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A blueprint for securing Brazil's marine biodiversity and supporting the achievement of global conservation goals

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Abstract

Aim: As a step towards providing support for an ecological approach to strengthening marine protected areas (MPAs) and meeting international commitments, this study combines cumulative impact assessment and conservation planning approach to undertake a large-scale spatial prioritization.

Location: Exclusive Economic Zone (EEZ) of Brazil, Southwest Atlantic Ocean.

Methods: We developed a prioritization approach to protecting different habitat types, threatened species ranges and ecological connectivity, while also mitigating the impacts of multiple threats on biodiversity. When identifying priorities for conservation, we accounted for the co-occurrence of 24 human threats and the distribution of 161 marine habitats and 143 threatened species, as well as their associated vulnerabilities. Additionally, we compared our conservation priorities with MPAs proposed by local stakeholders.

Results: We show that impacts to habitats and species are widespread and identify hot spots of cumulative impacts on inshore and offshore areas. Industrial fisheries, climate change and land-based activities were the most severe threats to biodiversity. The highest priorities were mostly found towards the coast due to the high cumulative impacts found in nearshore areas. As expected, our systematic approach showed a better performance on selecting priority sites when compared to the MPAs proposed by local stakeholders without a typical conservation planning exercise, increasing the existing coverage of MPAs by only 7.9%. However, we found that proposed MPAs still provide some opportunities to protect areas facing high levels of threats.

Main conclusions: The study presents a blueprint of how to embrace a comprehensive ecological approach when identifying strategic priorities for conservation. We advocate protecting these crucial areas from degradation in emerging conservation efforts is key to maintain their biodiversity value.

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1 | INTRODUCTION

With rapidly increasing human pressure on marine ecosystems, the design and implementation of marine protected areas (MPAs) has emerged as a cornerstone of management strategies for restoring biodiversity after disturbance events (Di Minin & Toivonen, 2015; Giakoumi et al., 2017; Lubchenco & Grorud-Colvert, 2015). However, while several studies have documented that ecological effects of MPAs depend on a well-designed and connected system of protected areas (e.g. Gill et al., 2017; Green et al., 2015; Lester et al., 2009; Roberts et al., 2018), many aspects of their design (e.g. location, size, spacing) are still decided based on opportunity or political interests (Devillers et al., 2015; Magris & Pressey, 2018). Such opportunistic placement of contemporary MPAs might undermine their long-term effectiveness for conserving biodiversity and wider ecosystem functions. Thus, conservation planning (Margules & Pressey, 2000), a framework for delimiting protection priorities based on principles of ecology, is fundamental to help create effective MPAs, and optimize resources and the diverse roles expected from them.

Conservation planning has become the most influential framework for identifying spatial priorities accounting for representation of biodiversity patterns (Kukkala & Moilanen, 2013), ecological connectivity (Beger et al., 2010), along with vulnerabilities and threats to these assets (Tulloch et al., 2015). The well-known principle of representation (i.e. sampling of each biodiversity feature of interest in MPAs) forms the basis of mathematical algorithms for MPA siting (e.g. Marxan, Zonation) (Possingham et al., 2000). Previous studies have also developed advanced methods to integrate ecological connectivity into planning for the selection of marine conservation priorities (Daigle et al., 2020; Magris et al., 2016; Weeks, 2017), given that this ecological principle is a means by which to achieve biological persistence (Gaines et al., 2010; Magris, et al., 2018). Conservation planning also enables the mitigation of impacts from human activities on biodiversity (e.g. Boon & Beger, 2016; Klein et al., 2013; Mazor et al., 2014), a task that is essential to strengthen ecosystem resilience and restore the natural assets for which MPA design is undertaken. While this body of research includes important conceptual and methodological advances, operational challenges remain, and applications to the real-world conservation planning are rare.

Conservation planning is also widely pursued as a tool to inform decision-making towards achieving commitments for marine conservation, including the post-2020 targets under the Convention on Biological Diversity (CBD) and the United Nations Sustainable Development Goals (SDGs) (Claudet et al., 2020; Tittensor et al., 2019; Visconti et al., 2020). For instance, world governments have collectively committed to expand MPA coverage over areas of particular importance for biodiversity and ecosystem services (codified as Target 11 in the CBD Strategic Plan 2011–2020). This expansion could be achieved by creating new MPAs in habitats that have experienced extensive and rapid biodiversity loss (Target 5) or by establishing new MPAs in areas where they could contribute to reducing species' extinction risk (Target 12) (Laffoley et al., 2017). Further, these efforts for conserving marine biodiversity based on best-practice planning principles are a prerogative to achieve the SDG 14 by 2030 (Sala et al., 2018; United Nations, 2015). In parallel, governmental agencies that foster conservation in a variety of locations across a country are also committed to identifying political opportunities to implement new, locally established MPAs, including through the use of conservation planning (Fox et al., 2013; Gleason et al., 2013). Although initiatives at global and local scales might not always be aligned (Weeks et al., 2010), they provide an unprecedented opportunity to establish comprehensive and coordinated conservation plans, thus contributing to the conditions under which MPAs will deliver the desired benefits.

Facing a future of intensifying vulnerability to anthropogenic activities (Halpern et al., 2019), effective conservation of marine biodiversity requires a strategic planning approach to help identifying locations where multiple threats occur together with biodiversity components in the same area and result in cumulative human impact (Crain et al., 2009; Halpern et al., 2008; Micheli et al., 2013). The few examples that have used multiple threats in spatial prioritization have used data developed at global scales (e.g. Boon & Beger, 2016) or devised methods that targeted areas with reduced human impacts (e.g. Klein et al., 2013). To better identify areas for potential protection, we need fine-scale mapping of human impacts for detailed conservation planning within a framework concerned with the mitigation of cumulative threats. Hence, in this study, we explore how cumulative impact assessment could be combined with conservation planning to identify priorities based on the severity of threats they face.

To address this challenge, we developed a prioritization approach as a step towards providing support for new MPAs in Brazil and help the achievement of global conservation goals. Previous prioritization analyses identifying new areas for protection, although informative, have focused on particular habitats (e.g. Almada & Bernardino, 2017; Magris et al., 2016) or taxonomic groups (e.g. Patrizzi & Dobrovolski, 2018; Vilar et al., 2020), or performed regional-scale planning exercises (e.g. Duarte de Paula Costa et al., 2018; Magris et al., 2015). Thus far, there has been a lack of nation-wide conservation assessment integrating fine-scale data on biodiversity distribution, multiple threats to biodiversity and ecological connectivity. Our approach allows the explicit incorporation of widely recognized conceptual design principles for MPA establishment: (a) protection of the full range of biological diversity and the associated oceanographic environment while ensuring that threatened components of biodiversity are represented within conservation priorities; (b) maintenance of spatially structured ecological processes of connectivity; and (c) mitigation of the effects of human threats on marine biodiversity, accounting for variation in levels of cumulative impact across all components of biodiversity. Additionally, we compared the priority areas identified by our method against MPAs proposed by stakeholder groups in Brazil in terms of conservation importance.

2 | METHODS

2.1 | Case study description

We assembled a large data set of the marine biodiversity and threats within the Exclusive Economic Zone (EEZ) of Brazil (3,642,070 km^2; Figure 1a) to identify priority areas for conservation using integer linear programming. The region encompasses distinct marine ecosystems including the most extensive rhodolith bed in the world (Amado-Filho et al., 2012), the second largest mangrove area on the globe (Spalding et al., 2010), the largest coral reef area in the South Atlantic (Leão et al., 2019), the mesophotic reefs of the Amazon River mouth (Moura et al., 2016) and a large but underestimated area of seagrass meadows (Copertino et al., 2016). Because conservation of marine biodiversity has long lagged far behind protection efforts for terrestrial realms, the Brazilian Ministry of the Environment sharply increased the total coverage of MPAs from 1.5% to 24.5%, following an uninformed and opportunistic designation process (Giglio et al., 2018; Magris & Pressey, 2018). However, the total extent of no-take MPAs (i.e. considered fully protected, which refers to the IUCN categories Ia and b, II and III) remained low with only 2.5% of the Brazilian EEZ covered by this type of MPA (Figure 1a). The Brazilian Ministry of the Environment and Policy Directives calls for an increase in the coverage of no-take MPAs beyond 10% of Brazilian jurisdictional marine area within the next 15 years (Maretti et al., 2019). Despite institutional uncertainties in conservation management due to the political state of affairs (Azevedo-Santos et al., 2017), the national agency for conservation has developed spatial plans for biodiversity conservation in collaboration with local stakeholders, hereafter the "proposed MPAs" (Figure 1a). Given the existence of several MPA proposals in various stages of development, the federal government is in an ongoing decision-making process of determining the most important areas for additional conservation action.

Due to the difference in the resolution of data to represent biodiversity and human attributes across a large study region and computational limitations, we used two different sizes of planning units (i.e. spatial units that could be potentially selected for protection). We superimposed a grid of square cells on deep-water areas (>200 m) with a 10 x10 km resolution and on shallow-water areas (<200 m) with a 1x1 km resolution to implement our conservation prioritization (N = 67,323 planning units). Towards the mainland, we restricted our analyses to the upper limit of the occurrence of mangroves.

2.2 | Data layers

2.2.1 | Habitat data

We conceptualized a GIS-based habitat map (also often labelled "ecosystems") at a national scale by assembling data from peer-reviewed literature, publicly available and unpublished data sets, governmental and non-governmental reports, and performed complementary GIS analysis (Appendix S1). Recognizing the spatial structure of the marine environment, we developed separate pelagic (N = 11) and benthic habitats (N = 150) to account for different types and resolution of available data. The benthic habitats were delineated following a nested hierarchical classification scheme as a result of specific combinations of ecoregions (sensu Spalding et al., 2007), depth zones, seascape units (delineation of the seabed into hard, soft or mixed substrate types) and within-habitat specificities (in places where more detailed data were available). Whenever possible, we obtained digital maps containing the extension of marine habitats (e.g. nearshore banks of coral reefs); however, some spatial data were manually digitized and inserted in the GIS (e.g. seagrass meadows). We assigned unique code identifiers, names and descriptions to the marine habitats (Table S1). We used expert opinion within the authors to revise the draft habitats produced by the analytic steps described in Appendix S1. Benthic habitats ranged from coastal to abyssal environments and included mangrove forests (4 classes), beaches and sand dunes (5 classes), seagrass meadows (8 classes), estuaries (5 classes), coral reefs (11 classes), mesophotic reefs (4 classes), seamounts (18 classes), rhodolith beds (i.e. crustose coralline algae, 6 classes), submarine canyons (11 classes), and deep-sea coral habitats (i.e. cold-water coral reefs, 5 classes). The pelagic habitats were classified based on a cluster analysis of ecological data that serve as surrogates for assemblages of pelagic species. Descriptions of each physical environmental characteristics underpinning each pelagic habitat are summarized in Table S2.

2.2.2 | Threatened species

We obtained species distribution ranges and assessment of identified threats available for 143 animal species (invertebrates, fishes, mammals, turtles and seabirds) listed under national legislation with a status of Critically Endangered, Endangered or Vulnerable. Range

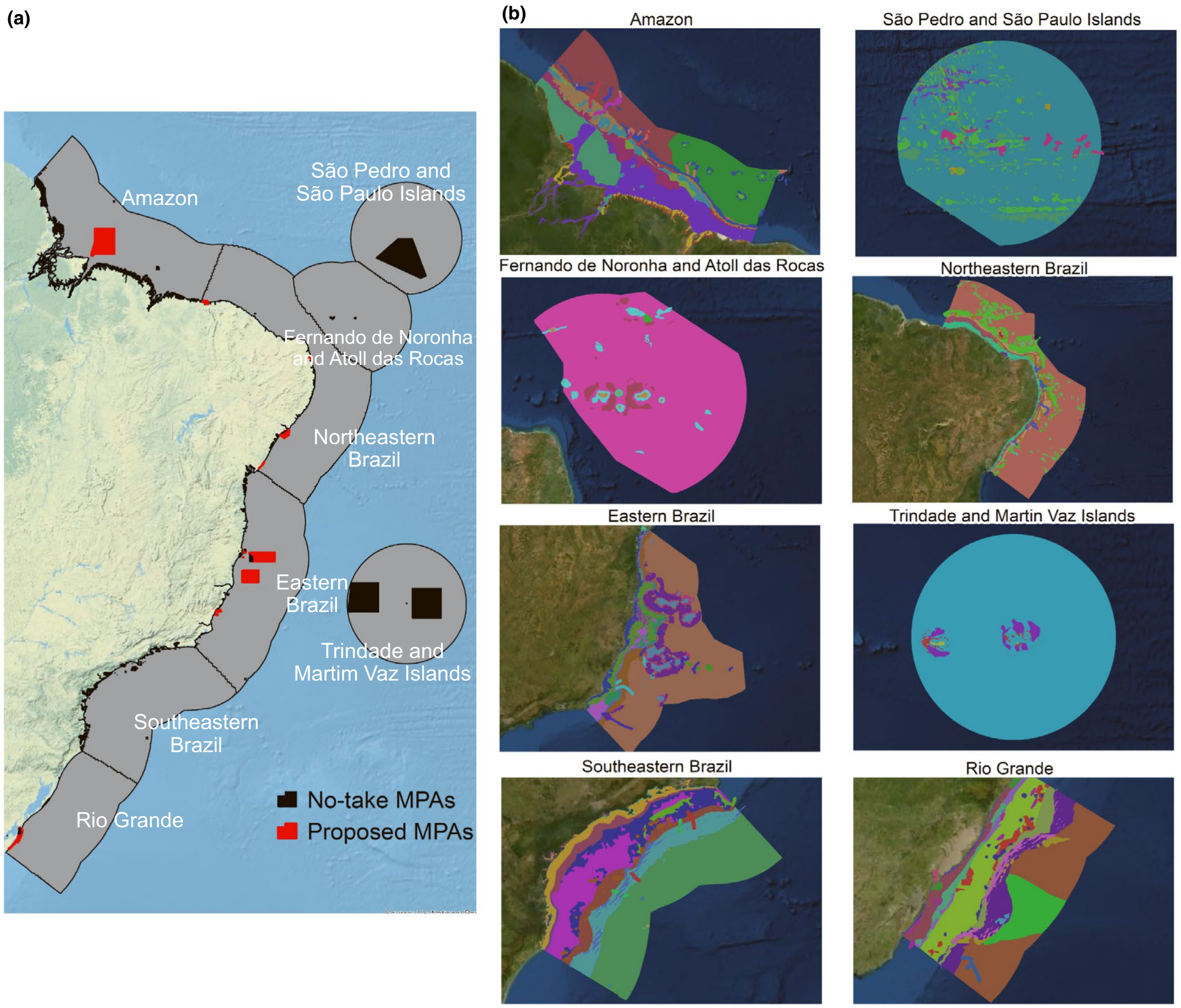


FIGURE 1 The study region, MPAs and the marine benthic habitats in Brazil. Panel (a) indicates the location of existing no-take and proposed MPAs within Brazil's EEZ (indicated by the shading area in grey). All habitats were first delineated by the ecoregions (sensu Spalding et al., 2007) occurring in the study region (a). Habitats (b) were further classified using information on the depth, substrate type (i.e. hard, soft or mixed substrate types related to the type of fauna present, geomorphological structures or presence of specific habitat-forming species), and other within-habitat specificities. Full description of each habitat, database codes and references used in the compilation of all marine habitats are in Table S1. Each habitat (*N* = 150) is represented by a distinct colour (please refer to the Figures S1–S7) and indicates an area where benthic community composition is assumed to be dissimilar from the surroundings

maps for all species were obtained from shapefiles downloaded from the National Red List of Threatened Species spatial data repository (i.e. the national agency for biodiversity conservation, ICMBio), the literature and the Aquamaps data set (Kesner-Reyes et al., 2016). The spatial data were processed by constraining them to the geographic distribution within the appropriate depth ranges for each species according to the text information in each species assessment. Following established practice, for wide-ranging (i.e. >20,000 km^2) mammals, seabirds and turtles, distribution maps corresponded to key areas for species conservation (breeding, foraging, calving or nursery areas) rather than encompassing large portions of habitats discontinuities (e.g. whale migration routes). However, spatial distribution data of breeding, foraging or nursery grounds for fishes and invertebrates with large range extents were not available at the national scale; thus, the geographic distribution of the species within these groups was delimited using occurrence records and reported depth ranges. We checked for the quality of all distribution data for each species when more than one data provider was identified. We are confident this represents the best available database on the distribution of threatened marine species in Brazil.

2.2.3 | Connectivity modelling

To incorporate connectivity into conservation planning, we re-analysed published spatial data about demographically significant dispersal links for reef-associated species that captured a range in species dispersal potential in the Brazilian reefs (Magris et al., 2016). The larval dispersal model was parameterized with corresponding spawning time and planktonic larval duration (PLD) for four hypothetical species (i.e. brooder coral, broadcasting coral, large carnivorous fish and roving herbivorous fish) to calculate connectivity matrices describing the probability of larvae being transported between a natal reef and a neighbouring or non-adjacent reef. Further details of the parameterization of larval simulations using daily data on ocean currents (from 2008 to 2012) are available in Magris et al. (2016).

For each modelled species, we calculated the average plume length, the furthest distance from the source reef at which connections occur, averaged over all connections, as described by Thomas et al. (2014). We then used these typical reef's influence extensions of all species and geographic distances between each individual reef-based habitat (edge-to-edge distance between reef polygons) to generate potential ecological networks. For this scale of analysis, we considered reef-based habitats not only the shallow-water coral reefs (<25 m), but also shallow-water rocky reefs (<25 m) and mesophotic reefs at relatively shallow depths (<60 m). This approach was adopted as we still lack biophysical modelling of connectivity within the entire study region. This assumption is reinforced by the high similarity between community composition in shallow-water coral reefs and rocky reefs (i.e. about 82%; Aued et al., 2018), and between shallow-water coral reefs and the upper mesophotic coral reefs (depths < 60 m) in the western Atlantic (i.e. about 87%; Rocha et al., 2018).

Connectivity between individual reef-based habitats was defined according to the geographical distance and the dispersal thresholds defined for each modelled species. For instance, the typical furthest distance at which demographic connectivity can occur for a brooder coral was 70 km, so we assigned a link to every pair of reef-based habitats located within 0–70 km from one another. This process was repeated for all modelled species. A broadcast spawning coral had an average plume length of 380 km, while a roving herbivorous fish can settle furthest from their natal reef, with an average plume length of 429 km. Large carnivorous fish might function as a long-distance disperser, with minimal larval exchange at distances of over 541 km from the source reef. The whole set of linkages was carefully revised and hand-edited to ensure that this process did not compromise geographic integrity (e.g. deviation of linkages around complex shorelines).

We used the probability of connectivity index (PC), an index that quantifies the amount of habitat available to species based on a reef-based attribute (i.e. reef area), and on a dispersal-related connectivity measure within the network of reefs (i.e. the set of linkages between all connected reef-based habitats) (Saura & Pascual-Hortal, 2007):

$$\mathrm{PC} = \frac{\Sigma_{i=1}^{n} \Sigma_{j=1}^{n} a_i \times a_j \times p_{ij}^{*}}{A_L^2} \tag{1}$$

where n is the total number of planning units containing reef-based habitats (i.e. patches), a_ie $a_j$are the area of the habitat patches i and j, $p_{ij}^{*}$is the maximum product probability of all possible paths between patches i and j, and $A_L$is the total seascape area attribute (maximum attribute).

The PC values range from 0 to 1, and higher values indicate greater connectivity (Saura & Pascual-Hortal, 2007). Thus, we calculated the individual reef-based habitat contribution for connectivity (ΔPC) using an individual habitat removal experiment (Saura & Pascual-Hortal, 2007). The sum of the individual reef-based habitat contributions was summarized at the planning unit level by considering all individual reef-based polygons within it. Planning units that contain no reef-based habitat have a value of 0. These connectivity analyses were performed using the command line Conefor Sensinode (Saura & Torné, 2009) in R software (R Core Team, 2015). See Appendix S2 for further detail on how we calculated the PC index.

2.2.4 | Spatial data on threats

Spatially explicit data on the distribution of threats to marine biodiversity were gathered and analysed separately for 24 of the most harmful pressures humans exert on the marine environment, including (a) industrial fishing (bottom gillnet, bottom trawl, live bait, pelagic longlines, demersal longlines, pelagic driftnets, bottom handline, pelagic handline, purse seines, traps); (b) climate change (global warming, ultraviolet radiation, acidification); (c) coastal development; (d) port-derived pollution; (e) shipping lanes; (f) land-based pollution (sediments, organic pollution, pesticides, fertilizers); (g) ocean mining; (h) oil/gas extraction activities (exploration, production); and (i) invasive species. We obtained detailed information and developed layers of most threats from monitoring data, scientific papers and reports; only data on ultraviolet, ocean acidification, shipping movement and invasive species were developed at global scale (Halpern et al., 2008). This makes the present data set the most up-to-date and comprehensive threat map available for Brazilian waters. Table 1 synthesizes how each threat layer was developed, the sources of raw data and analyses involved to create each layer (details of the different types of threat layers are provided in the Appendix S3).

2.3 | Conservation prioritization

Our framework employed systematic conservation planning using integer linear programming to identify areas of conservation priority within the Brazilian EEZ for the conservation features described above (i.e. habitats, threatened species and connectivity indexes). We aimed to ensure that the marine areas recommended for protection covered distributional ranges for a variety of habitats and threatened taxa for which such data were available and that the seascape connectivity was maintained. This was implemented with

TABLE 1 Description of each threat layer used for cumulative impact assessment on threatened species and marine habitats in Brazil

Major threat	Source of data set	Data model and attributes	Scale/temporal coverage of data	References
Industrial fishing	Vessel monitoring systems (VMS) data from the "*Programa nacional de rastreamento das embarcações pesqueiras por satélite*"—PREPS	Density map of the spatial location of 4,205,607 signals emitted from 905 industrial active vessels in fishing operations for 10 fisheries and associated gear types (bottom gillnet, bottom trawl, live bait, pelagic longlines, demersal longlines, pelagic driftnets, bottom handline, pelagic handline, purse seines, traps)	1 km, 2015–2017	This study
Climate change	NOAA Pathfinder Project (http://pathfinder.nodc.noaa.gov) for SST data	Global warming: rate of warming using non-linear mixed effect models based on monthly climatology of sea surface temperature	4 km, 1985–2009	This study
	https://knb.ecoinformatics.org/#view/; https://doi.org/10.5063/F19Z92TW	UV radiation: the number of times between 2000 and 2004 that the monthly average exceeded the climatological mean + 1 standard deviation within the entire data set	1-degree, 1996–2004	Halpern et al. (2008)
	https://knb.ecoinformatics.org/#view/; https://doi.org/10.5063/F19Z92TW	Ocean acidification: difference in global distribution of the aragonite saturation state of the ocean in pre-industrial (~1,870) and modern times (2000–2009)	1-degree, 2000–2009	Halpern et al. (2008)
Coastal development	DMSP/NOAA/NGDC night-time satellite imagery	Relative distance from each raster cell (centroid) to a source of night-time light (based on the metric LPI (light proximity index)	1 km, 2013	This study
Port-derived pollution	Brazilian Agency for the Environment (IBAMA)	Vector data set of spatial locations of ports ($N = 64$) as well as areas beyond their physical presence but within a zone of influence of that threat (i.e. dredging and disposal sites)	1 km, 2017	This study
Shipping lanes	https://knb.ecoinformatics.org/#view/; https://doi.org/10.5063/F19Z92TW	Network density of cargo ship movements based on data accumulating ship positions over time	1 km, 2004–2005	Halpern et al. (2008)
Land-based pollution	Coastal catchment data from HydroBasins (http://www.hydrosheds.org/page/hydrobasins), bathymetry (Supporting information), and oceanic current velocity (https://www.hycom.org/data/glbu0pt08/expt-19pt1)	Sediment: the maximum plume extent based on the effects of surface currents, depth and particle settling rates	1 km, 2016	This study
	Pollution data from the Brazilian National Water Agency (http://www.snirh.gov.br/portal/snirh/snirh-1/atlas-esgotos)	Organic pollution: the amount of diluted non-treated sewer discharge per council and its zone of influence (a 5-km buffer around each council)	1 km, 2017	This study

(Continues)

TABLE 1 (Continued)

Major threat	Source of data set	Data model and attributes	Scale/temporal coverage of data	References
	Coastal catchment boundaries (as above) and pesticides consumption per municipality from Pignati et al., 2017	Pesticides: Vector data set containing the consumption of pesticides used at the municipality level within catchments; Exponential decay model based on the distance from river mouth locations	1 km, 2017	This study
	Coastal catchment boundaries (as above) and fertilizer usage from the Brazilian Institute of Geography and Statistics (http://www.sidra.ibge.gov.br/bda/pesquisas/pam/default.asp)	Fertilizers: Vector data set containing the cropland area for rice, coffee, soy, sugarcane, and corn within catchments; information on the consumption of fertilizers for all these types of crops; Exponential decay model based on the distance from river mouth locations	1 km, 2014	This study
Ocean mining	National Department of Mineral Production (http://www.dnpm.gov.br/assuntos/ao-minerador/sigmine	Vector data set of spatial locations of areas converted for mining operations (N = 564)	1 km, 2017	This study
Oil/gas extraction activities	Brazil's National Agency of Petroleum, Natural Gas and Biofuels (http://app.anp.gov.br/webmaps/)	Vector data set of spatial locations of areas converted for oil/gas exploration and production (N = 2,912)	1 km, 2017	This study
Invasive species	https://knb.ecoinformatics.org/#view/; https://doi.org/10.5063/F19Z92TW	Modelled as a function of ballast water release in ports	1 km, 2008	Halpern et al. (2008)

the package *prioritizr* (version 4.0.2; Hanson et al., 2017 in R R Core Team, 2015).

Targets for each habitat were calculated based on information about rarity (i.e. geographic range size) and estimate of the potential cumulative impacts from multiple threats (e.g. fishing, climate change, coastal development, land-based pollution). This procedure gave larger targets for habitats with smaller distributions, given that the unprotected portions of their distributions are disproportionally more affected by impacts than others (Pressey et al., 2003). Likewise, larger targets were associated with sensitive habitats co-occurring with multiple threats, given that this condition plays a prominent role in determining the risks of ecosystem collapse (Kraberg et al., 2011; Rocha et al., 2015).

Estimates of cumulative impact were calculated as described by Halpern et al. (2008) and consisted of three components: (a) spatial distribution of each marine habitat; (b) spatial distribution of each threat; and (c) sensitivity weights to convert each threat to its relative impact on each of the habitats. We summarized the occurrence of all threat layers within each planning unit containing varying extents of each habitat throughout their distribution. We incorporated the sensitivity of each habitat to each threat (i.e. "vulnerability") following the framework developed by Halpern et al. (2007), based on expert judgement. Where there was an overlap, we multiplied each threat layer with each habitat layer and then multiplied each combination by the corresponding weight variable representing a relative vulnerability of that habitat to each overlapping threat. This means that the presence of a threat and a habitat in the same planning unit is not considered an impact unless the given habitat is known to be sensitive to that threat (Tables S3 and S4). We link the habitats and threats developed here to the ones presented by Halpern et al., 2007 to assign the weight values (Table S5). We then summed up across all of the individual impacts within the planning units for every habitat to measure the relative cumulative impact of human activities; thus, the cumulative impact assessment was made habitat specific, resulting in 161 total cumulative impact scores. Following previous approaches (Brown et al., 2014; Magris et al., 2018; Micheli et al., 2013), this study assumed that the cumulative impacts of multiple threats interact additively rather than synergistically or antagonistically, and that only direct effects were considered.

We used the following formula to derive the targets:

$$T_i = 0.1 + (0.2 \times r_i) + (0.2 \times CI_i) \tag{2}$$

where T_i is the target for the habitat i, r_i is log − indexed rarity score for habitat i, and CI_i is the average normalized cumulative impact across the area occupied by habitat i.

We worked out the formula for calculating conservation targets to guarantee a minimum theoretical value of about 10% of each habitat extent but it could increase to a maximum of 50% if the habitat was very rare and heavily impacted. Targets across habitats ranged from 14% (i.e. coastal [<25 m], soft-bottom habitat in the Amazon ecoregion) to 36% (i.e. mesophotic reef on the insular shelf of the São Pedro and São Paulo Islands ecoregion). Final targets for each habitat are shown in Table S6.

To set adequacy targets for threatened species protection, we scaled their targets based on information about rarity (geographic range size), cumulative impacts from multiple threats and life-history characteristics. We followed the method of de Novaes e Silva et al. (2014), ensuring that all species had a base target value of 10%. This value could then increase progressively for small-ranging species, with characteristics that reflect increased vulnerability to population declines, and facing a very high risk of extinction (i.e. species included within the Critically Endangered category in the National Red List). Similarly to the estimation of impact for the habitats, we calculated the extent of spatial overlap between each species distribution and all threat layers that have been identified in the Brazilian National Red List as threats to that species. Our assessment accounted for the overlap between threats and considered variable responses of species to threats based on their extinction risks (i.e. National Red List status categories). For the life-history component, we obtained information on biological traits relevant to the habitat requirements and ecological vulnerability of each species, including body size (i.e. maximum body length), maximum depth, habitat specialization and trophic category.

To calculate our targets, we used the following formula (adapted from de Novaes e Silva et al., 2014):

$$T_i = 0.065 + 0.1NR_i + 0.05VL_i + 0.05LH_i \tag{3}$$

where T_i is the target for species i, NR is natural rarity, VL is vulnerability, and LH is life-history characteristics for species i. The weights were tuned by de Novaes and Silva et al. (2014) to guarantee a minimum target of 10% and find a balance between widespread and rare species. Targets across species ranged from 12% (e.g. *Makaira nigricans*) to 34% (e.g. *Willeya loya*). Further details on how we formulated the target for each species and final targets used in the analysis are shown in Appendix S4 and Table S7.

For connectivity indexes, we followed the method proposed by Magris et al. (2016) to calculate representation objectives. We identified subsets of planning units with the highest contributions to connectivity (i.e. belonging to the top tercile for each modelled species) and calculated the percentage of top-ranked planning units that contributed to the total values of each modelled species to derive the minimum amounts to be targeted in the prioritization.

We implemented the minimum set objective function, which seeks to minimize the cost of the solution (measured in this case by the area of planning units as the cost) while ensuring that all targets are achieved. We used a boundary length modifier (BLM = 1) to spatially clump the priority areas. We followed best practices in carrying out calibration analyses to determine appropriate boundary length modifier. Existing no-take MPAs were set to be included in the conservation solution (i.e. locking them in). Our application did not include MPAs offering partial protection because we lack information about their ecological effectiveness. Thus, our solution found the optimal area outside of MPAs that maximized the representation of threatened species ranges, marine habitats and connectivity across Brazilian waters. While the conservation planning application did not ensure that heavily impacted areas were necessarily selected, our solution provided greater protection to those components of biodiversity facing high levels of threats in compensation for potential loss caused by cumulative impacts.

2.4 | Evaluation of the conservation prioritization

For our first assessment, we computed the irreplaceability score for each planning unit selected by prioritizr' best solution (i.e. the number of times each planning unit was selected in 1,000 runs of the algorithm). The irreplaceability scores provide a useful indication of the conservation importance of each planning unit. Similarly, we thus estimated the cumulative impacts on each of those planning units selected by prioritizr' best solution. We used the same calculation of cumulative impact for the formulation of targets and summed individual impact scores across all conservation features within selected planning units. We considered "priorities" those planning units belonging to either the upper tercile for irreplaceability or the upper tercile for cumulative impacts. Planning units that belonged to the upper terciles for both irreplaceability and cumulative impact were considered "top priorities." In doing so, our approach aimed to provide an indication of how urgently protection is necessary, pinpointing areas where MPAs would serve as management measures for minimizing the impacts of threats. As such, MPAs could prevent further biodiversity loss and increase potential resistance to, or recovery from, disturbance events.

To investigate spatial differences in the distribution of priority areas between scenarios designed with and without a typical conservation planning exercise, we performed two additional analyses. First, we compared the irreplaceability outputs and cumulative impact scores for the planning units within our spatial priorities with those that had been selected randomly. Second, we compared the irreplaceability outputs and scores to cumulative impact for planning units intersecting the proposed MPAs with those that had been selected randomly. For these two comparisons, we used the method described by Kuempel et al. (2019) and generated two random MPA systems using the function sample in R with replacement: the first using the same number of planning units of the priorities and the second using the same size of proposed MPAs to generate the random solutions. The random MPAs are hypothetical MPA systems and did not mean to reflect solutions for on-ground conservation. They were merely used to compare the performance between systematic and non-systematic approaches, given their difference in size (area). We used Wilcoxon's signed-rank tests to determine whether the irreplaceability outputs and cumulative impact scores of the random solutions were significantly different from the ones of the priorities and proposed MPAs, respectively.

3 | RESULTS

3.1 | Evaluation of spatial data

The maps depicting the spatial extent of marine benthic habitats, pelagic habitats, distribution of threatened species, and the ecological connectivity patterns are presented in Figures 1 and 2. The benthic habitat maps reflected the availability of data and variation in data resolution across the eight ecoregions (Figure 1a). For example, coastal habitats in ecoregions where a vast amount of survey effort was undertaken (e.g. Eastern Brazil) outnumbered those belonging to deep-water environments (e.g. Trindade and Martim Vaz), where data are often sparse, and conditions are more homogeneous (Figure 1b). Pelagic habitats were more spatially restricted along the coast of subtropical Brazil and spatially extended towards the tropics (Figure 2a). The distribution of threatened species was uneven along the Brazilian coast, with the highest occurrence of these species along the coastal waters of São Paulo state (between latitude 21° and 25°S), south-eastern Brazil ($N = 78$; Figure 2b). The four modelled taxa and the multi-species composite networks showing the ecologically significant linkages among reef-based habitats revealed two isolated clusters (Figure 2c): one formed by dispersal pathways between reefs bordering the shore of oceanic islands and shallow-water reefs along the Eastern Brazilian Margin, and another formed by dispersal pathways between reefs along the Equatorial Margin. The spatial pattern of connectivity highlights key seascapes where spawning habitats occur in close proximity and facilitate ecological connectivity among reefs.

The cumulative human impacts on threatened species and marine habitats were widespread and indicated that nearly the entire Brazilian EEZ has been facing some level of threat (Figure 3). There was a strong spatial variation in the intensity of cumulative impacts, with alarming peaks on the continental shelf of northern (i.e. Amazon ecoregion) and southern Brazil (i.e. South-eastern and Rio Grande ecoregions) (Figure 3). Hot spots of cumulative impacts were also found in offshore waters, with strong values around São Pedro and São Paulo Islands and on some deep-water environments of Fernando de Noronha and Atol das Rocas ecoregion. As expected, industrial fisheries, global warming and land-based activities were severe threats across many marine habitats and threatened species (Table 2). Concerningly, we found that 82.8% of the study region was threatened by pelagic longlines, the most widespread fishery, followed by pelagic driftnet (61.6%) and bottom trawl (60.1%). Global warming impacts extended across over 95% of the study region, while the proportion of the study region impacted by land-based activities was much smaller, ranging from 7.6% (organic pollution) to 22.6% (fertilizers and pesticides). See Appendix S5 for the distribution of impact for each threat layer.

3.2 | Prioritization analysis

Our prioritization analysis showed that planning units with higher selection were unevenly distributed among the ecoregions (Figure 4) and highlighted some areas currently underrepresented within the existing no-take MPAs. Large tracts of the ocean were often selected in the Amazon, South-eastern Brazil and Rio Grande ecoregions. In other ecoregions (i.e. Fernando de Noronha and Atoll das Rocas and São Pedro and São Paulo Islands), more frequently selected planning units were mostly non-contiguous.

The "best solution" from the prioritization corresponded to about 18% of the Brazilian EEZ (Figure 5a; including existing no-take MPAs). When combining conservation planning and cumulative impact assessment, our analysis prioritized 286,266 km^2 for

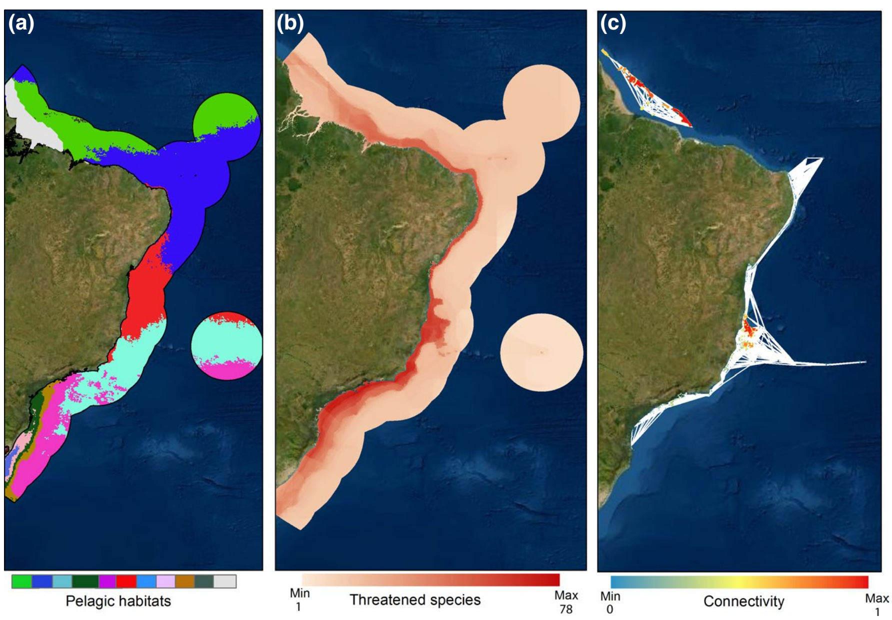


FIGURE 2 Distribution of pelagic habitats (a), threatened species (b) and potential dispersal connections between reef-based habitats estimated with data from Magris et al., 2016 (c). Pelagic habitats were delineated based on environmental data as surrogates for species distribution modelling. The gradient of colours in (b) represents the number of threatened species in a planning unit (1–78). The gradient of colours in (c) represents the importance of reef-based habitats in establishing connections across the seascape (0–1)

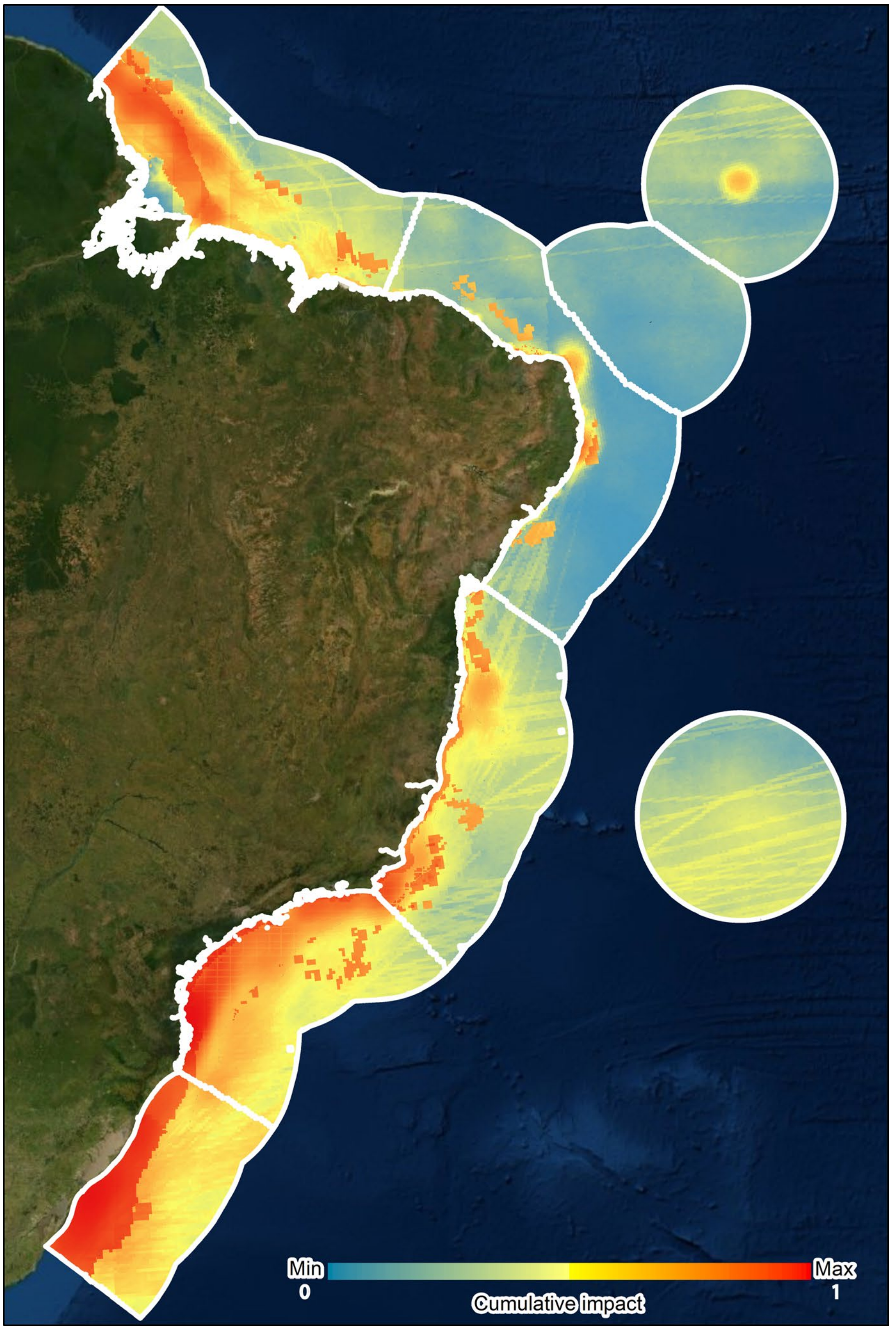


FIGURE 3 Spatial distribution of the cumulative impacts on marine habitats ($N = 161$) and threatened species ($N = 143$) across the Exclusive Economic Zone (EEZ) of Brazil (the boundaries of marine ecoregions are shown by white lines). The gradient of colours indicates the cumulative scores considering the 24 threat layers, including industrial fisheries, global warming, land-based activities and coastal development

conservation ("priority areas"; Figure 5b) which correspond to 7.9% of Brazilian EEZ. The priorities (Figure 5b) included both relatively intact and heavily impacted areas and were concentrated largely in several clumps including (a) outer shelf areas in the Amazon ecoregion, which contains a complex and diverse range of habitat types such as mesophotic reefs and rhodolith beds, and represents an important dispersal pathway along the Equatorial Margin; (b) offshore areas in North-eastern Brazil ecoregion, which contains an array of seamounts of the North Brazilian Chain; (c) coastal and offshores areas in the Eastern ecoregion, which was important for several threatened species, and for connectivity; (d) a combination of inner and mid-shelf areas in the Southern Brazil and Rio Grande ecoregions, which had several marine habitats with high cumulative impact scores. The top-priority areas (Figure 5c; ~2.30% of EEZ) were comprised of nearshore areas of Amazon, North-eastern, Eastern and South-eastern Brazil ecoregions, as well as mid-shelf areas of Eastern and South-eastern Brazil ecoregions with a patchy distribution. When identifying the priorities, more planning units fell within the top third of the cumulative impact score than within the top third of irreplaceability value (Figure 5d).

The priority areas identified here had irreplaceability values significantly higher ($p < .001$) from those that occur randomly (Figure 6a). However, there was no significant difference in the level of cumulative impact scores between the priority areas and the

TABLE 2 Summaries of the threat impacts on marine habitats and threatened species

Major threat	Subclass threats	Number of marine habitats impacted	Number of threatened species impacted	Proportion of Brazil EEZ impacted
Industrial fishing	Bottom gillnet	78	32	49.3
	Bottom trawl	129	84	60.1
	Live bait	109	15	8.8
	Pelagic longline	111	58	82.8
	Demersal longline	126	58	26.4
	Pelagic driftnet	111	8	61.6
	Bottom handline	123	24	58.8
	Pelagic handline	109	24	56.1
	Purse seine	109	8	33.4
	Traps	79	8	46.2
Climate change	Global warming	147	54	95.8
	UV radiation	58	4	73.3
	Ocean acidification	75	8	94.2
Coastal development	—	69	14	10.5
Port-derived pollution	—	58	14	4.8
Shipping lanes	—	94	24	58.2
Land-based pollution	Sediment	97	4	19.0
	Organic pollution	99	43	7.6
	Pesticides	101	43	22.6
	Fertilizers	101	43	22.6
Ocean mining	—	63	10	2.1
Oil/gas extraction activities	Exploration	48	14	9.3
	Production	14	14	5.6
Invasive species	—	43	3	3.4

random selection of MPAs of similar total size (Figure 6b). In addition, proposed MPAs did not perform better than random MPAs of a corresponding size for the comparison between irreplaceability values (Figure 6a). Finally, the proposed MPAs contained areas with high impact scores; there was no significant difference in the cumulative impact scores between proposed MPAs and the random selection of MPAs of similar size (Figure 6b).

4 | DISCUSSION

This study represents a case in which a cumulative impact assessment is accommodated within a conservation planning approach to undertake a national-scale prioritization and identify areas that help safeguard marine biodiversity in Brazilian waters over the next decades. Combining cumulative impact assessment with conservation planning is important because many marine planning efforts seek to deliver effective conservation outcomes, including the mitigation of threats to biodiversity (Álvarez-Romero et al., 2018; Kukkala & Moilanen, 2013), and hence require detailed spatial data sets on threatening processes. In Brazilian waters, existing MPAs are limited in their contribution to biodiversity conservation because they achieve some but not all conservation targets. In contrast, our approach, which was developed based on specific targets and quantitative ecological and threat data, strategically provides a conservation plan that entails countering biodiversity decline and contributes directly to fulfil global conservation commitments. Our approach is also repeatable and adaptable to other national frameworks, providing stakeholders and policymakers with comprehensive information about the conservation value across a study region.

While our study provides the current best estimate of the spatial distribution of human impacts on the marine environment, we find that all threatened species and marine habitats have been historically impacted by at least one relevant threat across some portion of their distribution. In particular, the ecological functions performed by nearshore habitats are deteriorating rapidly due to the cumulative impacts of multiple human disturbances (e.g. Copertino et al., 2016; Cruz et al., 2018; Duarte et al., 2020; Giglio et al., 2015; Gorman et al., 2020; Magris & Giarrizzo, 2020; Magris et al., 2019). Our study therefore demonstrates the utility of the cumulative impact maps in developing integrated conservation planning and offers critical insights for the strategic implementation of conservation actions in Brazil. For instance, such maps, although

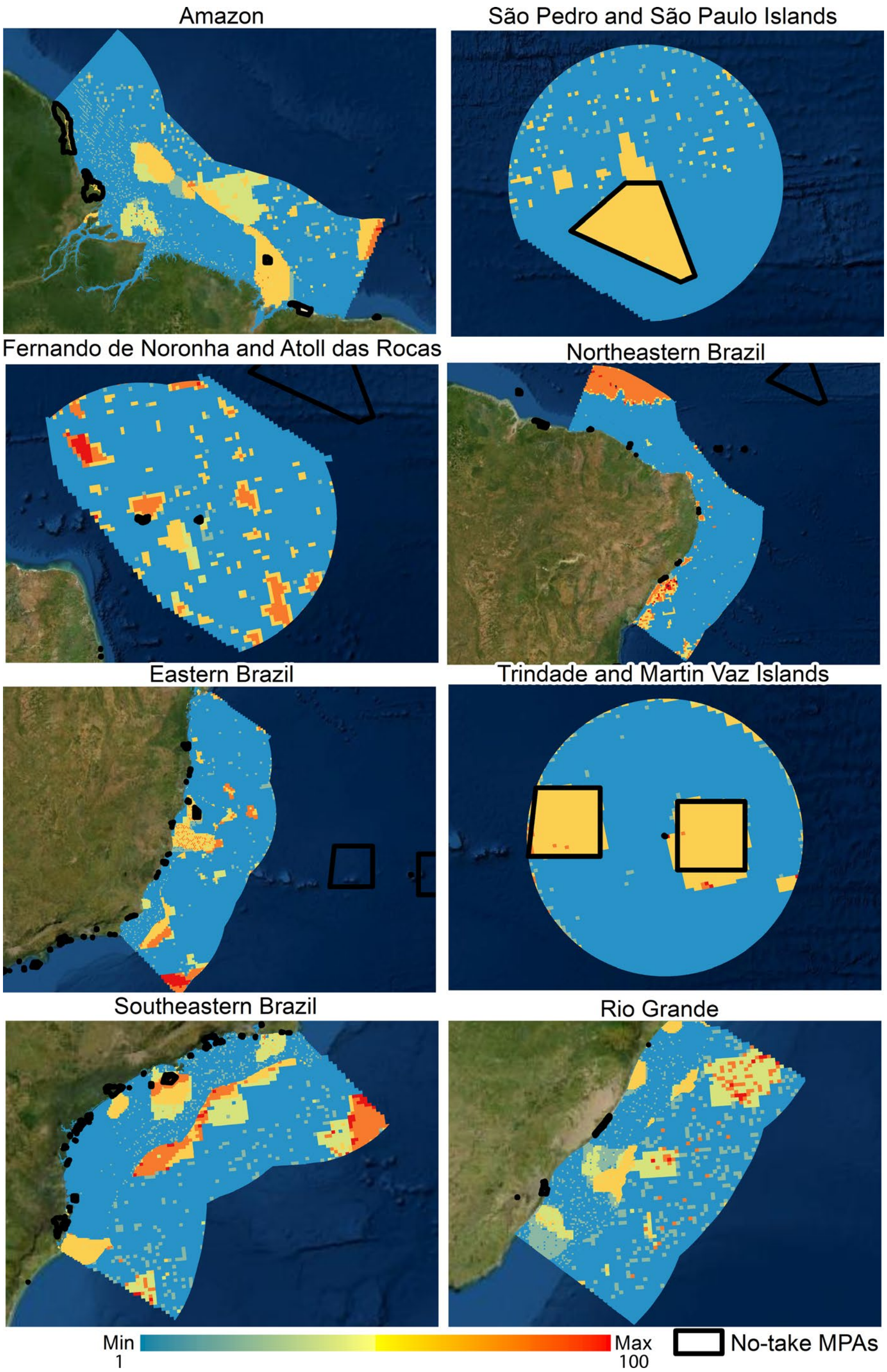
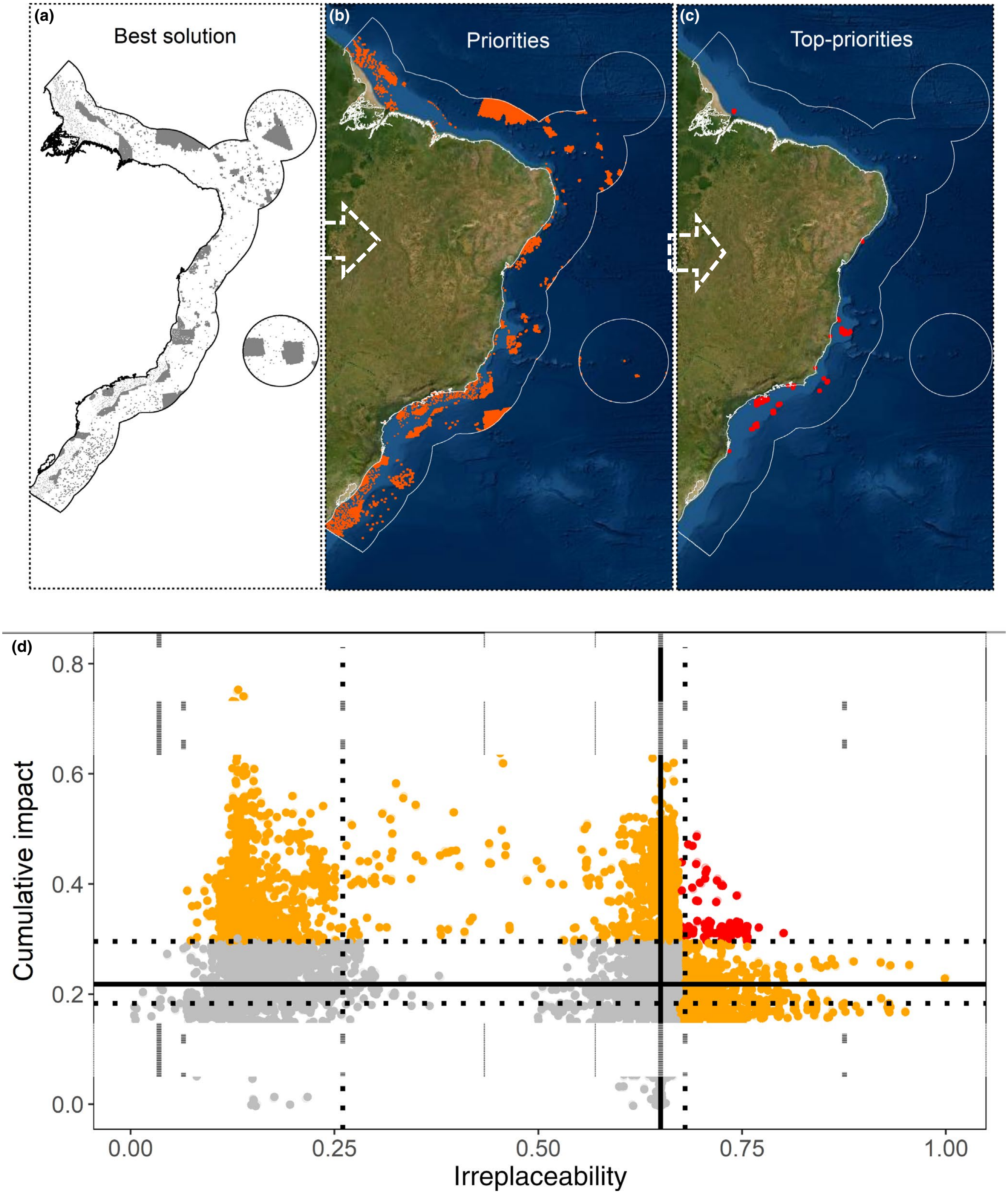


FIGURE 4 Spatial distribution of the prioritized areas for planning an extended network of MPAs in the Brazilian Exclusive Economic Zone based on the layers of Figures 1, 2 and 3. The gradient of colours represents the irreplaceability values when existing no-take MPAs are "locked-in." Panels correspond to the locations of the eight ecoregions showing in Figure 1a

presenting only a snapshot in time, could assist in prioritizing both areas facing imminently high levels of threat, and areas that are currently less disturbed but could become threatened in the future. Improved mapping and assessment of other human activities for which data did not exist at an adequate resolution (e.g. small-scale fisheries and tourism activities) are a critical next step for further refinement of the approach.

One limitation of this analysis that must be considered for further research efforts is the need to derive specific vulnerability scores for habitats in Brazil. Indeed, the ecological responses

FIGURE 5 Identification of priority areas for implementing new MPAs using conservation planning and cumulative impact assessment. Prioritizr best solution (a) was used to extract information on irreplaceability values and cumulative impact scores (planning units coloured in grey). In (b), planning units coloured in orange represent priorities when combining these two methodologies. In (c), planning units coloured in red represent top priorities. In (d), comparison of the levels of importance on each planning unit showing in (a) between irreplaceability values (x-axis) and cumulative scores (y-axis). All dots were coloured according to the description above (i.e. grey, orange and red colours referring to planning units selected as best solution, priorities and top priorities, respectively). Dotted lines represent the lower and upper terciles, and solid lines represent median values


(a)
Best solution
(b)
Priorities
(c)
Top-priorities
(d)
0.8
0.6
0.4
0.2
0.0
Cumulative impact
0.00
0.25
0.50
0.75
1.00
Irreplaceability

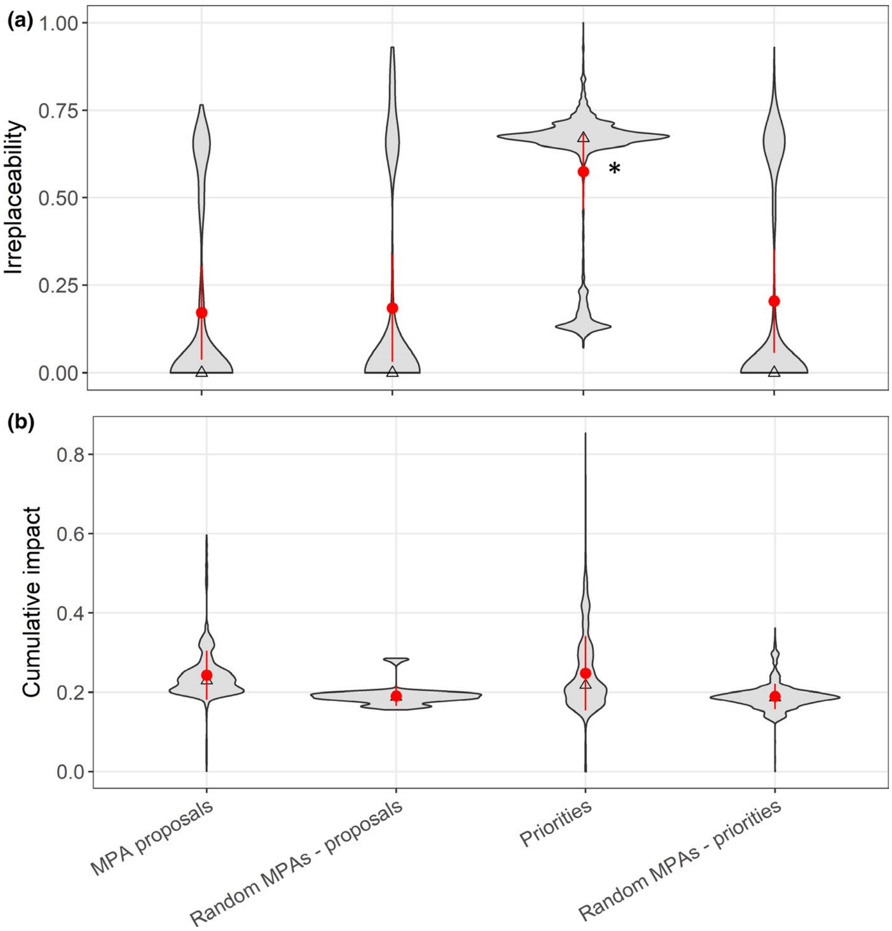


FIGURE 6 Violin graphs comparing irreplaceability (a) and cumulative impact scores (b) between the priorities identified in this study (as in Figure 5b) and the proposed MPAs against a random selection of MPAs. Circle dots in red represent the mean level while the triangles represent the median. *$p < .001$ for comparison against MPAs selected at random

following negative pressures vary markedly across taxonomic groups, human impact types, ecological metrics and geographic regions (Pimm et al., 2014), which might make the effect of multiple threats context dependent. However, a critical constraint in addressing this challenge is the lack of significant collection of field data over large temporal scales to assess the impact of human disturbances on a variety of habitats in Brazil. Although the reliability of using the global scores for regional applicability is unclear, more promising approaches use expert judgement to derive specific vulnerabilities (e.g. Hammar et al., 2020; Korpinen et al., 2013), when finer-scale and higher-quality data are not available.

Several priority-setting initiatives developed at the national and global scales have suggested conservation priorities within the South-western Atlantic Ocean. Previous initiatives have varied greatly in terms of objectives, taxonomic groups, scale of analysis, data and methods used for identifying priorities (e.g. Davidson & Dulvy, 2017; Jenkins & Van Houtan, 2016; Jones et al., 2020; Vilar et al., 2020). We developed an approach that includes, for the first time, threats, biodiversity representation, connectivity and a wide range of threatened species representing data assembled from multiple sources as input features. Still lacking are initiatives that consider the variability in the permitted and excluded uses within differing MPA protection levels and the costs of different conservation actions. Application of these methods with socio-economic considerations (e.g. Vilar et al., 2020) will create trade-offs between achieving different targets and minimizing conflicts and costs.

We have included in our analysis high-resolution spatial data representing historical temperature variation (i.e. rates of change in sea surface temperature) and their direct impacts on biodiversity. A challenge for all conservation prioritization analyses is the availability of high-quality and high-resolution data that accurately represent future ocean conditions (e.g. Wilson et al., 2020). Future work should encompass shifts in species geographic distributions, given the high sensitivity of marine species to environmental change (Pinsky et al., 2020), and identify locations that represent habitat refugia (e.g. Magris et al., 2015). Moreover, future biophysical model developments should accommodate projections of physical variables and ocean currents to forecast changes in patterns of species dispersal. Because the configuration of marine reserve networks might need to be modified to maintain connectivity under future climatic conditions (Gerber et al., 2014), we reinforce the need of the MPA planning process be adaptive in response to advances in knowledge and methods.

Our estimate that 7.9% of the Brazilian EEZ ("priorities") requires effective conservation should be viewed as another step in the sequential implementation of management actions and is a conservative estimate. This can be explained by the baseline target used in this study (i.e. 10% framed in terms of species' range and habitat distribution), which may simply not be enough. The minimum 10%

target has been frequently questioned as it might fail to maintain the long-term persistence of biodiversity (Baillie & Zhang, 2018; Gaines et al., 2010). Furthermore, we considered only threatened species, which represent only a fraction of all marine species. Including more comprehensive species-level data could influence prioritization results and improve information on patterns of biodiversity (e.g. Vilar et al., 2017) in the study region. Moreover, although we used historical data on threats, we did not consider spatial-temporal patterns of threats in the future, which might require an increase in the total area under conservation.

The results of our analysis showed that the currently proposed MPAs fall short on meeting prioritization targets, whereas they would do reasonably well for protecting areas that are facing high levels of cumulative impact (see Figure 6b). While the use of decision support tools can be very helpful in identifying the locations that generally contain several conservation features, we found that local stakeholders can also suggest areas that would provide important conservation benefits because of the severity of threats faced by these areas. Given the importance of these developing marine use plans in Brazil, our approach was able to visualize the conservation value of the collection of individual proposed MPAs which would have not been apparent through a one-by-one assessment. An important ongoing avenue of research will be to continue to compare and integrate systematic conservation planning with more refined information on region-specific ecological data of these areas.

In the near-term (2020–2030), Brazil has committed to expanding its no-take MPA system by over two-hundred-fifty thousand km^2 (to 10%). To direct this, our analyses highlight several areas that are currently unprotected, such as the shelf-edge reefs in the Amazon ecoregion, photic and shallow seamounts of the North Brazilian Chain, southern portions of the Abrolhos Bank, and deep-sea coral banks, rhodolith beds, alongside with soft-benthic communities in southern Brazil. These areas collectively are shown to be critical for achieving conservation targets in Brazilian waters as they exhibit high irreplaceability and face high levels of threat. In addition, they are shown to be important in global conservation prioritization analyses (Davidson & Dulvy, 2017; Jenkins & Van Houtan, 2016; Jones et al., 2020). So, alongside highlighting the conservation value of these areas, our study provides maps at a suitable scale for use by policymakers to delineate future MPA boundaries.

The protection of the areas identified by our analysis would provide important conservation benefits. We suggest the use of the analytical method presented here to assist decision-makers by providing the best available spatial information about natural systems and their human uses in the marine environment. An important future priority is to discuss the outcomes of our study with stakeholders and representatives of the relevant agencies to develop consensus on the management, revise the design and reach the final recommendation for the delimitation of conservation priorities. Stakeholder involvement and engagement are essential for future compliance with the protection of these environments. Lastly, we recognize that MPAs alone might not be sufficient to mitigate some threats to biodiversity, such as global climate change (e.g. Bates et al., 2019) and land-based pollution (e.g. Magris et al., 2019). Further investigations should consider the effects of other management tools (e.g. catchment-based restrictions) to achieve better conservation outcomes.

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PEER REVIEW

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DATA AVAILABILITY STATEMENT

Spatial data available via the Dryad Digital Repository https://doi.org/10.5061/dryad.xsj3tx9d1.

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BIOSKETCH

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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