. 2021). The presence of metals such as Cr, Ni, Zn, and Cu below toxic thresholds in most samples can be attributed to the volcanic origin of the Archipelago, which naturally contains high concentrations of these metals (Oliveira et al. 2011, Fabricio Neta et al. 2018). The multiple-use area, particularly Porto, showed higher levels of contaminants, indicating potential untreated sewage and chemical pollution potentially from boats.

The higher levels of Cu and Zn found at Porto compared to nearby sites can be linked to the anchorage area where ships and boats use antifouling paints containing these metals (and previously also arsenic), which can leach into the surrounding waters, particularly in areas of frequent anchorage (van Dam et al. 2011, Kroon et al. 2020). Porto beach is the primary location on the island for vessel maintenance and refurbishment activities (Fig. S6); which lack proper infrastructure and management for the correct disposal of toxic and hazardous chemicals, as well as other waste generated during these operations. This contributes to the release of toxic chemicals and waste into the surrounding waters, posing a significant source of contamination. Although the observed concentrations are below toxic thresholds, chronic exposure to antifoul contaminants can lead to bioaccumulation in marine organisms, with potential impacts extending beyond the immediate area due to sediment transport and water mixing (Kroon et al. 2020). Additionally, it is worrisome to note that higher levels of coprostanol were specifically observed in Porto, in comparison to the other sites, being close to those typically associated with sewage contamination. This indicates that sewage is likely being discharged at this location without adequate treatment either from coastal structures or directly from the boats. Among the sediment samples collected in Porto, one sample represents a significant concern in relation to chemical pollution by PAHs, exhibiting concentrations comparable to those found in highly polluted urban areas (Cavalcante et al. 2009). This status was observed in only one out of the three samples taken, indicating that it may be attributed to a localized contamination event, such as an oil spill, rather than a widespread issue. Nevertheless, this finding underscores the urgency for a comprehensive investigation and monitoring of this site to assess and mitigate potential risks. The small harbor at Porto is responsible for transferring large amounts of diesel fuel for the island's thermoelectric power plant, adding up to over 6.5 million liters per year (Horta et al. 2020), increasing the risk of oil spills (Fig. S7). Insufficient infrastructure for boat refuelling further exacerbates the situation. This combination of activities and inadequate preventive measures increases the risk of environmental contamination and accidents. Port dredging, which happens occasionally at this site, can remobilize contaminated sediments, releasing pollutants into the water and spreading contamination to new areas in the disposal of dredged material (van Dam et al. 2011).

Moderate levels of HPAs were detected not only on Boldró River, which is influenced by effluents from the WTP Boldró, but also in the sediments of Sueste, within the no-take area. This finding is probably related to an oil spill event recorded in 2021 when approximately 1.3 tons of plastic and oil reached the windward coast of the Archipelago (Bastos et al. 2022). The characteristics of Sueste, being a shallow protected bay, likely facilitated the deposition of oil at this site.

Golfinhos displayed higher levels of Ni, Cr, Cu, and As compared to other sites, with concentrations of Ni, Cr, and As exceeding toxic thresholds. Domestic sewage discharge, which is often associated with these metals (Bradl 2005, Prazeres et al. 2012), may explain the accumulation of these metals at Golfinhos, possibly transported there by ocean currents (Oliveira 2018). The resident population of spinner dolphins at Golfinhos may also contribute to the metal input through their feces, as top predators can accumulate and biomagnify pollutants through the food chain (Hauser-Davis et al. 2023). Therefore, the presence of these dolphins can potentially serve as a pathway for the transfer of metals, exacerbating their impact on the surrounding environment. Spinner dolphins in this area have also been reported to have low concentrations of persistent organic pollutants and emerging contaminants (Combi et al. 2022). Additionally, the seabird colonies located at this area may also be a source of metals, as seabird guano was reported to contain high concentrations of Fe, Cu, and Zn, and other metals (Justel-Díez et al. 2023). We were not able to detect any direct source of pollution on this bay that could justify such high levels of contamination.

Arsenic is a toxic metalloid, and its concentrations were found to be higher on the leeward coast, exceeding the toxic effects threshold at three sites in the multiple-use area (Biboca, Morro de Fora and Boldró River), indicating a local source of the metal. The concentration was also above the threshold effect level at Sueste, a no-take site on the windward coast. Although As is possibly associated with human activities and untreated wastewater (Makuwa et al. 2020), the exact sources of this metalloid remain unclear (Kroon et al. 2020, Tebbett et al. 2022). Furthermore, it is important to highlight that the sediments in Boldró River, which receive effluents from the Boldró WTP, were found to be contaminated by coprostanol. This finding, coupled with nutrient levels in this WTP effluents exceeding the limits permitted by Brazilian legislation, raises concerns about the efficiency of the sewage treatment plants.

Microplastics were detected ubiquitously in both the sediments and waters of Fernando de Noronha, irrespective of their protection category. Specifically, the sediments in Golfinhos exhibited higher concentrations of microplastics compared to the other sampled sites, reinforcing the hypothesis that coastal circulation patterns play a significant role in the deposition of contaminants from external sources in this site. The water samples taken from the reefs on the windward coast, exposed to the prevailing currents and winds, displayed higher microplastic levels than the leeward coast. This indicates that allochthonous sources of microplastics in the reef waters in Fernando de Noronha may be more substantial than autochthonous sources. This finding aligns with previous studies reporting a predominance of macro marine debris and microplastics on windward beaches of Fernando de Noronha and other islands (Ivar do Sul et al. 2009, Monteiro et al. 2018, Grillo and Mello 2021), which can be attributed to the influence of long-range transport facilitated by superficial marine currents.

Nevertheless, microplastics were identified in the effluents from both wastewater treatment plants and in the Sancho River, which flows into the no-take area, implying that locally generated microplastics are also contributing to the pollution affecting the reefs in the archipelago. Previous research focusing on microplastics in Fernando de Noronha reported their presence on beaches (Monteiro *et al*. 2020), surface tows (Ivar do Sul et al. 2014) and pelagic predators (Justino et al. 2023a). This is the first study to investigate the presence of microplastics into the water and sediments directly surrounding the reefs, and likely interacting with the reef fauna.

Impacts of pollution on reef communities

The sites Porto, Biboca, Morro de Fora, and Boldró, where we observed the highest ^{15}N enrichment, hosted distinct algae communities compared to other sites along the leeward coast. This difference was evident in the dominance of fast-growing green macroalgae *C. verticilata* and the brown macroalgae *S. schroederi*, both in cover and biomass. The algae *Spatoglossum schroederi* was exclusively found at Porto and Biboca, and its presence positively correlated with higher concentrations of coprostanol. Among all the sites in the archipelago, Biboca displayed the highest cover of brown macroalgae. Additionally, *Bryopsis* spp., a well-known indicator of nutrient enrichment (Lapointe et al. 2010, Han et al. 2023) was only found at Biboca (Fig. 10a). In contrast, the corals *M. cavernosa* and *S. stellata* appeared as indicators of the no-take area, evidencing its importance for the conservation of these organisms.

Despite being the most important benthic groups in the archipelago, the cover of brown macroalgae was higher in the multiple-use area, possibly indicating that nutrient enrichment favours its growth. In the most polluted sites, algal turf were replaced by brown macroalgae and *C. verticilata*, indicating a shift in the benthic community composition. At Boldró, *C. verticilata* was observed overgrowing the reef and even unconsolidated substrate, a growth pattern previously associated to reefs experiencing elevated nutrient concentrations (Lapointe et al. 2010). Blooms of *Ulva* sp. were also observed at Boldró and other beaches downstream of coastal currents (Fig. 10b). *Ulva* sp. blooms have often been associated with elevated nitrogen concentrations (Lapointe et al. 2010, Alldread et al. 2023).

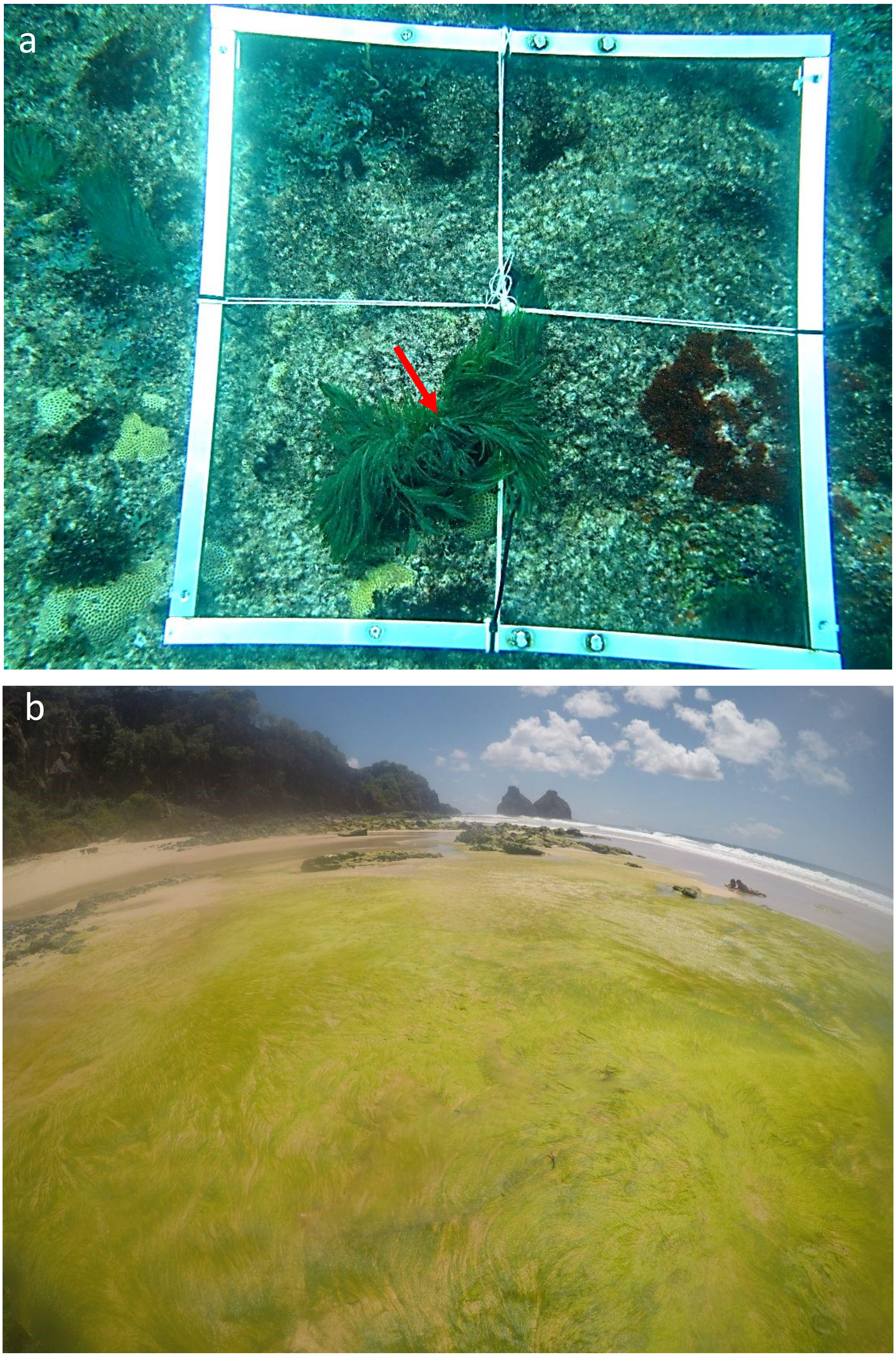


Figure 10: (a) *Bryopsis* sp. at Biboca reef (red arrow), (b) Green tide of *Ulva* sp. on Boldró beach (photo: F. Furlani).

One of the most reported responses to nutrient enrichment is the increase of benthic macroalgae (Pastorok and Bilyard 1985, Lapointe et al. 2004). While our study primarily focused on investigating the direct effects of nutrient enrichment, it is important to acknowledge that other factors, not directly examined here, can also influence macroalgae biomass, such as wave action and herbivory (Kraufvelin *et al*. 2010, Matheus et al. 2019, Longo et al. 2015). These variables may contribute to the observed variations among sites, particularly along the windward coast. However, among the sites on the leeward coast, we found significantly higher macroalgae biomass in the multiple-use areas when compared to leeward sites within the no-take area.

Nutrient enrichment is also linked to an increase in filter-feeding invertebrates such as bryozoans, sponges, and tunicates (Pastorok and Bilyard 1985). We observed a positive correlation between sponge cover and orthophosphate concentration, which provides a potential explanation for the elevated sponge cover observed at Morro de Fora, a site characterized by high orthophosphate concentrations. We also observed clear indications of nutrient pollution and its impact on the benthic communities within the multiple-use sites. Additionally, we identified signs of pollution from nutrients, metals, microplastics, and PAHs in the no-take sites, although at lower levels that may not have immediate consequences for the reef community structure. Nonetheless, pollution can have diverse effects on marine organisms, including sublethal and indirect impacts (Yu et al. 2020, Araújo et al. 2023), that can affect their fitness and reproduction. Assessing these impacts demand studies specifically designed to detect and measure such effects (e.g. Ferreira et al. 2023, Walkinshaw et al. 2023).

Despite considerable research on the impact of pollution on benthic communities, there has been a relative lack of focus on the effects it has on reef fish assemblages (Wenger et al. 2015). Interestingly, fish assemblages often exhibit a quicker response time to sewage disposal compared to sessile benthic organisms (Guidetti et al. 2003), yet this aspect remains understudied. The indirect effects of declining habitat quality and loss of coral cover are believed to have implications for coral-associated reef fishes, but there is a lack of information on the specific effects of nutrient enrichment, chemical pollution, and their interactions on coral reef fishes (Wenger et al. 2015).

The relationship between reef fishes and decline in water quality is complex, and although studies comparing low and highly-impacted watersheds have reported higher diversity, abundance and biomass on healthier reefs, the presence of other environmental gradients makes it challenging to differentiate natural patterns from anthropogenic effects (Rodgers et al. 2012). We observed lower fish biomass and abundance in the multiple-use area sites compared to the no-take areas along the leeward coast. As the multiple-use area is more polluted, these lower values may be related to anthropogenic impacts. However, when considering all sampling sites together, mean differences between protection categories were significant only for fish abundance. This pattern suggests that habitat conformation and oceanographic patterns have a predominant influence on shaping fish communities, while anthropogenic impacts exert their effects at a more localized scale on each coast. Habitat complexity had outstanding importance as a driver of fish community structure in our study. Fish assemblages in FN and other southwestern Atlantic oceanic islands are primarily influenced by their intricate phylogenetic history and geographical isolation (Gomes et al. 2023). While they also respond to biotic and abiotic factors (Krajewski and Floeter 2011), the role of primary productivity stands out as particularly significant (Gomes et al. 2023), evidencing the importance of nutrient limitation and enrichment in shaping fish communities.

We detected a negative effect of nutrient enrichment ($\delta^{15}N$ and ammonia) on fish richness and abundance, although some species displayed a positive correlation with $\delta^{15}N$ (*M. lianae*) and with ammonia (*S. axillare*, *A. multillineata, C. fulva* and *H. adcesnionis*). Additionally, orthophosphate was strongly positively related to *K. sectatrix*, but negatively related to *C. fulva* and *S. frondusum.* Indirect effects may underlie the observed patterns, as nutrient enrichment can alter the composition of the benthic community and subsequently influence the availability of specific food groups (Azzurro et al. 2010). For instance, *Kyphosus sectatrix* is known to primarily consume corticated algae such as *Dictyota* sp. (Mendes et al. 2018), which thrive in sites with higher orthophosphate concentrations. Additionally, previous studies have detected that the response of fishes to nutrient enrichment follows a non-linear pattern, with low levels of nutrient enrichment benefiting fish populations, and as nutrient enrichment surpasses certain thresholds, fish abundance declines (Wenger *et al* 2015). This underscores the complexity of the effects of nutrient enrichment, which can vary depending on the species, the type of nutrient, and its concentration.

Chemical pollution by metals and PAHs can also lead to the accumulation of toxins, which in turn can have detrimental effects on fish. Elevated concentrations of surfactants (detergent components) have been related to genomic damage in sharks in FN

(Araújo et al. 2023). Reduced growth rates and decreased larval survivorship are among the observed consequences of such exposure (Wenger et al. 2015). Although we observed positive correlations of the abundance of some fish species and metal concentrations, they are probably due to habitat associations of these species to the sites where higher concentrations of metals were observed (e.g. *C. fulva* and Golfinhos). Specific responses of fish to chemical pollution, in the levels of organisms and populations, should be investigated within the sites where we detected contamination in our study area.

The observed correlation between the concentration of microplastics in the water and the structure of fish communities, probably due to the difference in fish communities between coasts, coincides with the association of microplastics with the windward sites. While we did not directly assess the presence of microplastics in the organisms themselves, mounting evidence indicates that fish exposed to microplastics are susceptible to contamination (Justino et al. 2023a, Justino et al. 2023b). This emphasizes the significance of this impact on the no-take areas of FN.

Monitoring the evaluated variables is important, particularly considering the influence of varying climatic conditions, to comprehensively assess the temporal evolution and distribution of contaminants. Our assessment was conducted during the rainy season when there is typically increased terrestrial input. However, bottom resuspension events, often associated with elevated wave heights, are more prevalent during the dry period, when swells from the north predominate (Mello et al. 2023). These events can result in the release of sediment-bound contaminants into the water column, exerting significant impacts on the pelagic community.

This was the first study to provide a comprehensive investigation of marine pollution in FN, providing baseline data and management recommendations. The mapping and analysis employed in this study, in particular the monitoring of isotopic signatures in macroalgae have the potential to be widely used in other MPAs due to their low cost and ease of application. Considering the findings presented, future investigations should incorporate experimental methodologies to explore the specific effects of these contaminants on organisms and ecological communities.

Conclusions

We assessed multiple marine pollution parameters in the water and sediments around Fernando de Noronha archipelago. We also determined the isotopic signatures of macroalgae and characterized macroalgae biomass, benthic communities, and fish communities at the sampling sites in no-take and multiple-use areas. Although the effluents from wastewater treatment plants presented nutrient concentrations that did not comply with the legislation, we observed low concentrations of pollutants in most water samples but detected significant ^{15}N enrichment in macroalgae in the proximity of the sites where effluents are discharged, with evident signs of impacts on the benthic communities. Our findings suggest that nutrients, metals, and microplastics originating from the multiple-use area are being transported and accumulating at the no-take area due to coastal currents. Coastal circulation plays a role in dispersing pollutants, but it can also transport pollutants over considerable distances, such as from the multiple-use area to the no-take area where human activities are minimal.

Among the issues detected in our study, effluent treatment, and contamination by PAHs deserve urgent attention. It is imperative to improve the sewage treatment infrastructure, enhance the effectiveness of wastewater treatment processes, and ensure compliance with regulatory standards to mitigate the negative impact on the marine ecosystem of Fernando de Noronha. In particular, the planning of sewage infrastructure should consider implementing tertiary treatment methods to reduce nutrient inputs and prevent eutrophication (Onodera et al. 2021), the utilization of decentralized treatment stations, and account for oceanic circulation patterns and bathymetry.

Addressing contamination issues at Porto also requires immediate action to establish appropriate infrastructure for vessel maintenance and refurbishment activities, as well as fuel transfer operations. It is crucial to implement robust containment systems and enforce strict protocols to prevent oil spills and minimize the potential impact of chemical pollution on the marine ecosystem. Tackling fecal contamination is another critical area that demands attention in Porto.

In the windward coast, where there is no human occupation, the main pollutant were microplastics transported by ocean currents. In contrast, in the leeward coast, encompassing urban areas, there are evidences of nutrient enrichment and punctual sewage contamination, as well as accumulations of these stressors due to the currents. Marine protected areas are important tools for regulating the use and occupation of territory, but without proper waste management and sewage treatment the protected area alone cannot guarantee the integrity of the ecosystems. In other words, MPAs need to be associated with a comprehensive management of the impacts that along the coastline and the subsequent flow of pollutants into the marine environment. Similar scenarios are likely in most of coastal MPA in Brazil, and FN serves as a compelling case study because of its small scale and open-ocean environment, that should be theoretically more controlled, but in reality, demonstrates the challenges in effectively managing such areas.

Despite experiencing relatively low anthropogenic impacts compared to coastal reef environments (Magris et al. 2018, Araújo et al. 2023), the growing trends of tourism and urban development in FN indicate a potential worsening of marine pollution. It is important to recognize that the response of organisms and reef communities to pollution often exhibits a non-linear pattern, implying that there is a threshold level at which detrimental effects become apparent. Beyond this critical point, the impacts transition from being relatively benign to becoming harmful (Wenger et al. 2015). There is emerging evidence that pollution undermines the resilience of corals and other reef organisms in the face of global climate change, presenting an additional and escalating threat (van Dam et al. 2011).Therefore, if the current trend of increasing impacts continues, there is a risk of severe consequences for the marine ecosystem in FN, not only at the multiple-use area but also at the no-take area.

Acknowledgments

We thank A.C.G. Monteiro, J. Bleuel, and K. Y. Inagaki for their invaluable help in the fieldwork, and the staff at Sea Paradise Dive Center. This work was supported by the National Council for Scientific and Technological Development (CNPq; Chamada 31-2019-Grant 443329/2019-2, awarded to GOL), Serrapilheira Institute (Grant No. Serra-1708-15364, awarded to GOL) and partially financed by Coordenação de Aperfeiçoamento de Pessoal de Nível Superior–Brasil (CAPES)– Code 001. GOL is also grateful to a research productivity scholarship provided by CNPq (308072/2022-7). This work was conducted under the SISBIO permit #80434.

References

Adam, T.C., Burkepile, D.E., Holbrook, S.J., Carpenter, R.C., Claudet, J., Loiseau, C., Thiault, L., Brooks, A.J., Washburn, L., Schmitt, R.J., 2021. Landscape-scale patterns of nutrient enrichment in a coral reef ecosystem: implications for coral to algae phase shifts. Ecol. Appl. 31, 1–16. https://doi.org/10.1002/eap.2227

Alldred, F.C., Gröcke, D.R., Leung, C.Y., Wright, L.P., Banfiel, 2023. Diffuse and concentrated nitrogen sewage pollution in island environments with differing treatment systems. Sci. Rep. 1–9. https://doi.org/10.1038/s41598-023-32105-6

Anthony, K.R.N., Helmstedt, K.J., Bay, L.K., Fidelman, P., Hussey, E., Lundgren, P., Mead, D., Mcleod, I.M., Mumby, P.J., Newlands, M., Schaffelke, B., Wilson, K.A., Hardisty, P.E., 2020. Interventions to help coral reefs under global change — A complex decision challenge. PLoS One 15, e0236399. https://doi.org/10.1371/journal.pone.0236399

Araújo, C., Carneiro, P., Fidelis, L., Nascimento, B., Antunes, M., Viana, D., Oliveira, P., Torres, R., Hazin, F., Adam, M., 2023. Comparative genomic damage among three shark species with different habits: Sublethal impacts of human origin in a protected island environment in the South Atlantic. Mar. Pollut. Bull. 191. https://doi.org/10.1016/j.marpolbul.2023.114924

Araújo, M.P., Hamacher, C., Farias, C. de O., Soares, M.L.G., 2021. Fecal sterols as sewage contamination indicators in Brazilian mangroves. Mar. Pollut. Bull. 165, 112149. https://doi.org/10.1016/j.marpolbul.2021.112149

Assunção, R. V., Silva, A.., Martins, J., Flores Montes, M., 2016. Spatial-Temporal Variability of the Thermohaline Properties in the Coastal Region of Fernando de Spatial-Temporal Variability of the Thermohaline Properties in the. J. Coast. Res. 512–516. https://doi.org/10.2112/SI75-103.1

Azzurro, E., Matiddi, M., Fanelli, E., Guidetti, P., Mesa, G. La, Scarpato, A., Axiak, V., 2010. Sewage pollution impact on Mediterranean rocky-reef fish assemblages. Mar. Environ. Res. 69, 390–397. https://doi.org/10.1016/j.marenvres.2010.01.006

Barcellos, R.L., Oliveira, L.E. De, Flores-montes, M.D.J., 2018. Spatial sedimentary distribution, seasonality and the characteristics of organic matter on Fernando de Noronha insular shelf. Brazilian J. Oceanogr. 66, 131–156. https://doi.org/10.1590/S1679-87592018146206601

Bastos, L.P.H., da Costa Cavalcante, D., Alferes, C.L.F., da Silva, D.B.N., de Oliveira Ferreira, L., Rodrigues, R., Pereira, E., 2022. Fingerprinting an oil spill event (August of 2021) in the oceanic Fernando de Noronha archipelago using biomarkers and stable carbon isotopes. Mar. Pollut. Bull. 185. https://doi.org/10.1016/j.marpolbul.2022.114316

Baumann, J.H., Zhao, L.Z., Stier, A.C., Bruno, J.F., 2022. Remoteness does not enhance coral reef resilience. Glob. Chang. Biol. 28, 417–428. https://doi.org/10.1111/gcb.15904

Baumard, P., Budzinski, H., Garrigues, P., 1998. POLYCYCLIC AROMATIC HYDROCARBONS IN SEDIMENTS AND MUSSELS OF THE MEDITERRANEAN SEA. Environ. Toxicol. Chem. 17, 765–776.

Baum, J.K., Claar, D.C., Tietjen, K.L., Magel, J.M.T., Maucieri, D.G., Cobb, K.M., Mcdevitt-Irwin, J.M., 2022. Transformation of coral communities subjected to an unprecedented heatwave is modulated by local disturbance 5615. https://doi.org/10.1126/sciadv.abq5615

Beijbom, P.J. Edmunds, D.I. Kline, G.B. Mitchell, D. Kriegman. "Automated Annotation of Coral Reef Survey Images". IEEE Conference on Computer Vision and Pattern Recognition (CVPR), Providence, Rhode Island, June, 2012.

Bertini, L., Braga, E. de S., 2022. The Contribution of Nutrients and Water Properties to the Carbonate System in Three Particular Areas of the Tropical Atlantic (NE-BRAZIL). J. Geosci. Environ. Prot. 10, 135–161. https://doi.org/10.4236/gep.2022.102009

Bradl, H. (Ed.). (2005). *Heavy metals in the environment: origin, interaction and remediation*. Elsevier.

Brasil, 2005. CONAMA Resolução 357, de 17 de março de 2005. Available in: http://conama.mma.gov.br/. (Accessed 29 August 2022)

Brasil, 2008. CONAMA Resolução 397, de 03 de abril de 2008. Available in: http://conama.mma.gov.br/. (Accessed 29 August 2022)

Brasil, 2011. CONAMA Resolução 430, de 13 de maio de 2011. Available in: http://conama.mma.gov.br/. (Accessed 29 August 2022)

Bruno, J.F., Bates, A.E., Cacciapaglia, C., Pike, E.P., Amstrup, S.C., Van Hooidonk, R., Henson, S.A., Aronson, R.B., 2018. Climate change threatens the world's marine protected areas. Nat. Clim. Chang. 8, 499–503. https://doi.org/10.1038/s41558-018-0149-2

Buchman, M.F., 2008. NOAA Screenign Quick reference Tables, NOAA OR&R Report 08-1, Seatle WA, Office of response and Restoration Division, National Oceanic and Atmospheric Administration, 34 pages.

Burke, L., Reytar, K., Spalding, M., Perry, A., 2011. Reefs at Risk Revisited. World Resources Institute, Washington, DC.

Cavalcante, R.M., Sousa, F.W., Nascimento, R.F., Silveira, E.R., Freire, G.S.S., 2009. The impact of urbanization on tropical mangroves (Fortaleza, Brazil): Evidence from PAH distribution in sediments. J. Environ. Manage. 91, 328–335. https://doi.org/10.1016/j.jenvman.2009.08.020

Combi, T., Montone, R.C., Corada-Fernandez, C., Lara-Martín, P.A., Gusmao, J.B., Santos, M.C. de O., 2022. Persistent organic pollutants and contaminants of emerging concern in spinner dolphins (*Stenella longirostris*) from the Western

Atlantic Ocean. Mar. Pollut. Bull. 174, 113263. https://doi.org/10.1016/j.marpolbul.2021.113263

Costanzo, S., O'Donohue, M., Dennison, W., Loneragan, N., Thomas, M. 2001. A New Approach for Detecting and Mapping Sewage Impacts. *Marine Pollution Bulletin*, *42*(2), 149-156. https://doi.org/10.1016/S0025-326X(00)00125-9

Cristiano, C., Camboim, G., Carla, L., 2020. Beach landscape management as a sustainable tourism resource in Fernando de Noronha Island (Brazil). Mar. Pollut. Bull. 150, 110621. https://doi.org/10.1016/j.marpolbul.2019.110621

Dalongeville, A.A., Boulanger, E., Marques, V., Charbonnel, E., Hartmann, V., Santoni, M.C., Deter, J., Valentine, A., Lenfant, P., Boissery, P., Dejean, T., Velez, L., Pichot, F., Sanchez, L., Arnal, V., Bockel, T., Delaruelle, G., Holon, F., Milhau, T., Romant, L., Manel, S., Mouillot, D., 2022. Benchmarking eleven biodiversity indicators based on environmental DNA surveys: more diverse functional traits and evolutionary lineages inside marine reserves. J. Appl. Ecol. https://doi.org/10.1111/1365-2664.14276

Dudley, N. (Editor) (2008). Guidelines for Applying Protected Area Management Categories. Gland, Switzerland: IUCN. x + 86pp. WITH Stolton, S., P. Shadie and N. Dudley (2013). IUCN WCPA Best Practice Guidance on Recognising Protected Areas and Assigning Management Categories and Governance Types, Best Practice Protected Area Guidelines Series No. 21, Gland, Switzerland: IUCN.

Duprey, N.N., Wang, X.T., Thompson, P.D., Pleadwell, J.E., Raymundo, L.J., Kim, K., Sigman, D.M., Baker, D.M., 2017. Life and death of a sewage treatment plant recorded in a coral skeleton Δ15N record. Mar. Pollut. Bull. 120, 109–116. https://doi.org/10.1016/j.marpolbul.2017.04.023

Eddy, T.D., Lam, V.W.Y., Reygondeau, G., Cisneros-Montemayor, A.M., Greer, K., Palomares, M.L.D., Bruno, J.F., Ota, Y., Cheung, W.W.L., 2021. Global decline in capacity of coral reefs to provide ecosystem services Global decline in capacity of coral reefs to provide ecosystem services. One Earth 4, 1278–1285. https://doi.org/10.1016/j.oneear.2021.08.016

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S.C., Banks, S., S. Barrett, N.S. et al. 2014. Global Conservation Outcomes Depend on Marine Protected Areas with Five Key Features. Nature 506(7487):216–20. https://doi.org/10.1038/nature13022

Elser, J.J., Urabe, J., 1999. The Stoichiometry of Consumer-Driven Nutrient Recycling: Theory, Observations, and Consequences. Ecology 80, 735. https://doi.org/10.2307/177013

Fabricio Neta, A.D.B., Nascimento, C.W.., Biondi, C.M., Straaten, P. van, Bittar, S.M.B., 2018. Natural concentrations and reference values for trace elements in soils of a tropical volcanic archipelago. Environ. Geochemical Heal. 163–173. https://doi.org/10.1007/s10653-016-9890-5

Fabricius, K.E., 2005. Effects of terrestrial runoff on the ecology of corals and coral reefs: review and synthesis. Mar. Pollut. Bull. 50, 125–146. https://doi.org/10.1016/j.marpolbul.2004.11.028

Ferreira, L.D.S., Lopes, R.P., Ulbrich, M.N.C., Guaratini, T., Colepicolo, P., Lopes, N.P., Garla, R.C., Oliveira Filho, E.C., Pohlit, A.M., Zucchi, O.L.A.D., 2012. Concentration of Inorganic Elements Content in Benthic Seaweeds of Fernando de Noronha Archipelago by Synchrotron Radiation Total Reflection X-Ray Fluorescence Analysis (SRTXRF). Int. J. Anal. Chem. 2012, 1–8. https://doi.org/10.1155/2012/407274

Ferreira, N.M., Coutinho, R., de Oliveira, L.S., 2023. Emerging studies on oil pollution biomonitoring: A systematic review. Mar. Pollut. Bull. 192, 115081. https://doi.org/10.1016/j.marpolbul.2023.115081

Froese, R., Payly, D. 2023. Editors. FishBase. World Wide Web electronic publication. www.fishbase.org, (06/2023)

Guidetti, P., Terlizzi, A., Fraschetti, S., Boero, F., 2003. Changes in Mediterranean rocky-reef fish assemblages exposed to sewage pollution. Mar. Ecol. Prog. Ser. 253, 269–278. https://doi.org/10.3354/meps253269

Graham, N.A.J., Wilson, S.K., Carr, P., Hoey, A.S., Jennings, S., MacNeil, M.A., 2018. Seabirds enhance coral reef productivity and functioning in the absence of invasive rats. Nature 559, 250–253.

Grillo, A.C., Mello, T.J., 2021. Marine debris in the Fernando de Noronha Archipelago , a remote oceanic marine protected area in tropical SW Atlantic. Mar. Pollut. Bull. 164. https://doi.org/10.1016/j.marpolbul.2021.112021

Gomes, L. de M.J., Garcia, G.S., Cordeiro, C.A.M.M., Gouveia, N.A., Ferreira, C.E.L., Bender, M.G., Longo, G.O., Quimbayo, J.P., Gherardi, D.F.M., 2023. Complex phylogenetic origin and geographic isolation drive reef fishes response to environmental variability in oceanic islands of the southwestern Atlantic. Ecography (Cop.). 4, 527–534. https://doi.org/10.1111/ecog.06559

Halpern, B.S., Frazier, M., Potapenko, J., Casey, K.S., Koenig, K., Longo, C., Lowndes, J.S., Rockwood, R.C., Selig, E.R., Selkoe, K.A., Walbridge, S., 2015. Spatial and temporal changes in cumulative human impacts on the world’s ocean. Nat. Commun. 6, 1–7. https://doi.org/10.1038/ncomms8615

Han, H., Wen, R., Wang, H., Zhao, S., 2023. Comparison of growth and nutrient uptake capacities of three dominant species of Qinhuangdao green tides. Acta Oceanol. Sin. https://doi.org/10.1007/s13131-022-2100-7

Hauser-Davis, R.A., Bordon, I.C., Willmer, I.Q., Lopes, A.P., Moreira, S.C., Saint’Pierre, T.D., Vianna, M., 2023. First metal and metalloid study assessment for the Black Triggerfish *Melichthys niger* (Bloch, 1786): Baseline data from a pristine South Atlantic oceanic island. Mar. Pollut. Bull. 187. https://doi.org/10.1016/j.marpolbul.2023.114593

Horta, E.F., Ambrosio, J.K., Pires, J.M., 2020. Geração de energia em Fernando de Noronha Alternativas para a diminuição de emissões de CO2 no transporte e eletricidade, WWF- Brasil.

Hunter, C.L., Evans, C.W., 1995. Coral reefs in Kaneohe Bay, Hawaii: Two centuries of western influence and two decades of data. Bull. Mar. Sci. 57, 501–515.

ICMBio 2021. Painel dinâmico de informações. Accessed in 07/27/2022. http://qv.icmbio.gov.br/QvAJAXZfc/opendoc2.htm?document=painel_corporativo_6476.qvw&host=Local&anonymous=true

IUCN, 2020. Brazilian Atlantic Islands: Fernando de Noronha and Atol das Rocas Reserves - 2020 Conservation outlook assessment. Accessed in 06/04/2023. https://worldheritageoutlook.iucn.org/explore-sites/wdpaid/900631

Ivar do Sul, J.A., Spengler, Â., Costa, M.F., 2009. Here, there and everywhere. Small plastic fragments and pellets on beaches of Fernando de Noronha (Equatorial Western Atlantic). Mar. Pollut. Bull. 58, 1236–1238. https://doi.org/10.1016/j.marpolbul.2009.05.004

Ivar Do Sul, J.A., Costa, M.F., Fillmann, G., 2014. Microplastics in the pelagic environment around oceanic islands of the western Tropical Atlantic Ocean. Water. Air. Soil Pollut. 225. https://doi.org/10.1007/s11270-014-2004-z

Justel-Díez, M., Delgadillo-Nuño, E., Gutiérrez-Barral, A., García-Otero, P., Alonso-Barciela, I., Pereira-Villanueva, P., Álvares-Salgado, X.A., Velando, A., Teira, E., Fernández, E., 2023. Inputs of seabird guano alter microbial growth, community composition and the phytoplankton-bacterial interactions in a coastal system. Environ. Microbiol. 1–19. https://doi.org/10.1111/1462-2920.16349

Justino, A.K.S., Ferreira, G.V.B., Fauvelle, V., Schmidt, N., Lenoble, V., Pelage, L., Martins, K., Travassos, P., Lucena-Frédou, F., 2023a. From prey to predators: Evidence of microplastic trophic transfer in tuna and large pelagic species in the southwestern Tropical Atlantic. Environ. Pollut. 327. https://doi.org/10.1016/j.envpol.2023.121532

Justino, A.K.S., Ferreira, G.V.B., Fauvelle, V., Schmidt, N., Lenoble, V., Pelage, L., Lucena-Frédou, F., 2023b. Exploring microplastic contamination in reef-associated fishes of the Tropical Atlantic. Mar. Pollut. Bull. 192, 115087. https://doi.org/10.1016/j.marpolbul.2023.115087

Kohler, K.E., Gill, S.M., 2006. Coral Point Count with Excel extensions (CPCe): A Visual Basic program for the determination of coral and substrate coverage using random point count methodology. Computers and Geosciences, Vol. 32, No. 9, pp. 1259-1269, doi:10.1016/j.cageo.2005.11.009.

Krajewski, J.P., Floeter, S.R., 2011. Reef fish community structure of the Fernando de Noronha Archipelago (Equatorial Western Atlantic): The influence of exposure and benthic composition. Environ. Biol. Fishes 92, 25–40. https://doi.org/10.1007/s10641-011-9813-3

Kraufvelin, P., Lindholm, A., Pedersen, M.F., Kirkerud, L.A., Bonsdorff, E., 2010. Biomass, diversity and production of rocky shore macroalgae at two nutrient enrichment and wave action levels. Mar. Biol. 157, 29–47. https://doi.org/10.1007/s00227-009-1293-z

Kroon, F.J., Berry, K.L.E., Brinkman, D.L., Kookana, R., Leusch, F.D.L., Melvin, S.D., Neale, P.A., Negri, A.P., Puotinen, M., Tsang, J.J., van de Merwe, J.P., Williams, M., 2020. Sources, presence and potential effects of contaminants of emerging concern in the marine environments of the Great Barrier Reef and Torres Strait, Australia. Sci. Total Environ. 719, 135140. https://doi.org/10.1016/j.scitotenv.2019.135140

Lamb, J.B., Willis, B.L., Fiorenza, E.A., Couch, C.S., Howard, R., Rader, D.N., True, J.D., Kelly, L.A., Ahmad, A., Jompa, J., Harvell, C.D., 2018. Plastic waste associated with disease on coral reefs. Science 359, 460–462. https://doi.org/10.1126/science.aar3320

Lapointe, B.E., Barile, P.J., Yentsch, C.S., Littler, M.M., Littler, D.S., Kakuk, B., 2004. The relative importance of nutrient enrichment and herbivory on macroalgal communities near Norman's Pond Cay, Exumas Cays, Bahamas: A "natural" enrichment experiment. J. Exp. Mar. Bio. Ecol. 298, 275–301. https://doi.org/10.1016/S0022-0981(03)00363-0

Lapointe, B.E., Barile, P.J., Littler, M.M., Littler, D.S., 2005. Macroalgal blooms on southeast Florida coral reefs: II. Cross-shelf discrimination of nitrogen sources indicates widespread assimilation of sewage nitrogen. Harmful Algae 4, 1106–1122. https://doi.org/10.1016/j.hal.2005.06.002

Lapointe, B.E., Langton, R., Bedford, B.J., Potts, A.C., Day, O., Hu, C., 2010. Land-based nutrient enrichment of the Buccoo Reef Complex and fringing coral reefs of Tobago , West Indies. Mar. Pollut. Bull. 60, 334–343. https://doi.org/10.1016/j.marpolbul.2009.10.020

Lapointe, B.E., Tewfik, A., Phillips, M., 2021. Macroalgae reveal nitrogen enrichment and elevated N : P ratios on the Belize Barrier Reef. Mar. Pollut. Bull. 171, 112686. https://doi.org/10.1016/j.marpolbul.2021.112686

Longo, G.O., Morais, R.A., Martins, C.D.L., Mendes, T.C., Aued, A.W., Cândido, D. V., De Oliveira, J.C., Nunes, L.T., Fontoura, L., Sissini, M.N., Teschima, M.M., Silva, M.B., Ramlov, F., Gouvea, L.P., Ferreira, C.E.L., Segal, B., Horta, P.A., Floeter, S.R., 2015. Between-habitat variation of benthic cover, reef fish assemblage and feeding pressure on the benthos at the only atoll in South Atlantic: Rocas atoll, NE Brazil. PLoS One 10, 1–29. https://doi.org/10.1371/journal.pone.0127176

Magris, R.A., Grech, A., Pressey, R.L., 2018. Cumulative human impacts on coral reefs: Assessing risk and management implications for brazilian coral reefs. Diversity 10, 1–14. https://doi.org/10.3390/d10020026

Magris, R.A., Costa, M.D.P., Ferreira, C.E.L., Vilar, C.C., Joyeux, J.-C., Creed, J.C., Copertino, M.S., Horta, P.A., Sumida, P.Y.G., Francini-Filho, R.B., Floeter, S.R.,

2021. A blueprint for securing Brazil's marine biodiversity and supporting the achievement of global conservation goals. Divers. Distrib. 27, 198–215. https://doi.org/10.1111/ddi.13183

Makuwa, S., Tlou, M., Fosso-Kankeu, E., Green, E., 2020. Evaluation of fecal coliform prevalence and physicochemical indicators in the effluent from a wastewater treatment plant in the north-west province, south africa. Int. J. Environ. Res. Public Health 17, 1–13. https://doi.org/10.3390/ijerph17176381

Matheus, Z., Francini-Filho, R.B., Pereira-Filho, G.H., Moraes, F.C., Moura, R.L., Brasileiro, P.S., Amado-Filho, G.M., 2019. Benthic reef assemblages of the Fernando de Noronha Archipelago, tropical South-west Atlantic: Effects of depth, wave exposure and cross-shelf positioning. PLoS One 14, 1–15. https://doi.org/https://doi.org/10.1371/journal.pone.0210664

Mendes, T.C., Ferreira, C.E.L., Clements, K.D., 2018. Discordance between diet analysis and dietary macronutrient content in four nominally herbivorous fishes from the Southwestern Atlantic. Mar. Biol. 165, 1–12. https://doi.org/10.1007/s00227-018-3438-4

Mello, T.J., Vieira, E.A., Garrido, A.G., Zilberberg, C., Lima, J.L., Santos, L.P.S., Longo, G.O., 2023. Drivers of temporal variation in benthic cover and coral health of an oceanic intertidal reef in Southwestern Atlantic. Reg. Stud. Mar. Sci. 60, 102874. https://doi.org/10.1016/j.rsma.2023.102874

Michener, R., Lajtha, K., 2007. Stable Isotopes in Ecology and Environmental Science, 2nd ed. Blackwell Publishing, Carlton.

Monteiro, R.C.P., Ivar do Sul, J.A., Costa, M.F., 2018. Plastic pollution in islands of the Atlantic Ocean. Environ. Pollut. 238, 103–110. https://doi.org/10.1016/j.envpol.2018.01.096

Monteiro, R.C.P., Ivar do Sul, J.A., Costa, M.F., 2020. Small microplastics on beaches of fernando de noronha island, tropical atlantic ocean. Ocean Coast. Res. 68, 1–10. https://doi.org/10.1590/S2675-28242020068235

Oliveira, S.M.B., Pessenda, L.C.R., Gouveira, S.E.., Favaro, D.I.T., 2011. Heavy metal concentrations in soils from a remote oceanic island, Fernando de Noronha, Brazil. Ann. Brazilian Acad. Sci. 83, 1193–1206.

Oliveira, T.B.N., 2018. Hidrodinâmica e conectividade entre ambientes recifais no Arquipélago de Fernando de Noronha (PE). Universidade de São Paulo.

Onodera, T., Komatsu, K., Kohzu, A., Kanaya, G., Mizuochi, M., Syutsubo, K., 2021. Science of the Total Environment Evaluation of stable isotope ratios (δ 15 N and δ 18 O) of nitrate in advanced sewage treatment processes : Isotopic signature in four process types. Sci. Total Environ. 762, 144120. https://doi.org/10.1016/j.scitotenv.2020.144120

Ostle, C., Thompson, R.C., Broughton, D., Gregory, L., Wootton, M., Johns, D.G., 2019. The rise in ocean plastics evidenced from a 60-year time series. Nat. Commun. 10, 1622. https://doi.org/10.1038/s41467-019-09506-1

Pastorok, R.A., Bilyard, G.R., 1985. Effects of sewage pollution on coral-reef communities. Mar. Ecol. Prog. Ser. 21, 175–189. https://doi.org/10.3354/meps021175

Pimentel, C.R., Rocha, L.A., Shepherd, B., Phelps, T.A.Y., Joyeux, J., Martins, A.S., Stein, C.E., Teixeira, J.B., Gasparini, J.L., Reis-filho, J.A., Garla, R.C., Francini-filho, R.B., Delfino, S.D.T., Mello, T.J., Giarrizzo, T., Pinheiro, H.T., 2020. Mesophotic ecosystems at Fernando de Noronha Archipelago, Brazil (South- western Atlantic), reveal unique ichthyofauna and need for conservation. Neotrop. Ichthyol. 4, e200050. https://doi.org/https://doi.org/10.1590/1982-0224-2020-0050

Prazeres, M.F., Martins, S.E., Bianchini, A., 2012. Assessment of water quality in coastal waters of Fernando de Noronha, Brazil: biomarker analyses in *Amphistegina lessonii*. J. Foraminifer. Res. 42, 56–65.

Preisner, M., 2020. Surface Water Pollution by Untreated Municipal Wastewater Discharge Due to a Sewer Failure. Environ. Process. 2020, 767–780. https://doi.org//10.1007/s40710-020-00452-5

R Core Team (2022). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Risk, M.J., Lapointe, B.E., Sherwood, O.A., Bedford, B.J., 2009. The use of δ15N in assessing sewage stress on coral reefs. Mar. Pollut. Bull. 58, 793–802. https://doi.org/10.1016/j.marpolbul.2009.02.008

Roberts, C.M., Hawkins, J.P., 2000. Fully-protected marine reserves: a guide. WWF Endangered Seas Campaign, 1250 24th Street, NW, Washington, DC 20037, USA and Environment Department, University of York, York, YO10 5DD, UK

Rodgers, K.S., Kido, M.H., Jokiel, P.L., Edmonds, T., Brown, E.K., 2012. Use of integrated landscape indicators to evaluate the health of linked watersheds and coral reef environments in the Hawaiian Islands. Environ. Manage. 50, 21–30. https://doi.org/10.1007/s00267-012-9867-9

Roos, N.C., Pennino, M.G., Carvalho, A.R., Longo, G.O., 2019. Drivers of abundance and biomass of Brazilian parrotfishes. Mar. Ecol. Prog. Ser. 623, 117–130. https://doi.org/10.3354/meps13005

Roth, F., Lessa, G., Wild, C., Kikuchi, R., Naumann, M., 2016. Impacts of a high-discharge submarine sewage outfall on water quality in the coastal zone of Salvador (Bahia, Brazil). *Marine Pollution Bulletin, 106*(1-2), 43-48. https://doi.org/10.1016/j.marpolbul.2016.03.048

Sachaniya, B.M., Gosai, H.B., Panseriya, H.Z., Vala, A.K., Dave, B.P., 2019. Polycyclic Aromatic Hydrocarbons (PAHs): Occurrence and Bioremediation in the Marine

Environment. In: Marine Ecology: Current and Future Developments. De-Sheng Pei & Muhammad Junaid (eds), Bentham EBooks imprint, pp. 435-466.

Santos, L.P.S., Serafini, P.P., 2020. Programa de monitoramento de aves marinhas arquipélago de Fernando de Noronha - relatório anual.

Santos, R.G., Machovsky-Capuska, G.E., Andrades, R., 2021. Plastic ingestion as an evolutionary trap: Toward a holistic understanding. Science 373, 56–60. https://doi.org/10.1126/science.abh0945

Shaver, E.C., Burkepile, D.E., Silliman, B.R., 2018. Local management actions can increase coral resilience to thermally-induced bleaching. Nat. Ecol. Evol. 2, 1075–1079. https://doi.org/10.1038/s41559-018-0589-0

Showkat, U., Ahmed, I., 2019. Study on the efficiency of sequential batch reactor (SBR)- based sewage treatment plant. Appl. Water Sci. 9, 1–10. https://doi.org/10.1007/s13201-018-0882-8

Silva, F.J.L., Silva Jr, J.., 2009. Circadian and seasonal rhythms in the behavior of spinner dolphins (*Stenella longirostris*). Mar. Mammal Sci. 25, 176–186. https://doi.org/10.1111/j.1748-7692.2008.00238.x

Silveira, J.C., Oliveira, F.S., Schaefer, C.E.G.R., Varajão, A.F.D.C., Varajão, C.A.C., Senra, E.O., 2020. Phosphatized volcanic soils of Fernando de Noronha Island, Brazil : Paleoclimates and landscape evolution. Catena 195, 104728. https://doi.org/10.1016/j.catena.2020.104728

Simpson, S.L., Batley, G.E., 2007. Predicting Metal Toxicity in Sediments: A Critique of Current Approaches. Integr. Environ. Assess. Manag. 3, 18–31.

Tebbett, S.B., Bellwood, D.R., Johnson, E.R., Chase, T.J., 2022. Occurrence and accumulation of heavy metals in algal turf particulates and sediments on coral reefs. Mar. Pollut. Bull. 184, 114113. https://doi.org/10.1016/j.marpolbul.2022.114113

Tilley, E., Ulrich, L., Lüthi, C., Reymond, P., Schertenleib, R., Zurbrügg, C., 2014. Compendium of Sanitation Systems and Technologies, 2nd ed. Swiss Federal Institute of Aquatic Science and Technology (Eawag), Dübendorf, Switzerland.

Tuholske, C., Halpern, B. S., Blasco, G., Villasenor, J. C., Frazier, M., & Caylor, K. (2021). Mapping global inputs and impacts from of human sewage in coastal ecosystems. PLoS ONE, 16(November):1–16. https://doi.org/10.1371/journal.pone.0258898

UNESCO 2001 Report of the twenty-fifth session of the World Heritage Committee (Helsinki, Finland, 11 - 16 December 2001). Available: https://whc.unesco.org/en/decisions/2319 assessed at August 02 2022.

Van Dam, J., Negri, A.P., Uthicke, S., Mueller, J., 2011. Chemical Pollution on Coral Reefs: Exposure and Ecological Effects, in: Sánchez-Bayo, F., van den Brink, P.J., Mann, R.M. (Eds.), Ecological Impacts of Toxic Chemicals. Bentham Science Publishers Ltd., Vol.9 pp. 187–211. https://doi.org/10.2174/978160805121210187

Wear, S.L., Acuña, V., McDonald, R., Font, C., 2021. Sewage pollution, declining ecosystem health, and cross-sector collaboration. Biol. Conserv. 255. https://doi.org/10.1016/j.biocon.2021.109010

Wear, S.L., Thurber, R.V., 2015. Sewage pollution : mitigation is key for coral reef stewardship. Ann. N. Y. Acad. Sci. 1355, 15–30. https://doi.org/10.1111/nyas.12785

Wenger, A.S., Fabricius, K.E., Jones, G.P., Brodie, J.E., 2015. Effects of sedimentation, eutrophication, and chemical pollution on coral reef fishes. Ecol. Fishes Coral Reefs 145–153. https://doi.org/10.1017/CBO9781316105412.017

Wooldridge, S.A., Done, T.J., 2009. Improved water quality can ameliorate effects of climate change on corals. Ecol. Appl. 19, 1492–1499.

Yu, S.-P., Cole, M., Chan, B.K.K., 2021. Review: Effects of microplastic on zooplankton survival and sublethal responses, in: Oceanography and Marine Biology. pp. 351–394.

Zamoner, J.B., Aued, A.W., Macedo-Soares, L.C.P., Picolotto, V.A.P., Garcia, C.A.E., Segal, B., 2021. Integrating Oceanographic Data and Benthic Community Structure Temporal Series to Assess the Dynamics of a Marginal Reef. Front. Mar. Sci. 8, 1–15. https://doi.org/10.3389/fmars.2021.762453

Zaneveld, J.R., Burkepile, D.E., Shantz, A.A., Pritchard, C.E., McMinds, R., Payet, J.P., Welsh, R., Correa, A.M.S., Lemoine, N.P., Rosales, S., Fuchs, C., Maynard, J.A., Thurber, R.V., 2016. Overfishing and nutrient pollution interact with temperature to disrupt coral reefs down to microbial scales. Nat. Commun. 7, 1–12. https://doi.org/10.1038/ncomms11833

Zhang, C., Zhou, H., Cui, Y., Wang, C., Li, Y., Zhang, D., 2019. Microplastics in offshore sediment in the Yellow Sea and East China Sea, China. Environ. Pollut. 244, 827–833. https://doi.org/10.1016/j.envpol.2018.10.102

Supplementary Material

Table S1: Limits of Detection and Quantification (mg kg^{-1}) for metals (Al, Cd, Cu, Cr, Fe, Ni, Pb, and Zn) and metalloid (As) analysed in sediment samples by ICP-OES.

Element	**LOD (mg kg^{-1})**	**LOQ (mg kg^{-1})**
Aluminum	0.201	0.603
Arsenic	0.392	1.176
Cadmium	0.050	0.151
Cooper	0.010	0.030
Chrome	0.040	0.121
Iron	0.080	0.241
Lead	0.121	0.362
Nickel	0.131	0.392
Zinc	0.010	0.030

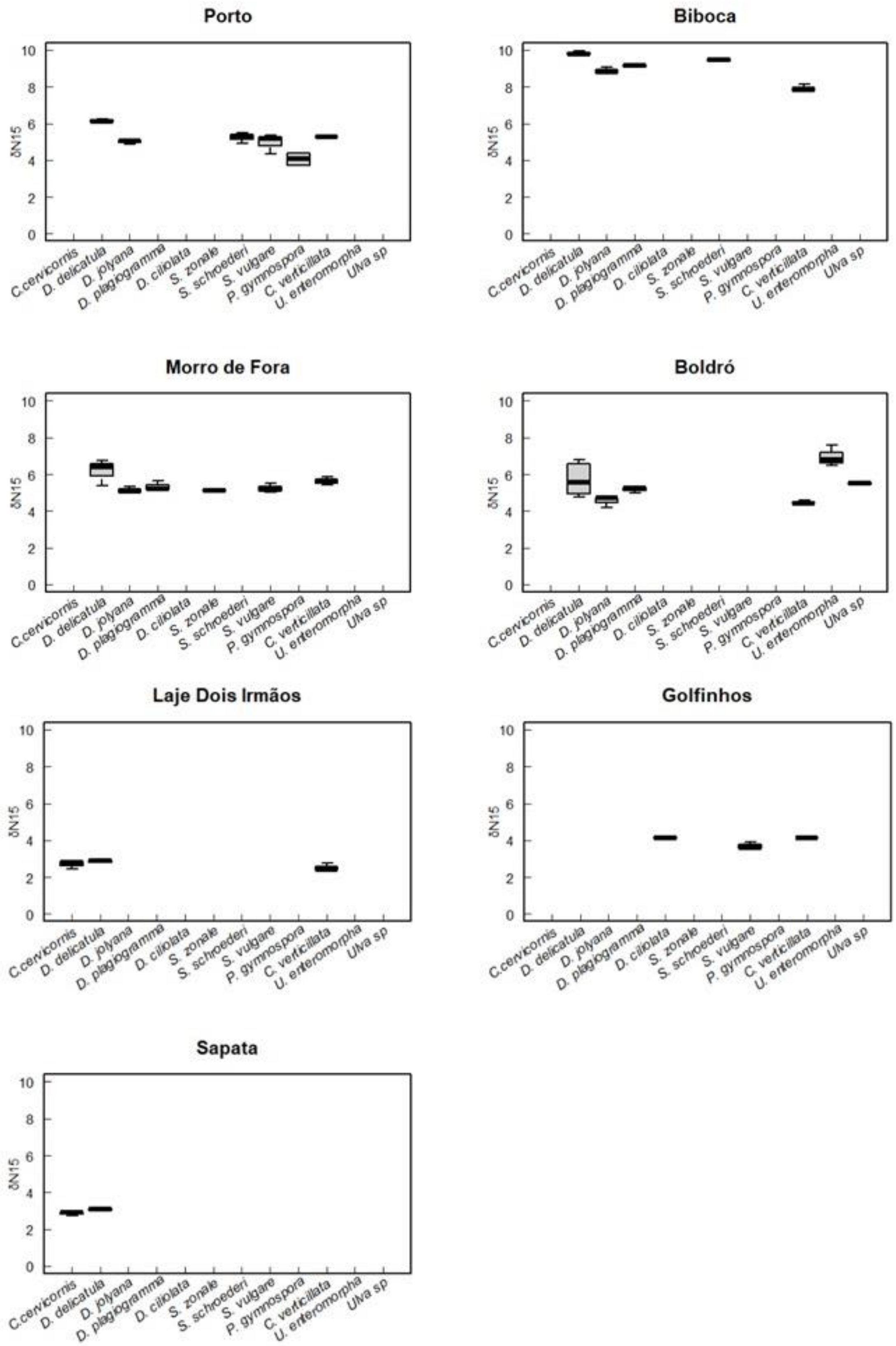


Figure S1: Isotopic signature ($\delta^{15}N$) of macroalgae in the leeward coast of Fernando de Noronha Archipelago.

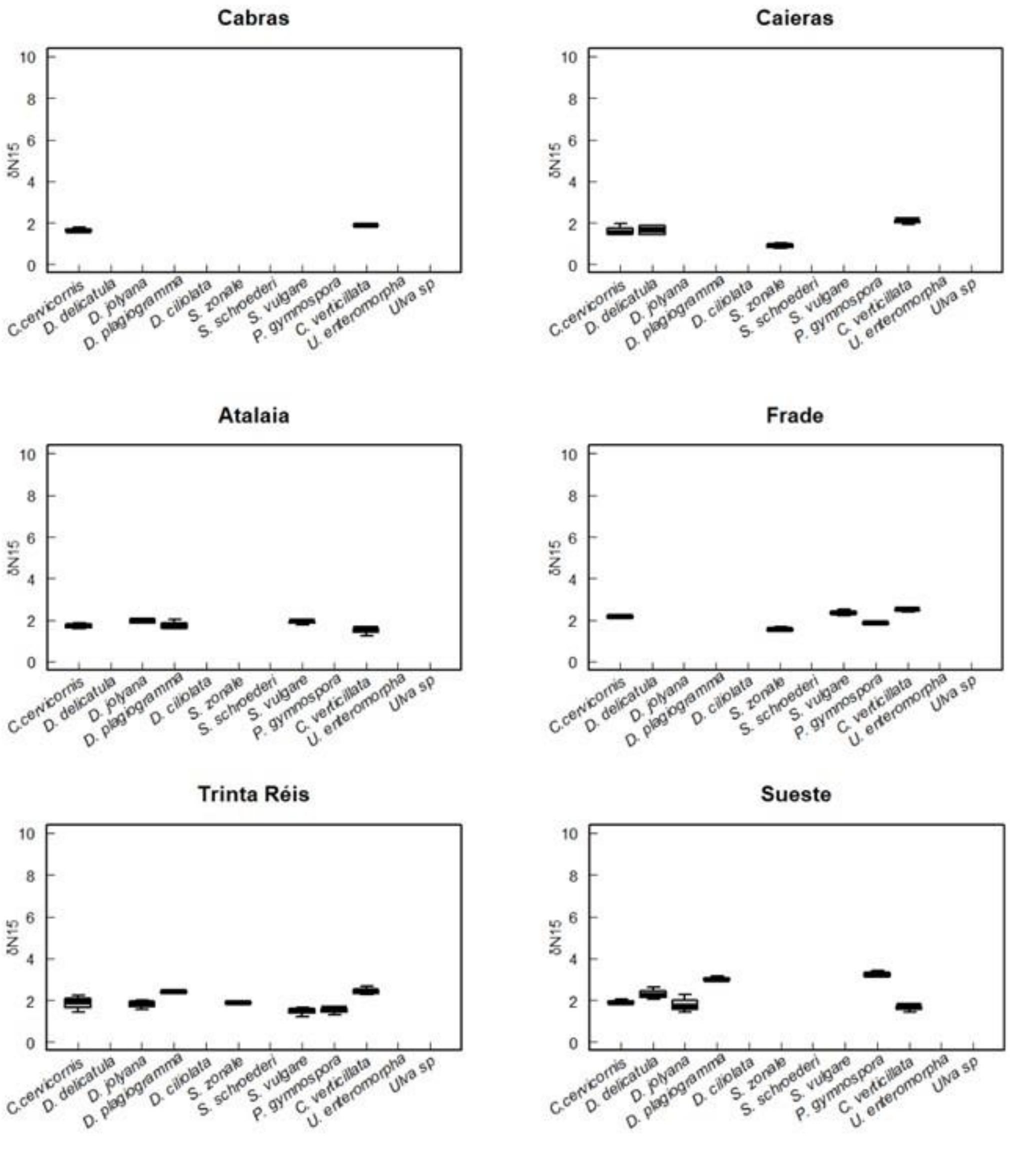


Figure S2: Isotopic signature ($\delta^{15}N$) of macroalgae in the windward coast of Fernando de Noronha Archipelago.

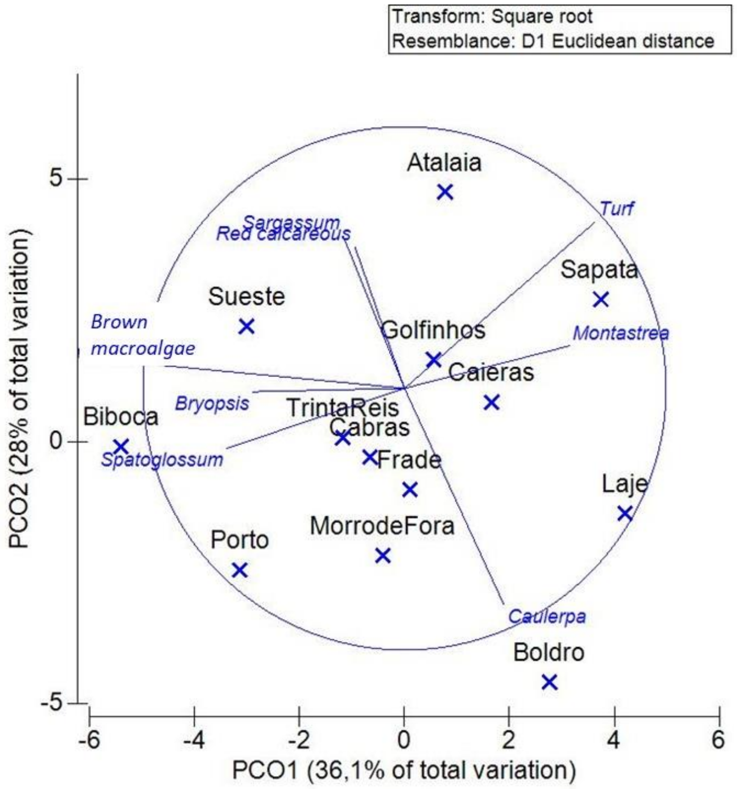


Figure S3: Principal coordinates analysis (PCO) of benthic cover. The vectors displayed represent the 8 benthic groups with the strongest correlation values to axes 1 and 2.

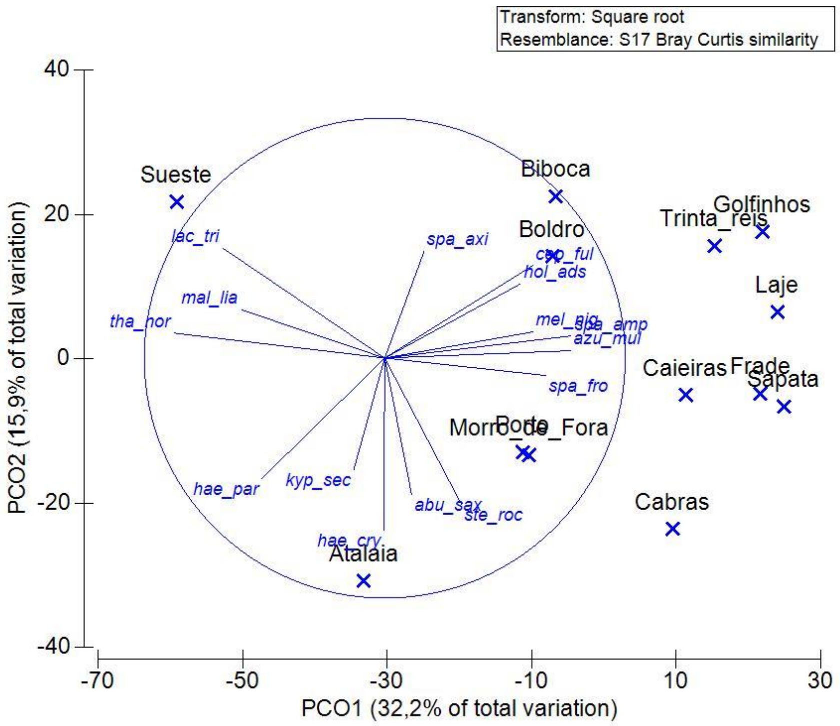


Figure S4: Principal coordinates analysis (PCO) of fish abundance. The vectors displayed represent the 15 species with the strongest correlation values to axes 1 and 2.

Figure S5: View of the creek that flows into Atalaia tidepool, with the presence of cattle.

Figure S6: View of Porto Beach, where vessel maintenance and refurbishment activities take place.

Figure S7: Oil spill at Porto Beach (red arrow), with the dispersion of an oil plume along the leeward coast.

CONCLUSÃO GERAL

Esta tese teve como objetivo avaliar múltiplos impactos locais e globais sobre ambientes marinhos de Fernando de Noronha, e o papel das UCs do arquipélago na mitigação destes impactos. Fernando de Noronha é protegido por duas UCs, uma Área de Proteção Ambiental (APA) e um Parque Nacional (PN) (denominados respectivamente de "multiple-use area" e "no-take area" nos capítulos da tese). Concluímos que, em relação aos aspectos estudados nesta tese, a APA não está cumprindo efetivamente todos os seus objetivos, devido principalmente ao número excessivo de pessoas em FN em relação à infraestrutura e tratamento de efluentes do arquipélago, portanto a APA atualmente não contribui plenamente para a conservação dos ambientes marinhos de FN. Além disso, os impactos gerados na APA afetam negativamente o PN, prejudicando a conservação dos ambientes marinhos também nesta UC. Assim, há uma necessidade urgente de fortalecimento da APA, particularmente na regulação da ocupação da Ilha e tratamento de efluentes.

Observamos que o crescimento urbano desordenado na APA gera impactos que levam à degradação da qualidade ambiental. O impacto mais severo observado foi a poluição nos ambientes recifais, porém também observamos a presença de lixo nas praias. Recifes e praias são os ambientes marinhos mais procurados por turistas em Fernando de Noronha, seja através do mergulho, lazer e contemplação. Assim, o lixo nesses ambientes coloca em risco alguns dos principais atrativos turísticos do arquipélago. O lixo encontrado nas praias da APA é gerado localmente, e é composto de embalagens de alimentos, plástico descartável, latas e garrafas de bebidas e bitucas de cigarro. Em 2018 foi publicado um Decreto Distrital em FN proibindo o uso e comercialização de descartáveis plásticos de uso único no arquipélago (Pernambuco 2018). Embora a medida possa contribuir para a redução da geração de lixo e seu depósito nas praias, ela não representa uma solução completa para o problema, uma vez que a maioria dos itens à venda nos estabelecimentos da ilha vem embrulhado em plástico (Pimentel *et al.* 2020). O Decreto foi promulgado logo após a conclusão da coleta de dados do Capítulo 1, o que torna os dados apresentados neste artigo uma referência relevante para avaliar a efetividade da medida para a redução do lixo nas praias. Sugerimos uma intensificação de pontos de coleta de lixos nas praias da APA e esforços diários de limpeza de praias ao final do dia, a fim de reduzir a concentração de resíduos sólidos nessas áreas e seu carreamento direto para o oceano.

Diferente do lixo visto nas praias da APA, a maior parte do lixo encontrado nas praias do PN é levado pelas correntes marinhas para a costa do mar de fora (face barlavento), embora também tenha sido encontrado lixo gerado localmente, como bitucas de cigarro no Sueste e nos Alagados da Raquel. No mar de fora encontramos principalmente lixo plástico leve, com alta flutuabilidade, como garrafas e tampas de garrafa, seringas e canudos. A presença de embalagens internacionais em estágio inicial de degradação indica que estes objetos podem ter sido descartados de navios ao largo do arquipélago. Além do macro lixo marinho, detectamos microplástico em todas as amostras de água e sedimento coletadas, com maiores concentrações de microplástico na água do mar de fora, principalmente nos pontos Cabras e Caieras. Uma parte do microplástico se deposita no sedimento, a depender dos padrões de correntes locais. A presença de plástico de diferentes tipos e tamanhos nas praias, coluna d'água e sedimento demostrando a conexão entre diferentes compartimentos ambientais em termos de poluição por plástico (Monteiro *et al.* 2018). A vulnerabilidade desta região do PN à poluição por plástico demonstra que a distância da costa e a proteção legal não impedem que os impactos antrópicos atinjam o PN. Além do lixo marinho, o PN também está vulnerável a outros poluentes trazidos pelas correntes como óleo fruto de vazamentos (Magalhães *et al*. 2022). Assim, solucionar este problema é complexo pois depende de políticas internacionais de combate ao lixo marinho, no entanto sugerimos que haja monitoramento e limpeza periódica dessas praias de difícil acesso para reduzir o acúmulo de resíduos e degradação local.

Além do plástico, nós avaliamos uma série de outros poluentes na água e nos sedimentos dos ambientes recifais de FN. Embora a qualidade da água na maior parte dos recifes estivesse satisfatória no momento da amostragem, provavelmente devido à intensa dinâmica oceânica ou à assimilação biológica dos nutrientes, a qualidade dos efluentes das Estações de Tratamento de Esgoto – ETE Boldró e Biboca não estava em conformidade com o estabelecido na legislação. Usando análise de assinatura isotópica, verificamos sinais de enriquecimento por nitrogênio proveniente de esgoto em macroalgas coletadas em todos os sítios amostrais da APA, indicando poluição crônica nestes sítios. A comunidade bentônica nestes locais diferiu dos locais menos impactados. Dentre outras características, destacaram-se a maior biomassa de macroalgas e dominância de macroalgas de crescimento rápido. Estavam presentes também macroalgas indicadoras de poluição, como *Bryopsis* sp. e *Ulva* sp.. Além disso, há indícios de que nutrientes, metais e microplásticos originários da APA podem ser transportados para o PN por meio de correntes costeiras, especialmente no mar de dentro (face sotavento). Um exemplo disso é a Baía dos Golfinhos, área do PN no mar de dentro onde a visitação não é permitida. Neste local observamos sinais de enriquecimento por nitrogênio, concentrações altas de metais pesados, e a maior concentração de microplásticos no sedimento dentre todos os locais amostrados no arquipélago. Isso indica que a falta de infraestrutura adequada para tratamento de efluentes na APA impacta uma das áreas mais relevantes para a conservação do PN.

Outro ponto crítico detectado neste estudo foi a contaminação na área do Porto, onde há grande quantidade de lixo descartado próximo ao píer, indicativos de despejo de esgoto sem tratamento, e poluição por HPAs - Hidrocarbonetos Policíclicos Aromáticos (indicadores de poluição por óleos combustíveis e lubrificantes de motores). Além da ausência de saneamento adequado, essa contaminação provavelmente é resultado das atividades de reforma e manutenção de embarcações que são realizadas neste local sem que haja nenhuma infraestrutura para tal, e sem controle sobre o descarte adequado dos produtos químicos potencialmente perigosos utilizados. Adicionalmente, grandes quantidades de óleo diesel e outros combustíveis destinados ao abastecimento da usina de geração de energia e outras necessidades da ilha são desembarcados nessa área, e a estrutura para evitar acidentes e vazamentos é limitada.

Assim, dentre os problemas identificados neste estudo, o tratamento de efluentes e a contaminação por HPAs merecem atenção prioritária. É imprescindível ampliar e melhorar a infraestrutura de tratamento de esgoto de FN, aprimorar a eficácia dos processos de tratamento de águas residuais e garantir o cumprimento das normas regulatórias para mitigar o impacto negativo no ecossistema marinho tanto da APA quanto do PN. O planejamento da infraestrutura de esgoto deve considerar a implementação de métodos de tratamento terciário para reduzir a entrada de nutrientes e prevenir a eutrofização, e considerar a utilização de estações de tratamento descentralizadas (Tilley *et al.* 2014). Em relação ao Porto, é necessário implementar medidas de controle e estrutura adequada para manutenção das embarcações, que são vitais para a dinâmica socioeconômica da Ilha, para que essas atividades deixem de contaminar o ambiente de entorno que é amplamente utilizado por turistas e população local.

O turismo atualmente é um vetor de degradação ambiental em FN pelo fato de o número de visitantes estar além da capacidade de suporte da ilha. Por outro lado, este estudo demonstra que é possível reduzir substancialmente o impacto do turismo desde que seja realizado de forma controlada, monitorada e dentro dos limites de capacidade, como ocorre na piscina do Atalaia, recife estudado no Capítulo 2. Além deste recife, normas similares de visitação se aplicam a outras duas poças de maré do PN (Abreu e Caiera). Embora sejam ambientes de alta relevância ecológica, elas abrangem apenas uma pequena área do PN. No restante da área marinha desta UC, a restrição do número de visitantes é feita por meio do controle do número de embarcações que acessam os locais. No entanto, há uma pressão constante por parte do setor turístico para ampliar o número de embarcações autorizadas a visitar os pontos turísticos e locais de mergulho autônomo do Parque Nacional. Isso ocorre porque, assim como o número de moradores, turistas e veículos, o número de embarcações na ilha também cresceu vertiginosamente nos últimos anos. Diante desse cenário, é fundamental garantir que o número de visitantes permaneça dentro dos limites de capacidade desses atrativos. Para isso, é necessário monitorar e atualizar o estudo de capacidade de carga das atividades de turismo náutico (ICMBio 2009) e ajustar as autorizações de acesso das embarcações de acordo com as regulamentações legais. Além disso, é importante monitorar e avaliar a necessidade de limitar o número de turistas que acessam o PN por via terrestre, caso seja necessário.

Apesar de haver impacto humano local no PN, especialmente no mar de dentro, os ambientes marinhos do PN estão em melhor estado de conservação em comparação à APA. Isso demonstra que, apesar de haver problemas, o PN tem sido relativamente eficiente em contribuir com a conservação dos ambientes marinhos de FN. No entanto, destacamos desafios a manutenção da efetividade do PN observadas neste estudo relacionadas às mudanças climáticas globais, e à poluição gerada em fontes distantes e trazida pelas correntes, especialmente no mar de fora e na Baía dos Golfinhos. Verificamos que o principal fator afetando a saúde dos corais na piscina do Atalaia, foi a anomalia de temperatura da água do mar, resultando em branqueamento, palidez e outras alterações de coloração nos corais, embora não tenha sido observada mortalidade significativa dos corais nesse local provavelmente porque outros impactos locais, como o da visitação, estejam bem controlados. No entanto, eventos consecutivos de branqueamento podem diminuir a resiliência e resistência destes organismos, tornando-os mais vulneráveis a outros estressores, sejam naturais, como o soterramento, ou antropogênicos, como a poluição e eventos futuros de aquecimento.

Embora o aquecimento dos oceanos e outras ameaças globais estejam além do controle dos gestores das AMPs, existem medidas que podem ser tomadas localmente para aumentar a resistência e resiliência dos ecossistemas recifais e reduzir a perda de biodiversidade resultante da interação entre os impactos em diferentes escalas (Brown *et al.* 2013, Shaver *et al.* 2018). Controlar os impactos locais garantindo a qualidade da água e manejando a pesca são exemplos de medidas que promovem a resiliência, e podem proteger os ecossistemas recifais da mortalidade associada ao aumento da temperatura (Zaneveld *et al.* 2016).

Áreas isoladas e com problemas localizados, como é o caso de FN, representam oportunidades únicas para implementação de ações de manejo eficazes e oportunas para mitigar impactos em diferentes escalas (Sandin *et al.* 2008), especialmente quando comparadas a ambientes costeiros densamente povoados. Considerando a alta relevância de FN para a conservação marinha, evidenciada pela existência das duas AMPs, seu status como Patrimônio Marinho da Humanidade, e a importância do ambiente conservado em uma economia baseada no turismo, é paradoxal que recursos simples e essenciais para a saúde do ecossistema e das pessoas, como é o caso do tratamento do esgoto, não sejam garantidos e priorizados pela gestão.

Neste trabalho comprovamos a importância do monitoramento para subsidiar a gestão de AMPs. O monitoramento já é uma prática adotada pelo órgão gestor das áreas protegidas federais no Brasil, o ICMBio – Instituto Chico Mendes de Conservação da Biodiversidade, que possibilitou as análises realizadas nos capítulos 1 e 2 da tese. O mapeamento e as análises empregados no capítulo 3 têm o potencial de serem largamente utilizados em outras AMPs, devido ao seu baixo custo e facilidade de aplicação. O monitoramento da assinatura isotópica em macroalgas, em particular, se mostrou um método direto e eficaz para verificar o impacto do esgoto em ambientes recifais em AMPs, permitindo identificar onde estão os problemas e qual a sua magnitude. Essa informação é essencial para direcionar os esforços para minimizar o impacto na poluição, além de servir como embasamento para demandar medidas corretivas por parte das instituições e empresas responsáveis.

No contexto específico de FN, recomendamos a expansão do programa de monitoramento de recifes, já existente no PN, para a APA. Essa medida é justificada pelo fato de que é na APA onde encontramos maiores evidências de impacto, e é importante monitorar a evolução da situação para avaliar a efetividade de eventuais medidas para mitigar os impactos ou o agravamento dos impactos através da comparação com ambientes controles (PN) e do acompanhamento temporal das mesmas localidades. A falta de informações de longo prazo sobre os ecossistemas recifais na APA dificulta a compreensão das tendências temporais e espaciais. Adicionalmente, sugerimos realizar uma avaliação específica e manter o monitoramento dos sedimentos nos locais onde a contaminação por HPAs foi identificada.

Além dos impactos discutidos nesta tese, há uma série de outros impactos que, em conjunto, levaram a IUCN – União Internacional para Conservação da Natureza a classificar o estado de conservação do Sítio do Patrimônio da Humanidade composto por Fernando de Noronha e Atol das Rocas como "preocupação significativa" na última avaliação realizada em 2020 (IUCN 2020). O crescimento do turismo e o desenvolvimento urbano são as principais ameaças apontadas pela IUCN, aos quais se juntam a pesca industrial nas proximidades do arquipélago, espécies exóticas invasoras no ambiente terrestre (Mello & Oliveira 2016, Micheletti *et al.* 2020) e marinho (Luiz *et al.* 2021), mudanças climáticas e elevação do nível do mar.

Neste estudo, realizamos uma avaliação abrangente dos impactos antrópicos, tanto locais quanto globais, nos ambientes marinhos ao redor de todo o Arquipélago de Fernando de Noronha. Investigamos questões como a presença de macrolixo nas praias, poluição por esgoto, metais, HPAs e microplásticos nos recifes, bem como a efetividade do manejo do turismo e o impacto do aquecimento das águas numa poça de maré. As evidências obtidas neste trabalho se somam aos resultados de estudos anteriores, que também destacaram os impactos antrópicos relacionados à poluição em FN (por ex: Braga *et al.* 2018, Carvalho *et al*. 2021, Combi *et al.* 2022, Cristiano *et al.* 2020, Ivar do Sul *et al.* 2009, Justino *et al.* 2023, Monteiro *et al.* 2020, Prazeres *et al.* 2012 entre outros).

Esses estudos se alinham ao resultado da avaliação realizada pela UNESCO, e revelam que o desenvolvimento de FN para sustentar o turismo em expansão está resultando em um paradoxo, comprometendo os recursos naturais que são o principal atrativo turístico da região e colocando em risco a sustentabilidade da atividade em longo prazo. Desde 2009, o Estudo de Capacidade de Suporte já alertava para o fato de que o desenvolvimento turístico estava à beira de um colapso, ultrapassando os limites aceitáveis de risco em diversos aspectos (Watson 2011). A menos que medidas urgentes, como as sugeridas nesta tese, sejam adotadas para compatibilizar o número de visitantes com a infraestrutura disponível, e que um modelo sustentável de desenvolvimento seja implementado, a biodiversidade do arquipélago segue seriamente ameaçada.

Referências

Abessa, D.M.S., Albuquerque, H.C., Morais, L.G., Araújo, G.S., Fonseca, T.G., Cruz, A.C.F., Campos, B.G., Camargo, J.B.D.A., Gusso-Choueri, P.K., Perina, F.C., Choueri, R.B., Buruaem, L.M., 2018. Pollution status of marine protected areas worldwide and the consequent toxic effects are unknown. Environ. Pollut. https://doi.org/10.1016/j.envpol.2018.09.129

Braga, E.D.S., Chiozzini, V.G., Berbel, G.B.B., 2018. Oligotrophic water conditions associated with organic matter regeneration support life and indicate pollution on the western side of Fernando de Noronha Island - NE, Brazil (3°S). Brazilian J. Oceanogr. 66, 73–90. https://doi.org/10.1590/s1679-87592018148306601

Brown, C.J., Saunders, M.I., Possingham, H.P., Richardson, A.J., 2013. Managing for Interactions between Local and Global Stressors of Ecosystems. PLoS One 8, 1–10. https://doi.org/10.1371/journal.pone.0065765

Carvalho, J.P.S., Silva, T.S., Costa, M.F., 2021. Distribution , characteristics and short-term variability of microplastics in beach sediment of Fernando de Noronha Archipelago, Brazil. Mar. Pollut. Bull. 166, 112212. https://doi.org/10.1016/j.marpolbul.2021.112212

Combi, T., Montone, R.C., Corada-Fernandez, C., Lara-Martín, P.A., Gusmao, J.B., Santos, M.C. de O., 2022. Persistent organic pollutants and contaminants of emerging concern in spinner dolphins (Stenella longirostris) from the Western Atlantic Ocean. Mar. Pollut. Bull. 174, 113263. https://doi.org/10.1016/j.marpolbul.2021.113263

Cristiano, C., Camboim, G., Carla, L., 2020. Beach landscape management as a sustainable tourism resource in Fernando de Noronha Island (Brazil). Mar. Pollut. Bull. 150, 110621. https://doi.org/10.1016/j.marpolbul.2019.110621

IUCN, 2020. Brazilian Atlantic Islands: Fernando de Noronha and Atol das Rocas Reserves - 2020 Conservation outlook assessment.

Ivar do Sul, J.A., Spengler, Â., Costa, M.F., 2009. Here, there and everywhere. Small plastic fragments and pellets on beaches of Fernando de Noronha (Equatorial Western Atlantic). Mar. Pollut. Bull. 58, 1236–1238. https://doi.org/10.1016/j.marpolbul.2009.05.004

Justino, A.K.S., Ferreira, G.V.B., Fauvelle, V., Schmidt, N., Lenoble, V., Pelage, L., Martins, K., Travassos, P., Lucena-Frédou, F., 2023. From prey to predators:

Evidence of microplastic trophic transfer in tuna and large pelagic species in the southwestern Tropical Atlantic. Environ. Pollut. 327. https://doi.org/10.1016/j.envpol.2023.121532

Luiz, O.J., 2009. Estudo de capacidade de carga e operacionalização das atividades de turismo náutico No Parque Nacional Marinho de Fernando de Noronha. Brasília.

Luiz, O.J., dos Santos, W.C.R., Marceniuk, A.P., Rocha, L.A., Floeter, S.R., Buck, C.E., de Klautau, A.G.C.M., Ferreira, C.E.L., 2021. Multiple lionfish (Pterois spp.) new occurrences along the Brazilian coast confirm the invasion pathway into the Southwestern Atlantic. Biol. Invasions 23, 3013–3019. https://doi.org/10.1007/s10530-021-02575-8

Magalhães, K.M., Rosa Filho, J.S., Teixeira, C.E.P., Coelho-Jr, C., Lima, M.C.S., Costa Souza, A.M., Soares, M.O., 2022. Oil and plastic spill: 2021 as another challenging year for marine conservation in the South Atlantic Ocean. Mar. Policy 140. https://doi.org/10.1016/j.marpol.2022.105076

Mello, T.J., Oliveira, A.A. de, 2016. Making a Bad Situation Worse: An Invasive Species Altering the Balance of Interactions between Local Species. PLoS One 11, e0152070. https://doi.org/10.1371/journal.pone.0152070

Micheletti, T., Fonseca, F.S., Mangini, P.R., Serafini, P.P., Krul, R., Mello, T.J., Freitas, M.G., Dias, R.A., Silva, J.C.R., Marvulo, M.F. V., Araujo, R., Gasparotto, V.P.O., Abrahão, C.R., Rebouças, R., Toledo, L.F., Siqueira, P.G.S.C., Duarte, H.O., Moura, M.J.C., Fernandes-Santos, R.C., Russell, J.C., 2020. Terrestrial invasive species on Fernando de Noronha Archipelago: What we know and the way forward, in: Vinícius Londe (Ed.), Invasive Species: Ecology, Impacts, and Potential Uses. Nova Science Publishers, pp. 51–94.

Monteiro, R.C.P., Ivar, J.A., Costa, M.F., 2018. Plastic pollution in islands of the Atlantic Ocean. Environ. Pollut. 238, 103–110. https://doi.org/10.1016/j.envpol.2018.01.096

Monteiro, R.C.P., Do Sul, J.A.I., Costa, M.F., 2020. Small microplastics on beaches of Fernando de Noronha island, tropical atlantic ocean. Ocean Coast. Res. 68, 1–10. https://doi.org/10.1590/S2675-28242020068235

Pernambuco 2018. Decreto Distrital Nº 002, de 12 de dezembro de 2018.

Pimentel, C.R., Rocha, L.A., Shepherd, B., Phelps, T.A.Y., Joyeux, J., Martins, A.S., Stein, C.E., Teixeira, J.B., Gasparini, J.L., Reis-filho, J.A., Garla, R.C., Francini-filho, R.B., Delfino, S.D.T., Mello, T.J., Giarrizzo, T., Pinheiro, H.T., 2020. Mesophotic ecosystems at Fernando de Noronha Archipelago , Brazil (South-western Atlantic), reveal unique ichthyofauna and need for conservation. Neotrop. Ichthyol. 4, e200050. https://doi.org/https://doi.org/10.1590/1982-0224-2020-0050

Prazeres, M.F., Martins, S.E., Bianchini, A., 2012. Assessment of water quality in coastal waters of Fernando de Noronha, Brazil: biomarker analyses in *Amphistegina lessonii*. J. Foraminifer. Res. 42, 56–65.

Sandin, S.A., Smith, J.E., DeMartini, E.E., Dinsdale, E.A., Donner, S.D., Friedlander, A.M., Konotchick, T., Malay, M., Maragos, J.E., Obura, D., Pantos, O., Paulay, G., Richie, M., Rohwer, F., Schroeder, R.E., Walsh, S., Jackson, J.B.C., Knowlton, N., Sala, E., 2008. Baselines and degradation of coral reefs in the Northern Line Islands. PLoS One 3. https://doi.org/10.1371/journal.pone.0001548

Shaver, E. C., Burkepile, D. E., & Silliman, B. R. 2018. Local management actions can increase coral resilience to thermally-induced bleaching. Nature Ecology & Evolution, 2(7), 1075-1079. https://doi.org/10.1038/s41559-018-0589-0

Tilley, E., Ulrich, L., Lüthi, C., Reymond, P., Schertenleib, R., Zurbrügg, C., 2014. Compendium of sanitation systems and technologies. 2nd revised Edition, 2nd ed. Swiss Federal Institute of Aquatic Science and Technology (Eawag), Dübendorf, Switzerland.

Watson, A. 2011. Fernando de Noronha, o paraíso ameaçado. O Eco. Disponível em https://oeco.org.br/reportagens/25011-fernando-de-noronha-o-paraiso-ameacado/ . Acesso em 08/06/2023.

Zaneveld, J.R., Burkepile, D.E., Shantz, A.A., Pritchard, C.E., McMinds, R., Payet, J.P., Welsh, R., Correa, A.M.S., Lemoine, N.P., Rosales, S., Fuchs, C., Maynard, J.A., Thurber, R.V., 2016. Overfishing and nutrient pollution interact with temperature to disrupt coral reefs down to microbial scales. Nat. Commun. 7, 1–12. https://doi.org/10.1038/ncomms11833