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Strengthening the synergies among global biodiversity targets to reconcile conservation and socio-economic demands

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Abstract

1. Most of the world's nations adopted the 20 Aichi global biodiversity targets to be met by 2020, including the protection of at least 10% of their coastal and marine areas (Target 11) and the avoidance of extinction of threatened species (Target 12). However, reconciling these biodiversity targets with socio-economic demands remains a great dilemma for implementing conservation policies.
2. In this paper, Aichi Targets 11 and 12 were simultaneously addressed using Brazil's exclusive economic zone as an example. Priority areas for expanding the current system of marine protected areas within the country's eight marine ecoregions were identified with data on threatened vertebrates under different scenarios. Additionally, the potential effects of major socio-economic activities (small and large-scale fishing, seabed mining, and oil and gas exploration) on the representation of conservation features in proposed marine protected areas were explored.
3. Areas selected for expanding marine protected areas solely based on biodiversity data were different (spatial overlap from 62% to 93%) from areas prioritized when socio-economic features were incorporated into the analysis. The addition of socio-economic data in the prioritization process substantially decreased opportunity costs and potential conservation conflicts, at the cost of reducing significantly (up to 31%) the coverage of conservation features. Large and small-scale fisheries act in most of the exclusive economic zone and are the major constraints for protecting high-priority areas.
4. Nevertheless, there is some spatial mismatch between areas of special relevance for conservation and socio-economic activities, suggesting an opportunity for reconciling the achievement of biodiversity targets and development goals within the intricate Brazilian seascape by 2020 and beyond.

KEYWORDS
Aichi targets, endangered species, fishing, marine protected area, mining, ocean, coastal, spatial prioritization

1 | INTRODUCTION

Under the auspices of the 2011–2020 Strategic Plan of the Convention on Biological Diversity (CBD), most of the world's nations adopted in 2010 a set of 20 targets, known as Aichi biodiversity targets. Target 11 of this plan requires signatory countries to protect at least 10% of their coastal and marine areas by 2020, through 'effectively and equitably managed, ecologically representative and wellconnected systems of protected areas and other effective area-based conservation measures' (CBD, 2010). Such a target represents a significant opportunity to expand the global coverage of protected areas and then to reduce pervasive biodiversity decline (De Santo, 2013; Venter et al., 2014). Most nations are developing and reviewing their own national conservation strategies and establishing new marine protected areas (MPAs), in line with the Aichi targets (CBD, 2010; De Santo, 2013). In several cases, however, large MPAs have been created in remote places with relatively few threats and that typically provide little protection for the most threatened species and ecosystems (Devillers et al., 2015; Giglio et al., 2018). The crucial question, therefore, is where should MPAs be established to minimize biodiversity loss?

Strategically designed protected areas can reduce human impacts and help restore the abundance of marine species while contributing to achieving different international targets (Edgar et al., 2014; Speed, Cappo, & Meekan, 2018; White et al., 2017). Aichi Target 12, for example, states that the extinction of known threatened species should be prevented and their conservation status should be improved and sustained (CBD, 2010). However, existing MPAs provide inadequate or no cover for many threatened species (Butchart et al., 2015; Davidson & Dulvy, 2017). If actions taken to expand the global MPA system (Aichi Target 11) prioritize the coverage of threatened species, they would directly contribute to, concomitantly, reaching these two targets (Di Marco et al., 2016; Venter et al., 2014). Such actions could also help in achieving targets related to other governmental commitments, such as the United Nations Sustainable Development Goal 14, the Agreement on the Conservation of Albatrosses and Petrels, and the Ramsar Convention, among others (CBD, 2010).

Though scientists advocate for protection of 30% (or more) of the ocean by 2030 (e.g. Baillie & Zhang, 2018; Dinerstein et al., 2019; Roberts, 2019), implementing previous marine conservation goals has been a major challenge (Butchart et al., 2015). Failure to protect 10% of the marine area as of 2012, as formerly agreed under the CBD (Mittermeier et al., 2010), is an example of the implementation crisis in conservation efforts. The deadline to meet the Aichi targets is approaching, and inadequate consideration of stakeholders' needs and demands can undermine implementation actions (Fox et al., 2013; Knight et al., 2008; Knight, Cowling, Possingham, & Wilson, 2009). Given that marine conservation efforts happen in a scenario shared by multiple societal and governmental sectors (including commercial fishing, mining, hydrocarbon industry, tourism, maritime transport, etc.), both biological and socio-economic goals should be integrated from the onset of conservation efforts (Halpern et al., 2013; Jumin et al., 2017; Knight, Cowling, Boshoff, Wilson, & Pierce, 2011). This could avoid polarization among different and often competing stakeholders and facilitate the implementation of conservation actions (Ban & Klein, 2009; Mazor, Giakoumi, Kark, & Possingham, 2014; Naidoo et al., 2006).

The inclusion of spatially explicit information on socio-economic activities in the planning process can reduce opportunity costs (the foregone revenue from other forms of space use), sometimes with no negative effects on the coverage of conservation features (Almada & Bernardino, 2017; Ban & Klein, 2009; Klein, Steinback, Scholz, & Possingham, 2008; Naidoo et al., 2006; Vilar, Joyeux, Loyola, & Spach, 2015). Contrarily, considering costs to a single socio-economic activity or assuming a uniform cost through a given planning region can result in negative socio-economic impacts if important sites for conservation are occupied by resource users (Ban & Klein, 2009; Hamel, Pressey, Evans, & Andréfouët, 2018; Mazor et al., 2014; Naidoo et al., 2006). This is a particularly relevant problem for developing countries where coastal communities are heavily dependent on marine resources (Food and Agriculture Organization of the United Nations, 2014; Vasconcellos, Diegues, & Sales, 2007) and a large part of gross domestic product comes from commodities at sea (United Nations & Food and Agriculture Organization of the United Nations, 2017). Since current investments in conservation actions are usually inadequate, the development of cost-effective strategies for marine conservation is vital to help nations in efficiently achieving global biodiversity targets (Di Marco et al., 2016).

Spatial conservation prioritization provides an analytical and conceptual framework to identify areas of high biodiversity value and low cost for protection (Ferrier & Wintle, 2009). Although the use of this framework for the design of MPA networks has increased over the last decades, frameworks have often been based solely on biological and ecological data (Álvarez-Romero et al., 2018). Integrated planning for both biodiversity and development goals can be successful if undertaken through multicriteria analyses (Di Minin et al., 2013; Moilanen et al., 2011). Multicriteria analyses represent a powerful tool to enhance our understanding about the potential implications of multiple socio-economic activities on the effectiveness of MPA networks to conserving biodiversity (Halpern et al., 2013; Klein et al., 2008; Li et al., 2019). They could be particularly helpful to implement the targets, revealing synergies and trade-offs between numerous environmental and socio-economic objectives from alternative MPA configurations.

This paper addresses the current policy dilemma of how to balance the wide range of diverse economic activities and global biodiversity targets. To do so, we envisioned multiple conservation scenarios to identify ecologically relevant and cost-effective priority areas for protection of threatened marine vertebrate species in Brazil (Figure 1), which has the second largest exclusive economic zone (EEZ) in South America and the 11th largest in the world. Specifically, we investigated:

1. Whether it is possible to reconcile conservation and socio-economic objectives without significant losses in the representation of threatened species in the scenarios of MPA expansion.
2. The extent to which the trade-offs between representation of threatened species and opportunity costs differ among four relevant socio-economic activities (small and large-scale fishing, seabed mining, and oil and gas exploration).

For this, multicriteria analyses were applied in an integrated socio-economic–ecological assessment of the performance of MPA networks with different configurations.

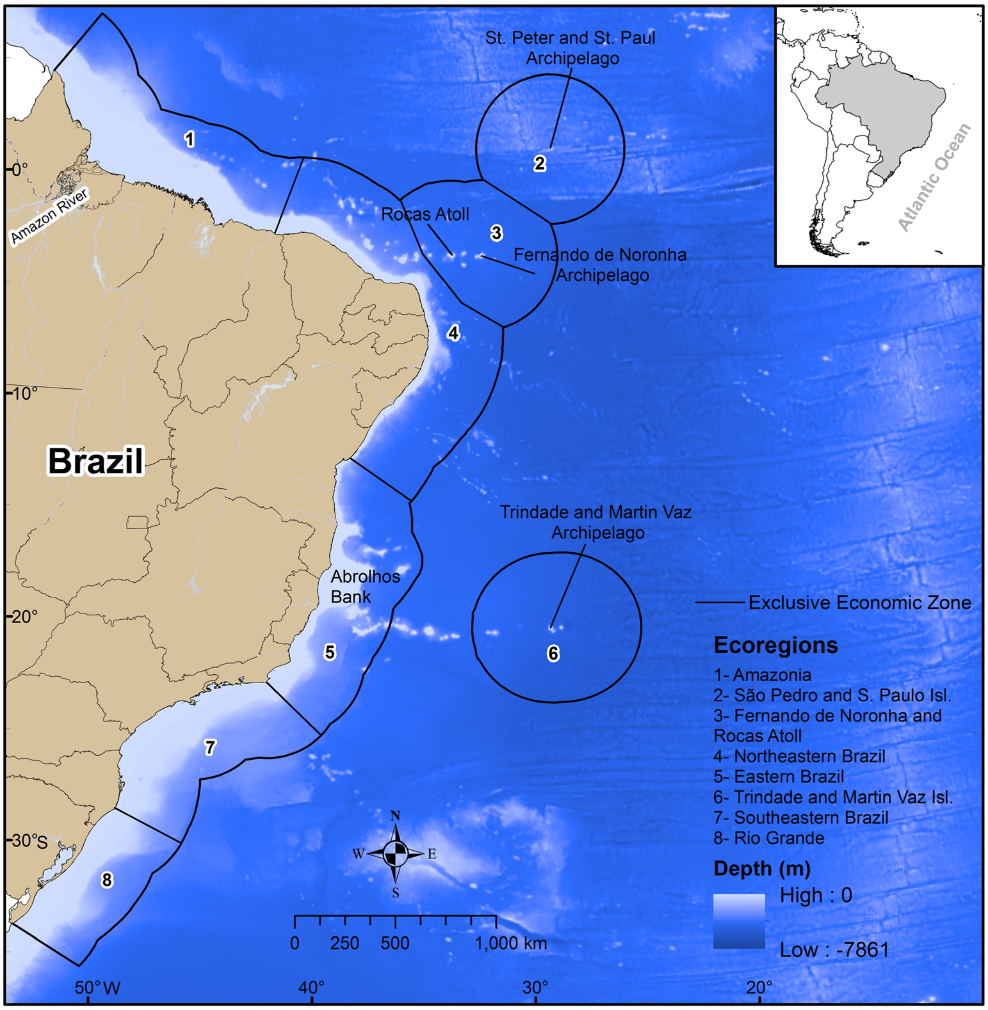

FIGURE 1 Map of the study area in Brazilian jurisdictional waters showing the locations mentioned in the text and marine ecoregions (sensu Spalding et al., 2007). Bathymetric data from the GEBCO_2014 grid, v. 20150318 (http://www.gebco.net)

2 | MATERIAL AND METHODS

2.1 | Study region

The study region (Figure 1) has a coastline of ~8,700 km and extends from the coast out to the outer limit of Brazil's EEZ, covering an area of 3.5 × 10^6 km^2 (Ministério do Meio Ambiente [MMA], 2008). The region encompasses an enormous diversity of ecosystems distributed in eight marine ecoregions (Figure 1). The northernmost ecoregion (i.e. Amazonia) harbours the largest continuous mangrove area in the world (Souza Filho, 2005) and is notable for the high freshwater discharge from the Amazon River and other large estuaries (Knoppers, Ekau, Figueiredo Júnior, & Soares-Gomes, 2002). The Northeastern Brazil ecoregion receives a relatively low riverine input and has a narrow continental shelf dominated by biogenic carbonate sediments (Knoppers, Ekau, & Figueiredo, 1999). Fringing and submerged biogenic reefs are characteristic features of this ecoregion (Knoppers et al., 2002). The Eastern Brazil ecoregion supports the largest and most speciose coral reef system in the south-west Atlantic, the Abrolhos Bank (Leão & Kikuchi, 2001). The Southeastern Brazil ecoregion comprises large estuarine bays, mangroves, rocky shores, and hundreds of coastal islands (Knoppers et al., 2002). The region also corresponds to the southern limit of distribution of mangrove forests (Soares, Estrada, Fernandez, & Tognella, 2012), rhodolith beds (Pascelli et al., 2013), and coralline reefs (Pereira-Filho et al., 2019). Lastly, the Rio Grande ecoregion is dominated by extensive sand beaches, saltmarshes, and lagoons. The large Patos-Mirim lagoon system is the most remarkable feature of this ecoregion (Asmus, 1998). The three other ecoregions encompass seamounts, deep-sea ecosystems, and shallow reefs surrounding oceanic islands, which are notable for their high level of endemism (Pinheiro et al., 2018).

Human population density is highest in the coastal zone, which accounts for close to 27% of Brazil's population (Instituto Brasileiro de Geografia e Estatística, 2011). This has contributed to the scaling up of anthropogenic stressors on coastal ecosystems, including excessive fishing effort (Frédou, Ferreira, & Letourneur, 2009), untreated sewage discharge (Pinheiro et al., 2019), and the installation of industrial and harbour complexes (Pinheiro et al., 2019). Deep-sea ecosystems also face increasing threats from extractive offshore activities, such as large-scale fishing, mining, and oil and gas production (Almada & Bernardino, 2017; Pinheiro et al., 2019).

2.2 | Conservation features

Distribution maps of 126 conservation features targeted by national (MMA, 2014a) and international policies (CBD, 2010) were used in the analyses. The features consisted of 97 fish species, eight marine mammals, eight seabirds, and five marine turtle species, as well as eight marine ecoregions (Supporting Information Table S1). All species considered are nationally threatened with extinction; that is, listed as vulnerable, endangered, or critically endangered in MMA (2014b, 2014c). Instead of representing the species extent of occurrence, which often encompasses large portions of inappropriate habitats, distribution maps corresponded to key areas for species survival (breeding, foraging, or nursery areas) or their geographic distribution within the appropriate depth ranges.

For marine turtle species, the depth range inhabited by females during the inter-nesting period (the period of time between subsequent attempts to lay their clutches of eggs on nesting beaches, during a breeding season) was mapped in areas adjacent to the main nesting beaches in Brazil (described in Marcovaldi, Santos, & Sales, 2011). Information about the habitat depth range for each species was extracted from satellite-tracking studies of females nesting on Brazilian grounds (see references in Supporting Information Table S1). All five marine turtle species recorded in the country were included in this study.

To define the distribution of seabirds, a circular buffer with radius equal to the maximum breeding foraging distance (in kilometres) of each species was created around established breeding colonies at Brazil's islands. The locations of the breeding colonies were obtained from scientific literature and BirdLife International (see references in Supporting Information Table S1). As for marine turtles, the maximum foraging distance of each species from the breeding colonies came from satellite-tracking studies (Supporting Information Table S1). About 11 other seabird species are also threatened in Brazil (MMA, 2014b), but the data available were not adequate to determine their distribution.

Spatial distribution data of breeding, foraging, or nursery grounds for most marine fish species threatened with extinction in Brazil are currently incomplete or not available at the national scale. Hence, the geographic distribution of these fish species within Brazil's marine waters was determined using occurrence records and depth range information obtained from FishBase (Froese & Pauly, 2017), the International Union for Conservation of Nature (IUCN), and the literature (Table S1). Species distributions were first digitized in the form of range maps. These maps were then refined based on the depth limits reported for the species. Bathymetric lines representing each fish depth range were superimposed over their range maps. The area between the minimum and maximum depths used by the species was then clipped off from range maps. This resulted in distribution maps containing only the depth range inhabited by each species. Because *Thunnus thynnus* has not been found in Brazil for a long time (Collette et al., 2011), only 97 of the 98 marine fish species that are currently threatened in the country were mapped. The complete list of the 97 species analysed is provided in Supporting Information Table S1.

For marine mammals, distribution maps were produced as for fishes. Since some mammals have a notoriously disjunct distribution along the coast (e.g. *Pontoporia blainvillei*), areas with no occurrences of such species were excluded from the maps. Information on the species' depth and geographical ranges came from the IUCN and the literature (see Supporting Information Table S1). All marine mammal species threatened in Brazil were considered. To refine the distribution of turtle species, fishes, and mammals, the bathymetry of the study area was obtained from the *Companhia de Pesquisa de Recursos Minerais* of the Brazilian government (http://www.cprm.gov.br). These data are derived from the SRTM 30 Plus V.8.0 (Becker et al., 2009) and have a spatial resolution of about 900 m.

All eight marine ecoregions (Figure 1) were used to provide a broad-scale representation of ecosystems and biodiversity patterns along the study area, as recommended by CBD (2010). Each one is characterized by a distinct suite of ecosystems, topographic and geochemical features. These ecoregions are the smallest-scale units in the Marine Ecoregions of the World system and represent areas of relatively homogeneous species composition (for details, see Spalding et al., 2007).

Distribution maps were overlaid onto a grid with 5,275 $0.25^\circ \times 0.25^\circ$ cells (~27.7 km of latitude and longitude near the equator) covering the study region. We then built binary maps for each conservation feature assuming it is present in cells that overlap with any portion of its distribution range and it is absent elsewhere. These maps were used as inputs for spatial prioritization analyses.

2.3 | Socio-economic features

To minimize opportunity costs incurred when implementing MPAs, the three main socio-economic activities in Brazilian jurisdictional waters were included in the study: commercial fishing (small- and large-scale), oil/gas operations, and mining.

Georeferenced data on small-scale fishing were obtained from a global model of artisanal fisheries catches (Halpern et al., 2008). This model combines data about annual artisanal landings, population density within the coastal zone and distance from nearest land, to estimate the percentage of national total catches into 1 km^2 cells covering the continental shelf (depth < 200 m) of the world's maritime countries. It assumes, therefore, an exponential decay of catch with distance from land given most artisanal fishing occurs in relatively small, sometimes non-motorized vessels, which tend to fish close to the coast. Data were rescaled to a $0.25^\circ \times 0.25^\circ$ grid, by summing the estimated proportions within each cell of total catch in Brazilian waters. These data were used as a proxy for small-scale fishing effort (Figure 2).

Large-scale fishing effort was mapped using the most recent processed data available (2010), from the Brazilian programme for satellite tracking of fishing vessels (*Programa nacional de rastreamento das embarcações pesqueiras por satélite* – PREPS) at a spatial resolution of 0.016° (Ministério da Pesca e Aquicultura, 2012). Adherence to this programme is mandatory in Brazil for fishing vessels equal to or larger

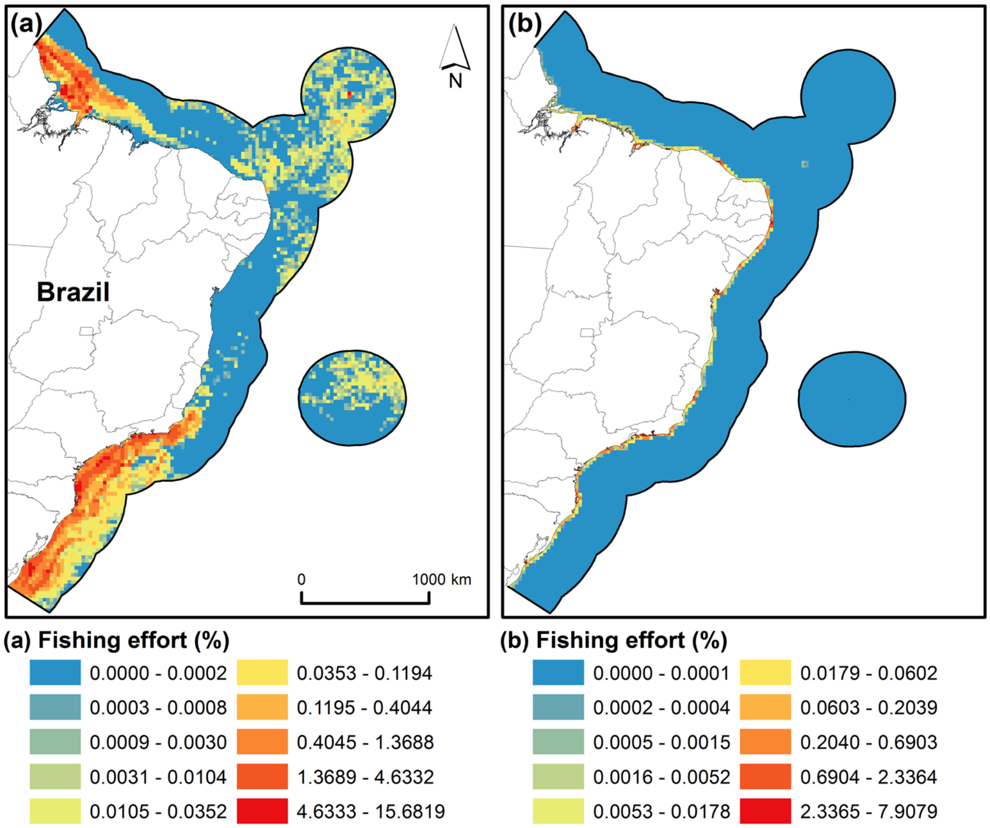

FIGURE 2 Fishing effort of (a) large and (b) small-scale commercial fisheries in Brazilian jurisdictional waters. Large-scale fishing effort represents the summed proportions of the total number of signals within each grid cell for 12 fisheries; small-scale fishing effort represents the estimated proportion of the total catch in Brazilian waters within each cell (see Section 2 for details)

than 15 m, or that have a gross tonnage equal to or larger than 50 tons, as well as for all vessels – independent of their size – from the following commercial fisheries: southern red snapper (*Lutjanus purpureus*), deep sea red crab (*Chaceon notialis*), deep sea crab (*Chaceon ramosae*), blackfin anglerfish (*Lophius gastrophysus*), octopus (*Octopus* spp.), Brazilian codling (*Urophycis mystacea*), silvery John Dory (*Zenopsis conchifer*), Argentine hake (*Merluccius hubbsi*), Argentine shortfin squid (*Illex argentinus*), and lobsters (*Panulirus argus* and *Panulirus laevicauda*).

Data from *Programa nacional de rastreamento das embarcações pesqueiras por satélite* are provided separately for 12 fishing activities, established on the basis of the fishing gear, target species, and areas of operation. The data represent the density of georeferenced signals emitted at 1 h by 905 vessels authorized for the aforementioned fisheries. It includes only signals from vessels in fishing areas and with a speed compatible with fishing operations, which varies according to the fishing activity and vessel characteristics (Ministério da Pesca e Aquicultura, 2012). To determine the relative fishing effort within grid cells for each large-scale fishery, the number of signals in each grid cell was divided by the total number of signals emitted by all vessels of one fishery. The relative fishing effort for each of the 12 fisheries for each grid cell was summed and used as a surrogate for large-scale commercial fishing effort (Figure 2).

Shapefiles of offshore areas currently (June 2017) leased for the oil and gas industry were obtained from Brazil's National Agency of Petroleum, Natural Gas and Biofuels (http://app.anp.gov.br/webmaps/). These areas were divided into two categories: 'exploration blocks' leased for prospecting (e.g. seismic surveys and well drilling) for hydrocarbon reserves, and 'production fields', which consist of areas with commercially viable sources of oil and gas discovered during the exploration phase, already producing, or where the infrastructure necessary to production is being developed (Figure 3). The proportion of each grid cell overlaid on oil and gas exploration/production areas was quantified and used as a measure of the cell's importance for the hydrocarbon industry.

Data on offshore mining areas were downloaded from the homepage of Brazil National Department of Mineral Production (http://www.dnpm.gov.br/assuntos/ao-minerador/sigmine) as of July 2017. The data included areas licensed for mineral research for a period of 2 or 3 years (i.e. at research authorization phase) and areas being licensed for or already producing mineral resources (i.e. at mining concession or extraction phases) discovered in the first phase (Figure 3). The proportion of each grid cell occupied by mining areas was determined and used as a measure of the occurrence of mining operations within cells.

2.4 | Conservation prioritization analyses

The Zonation v.4.0 software (Di Minin, Veach, Lehtomäki, Montesino-Pouzols, & Moilanen, 2014; Moilanen et al., 2005) was used to identify spatial priorities for expanding the existing MPAs in Brazil. The algorithm in Zonation produces a complementarity-based hierarchical ranking of conservation priorities for all cells over an area, accounting

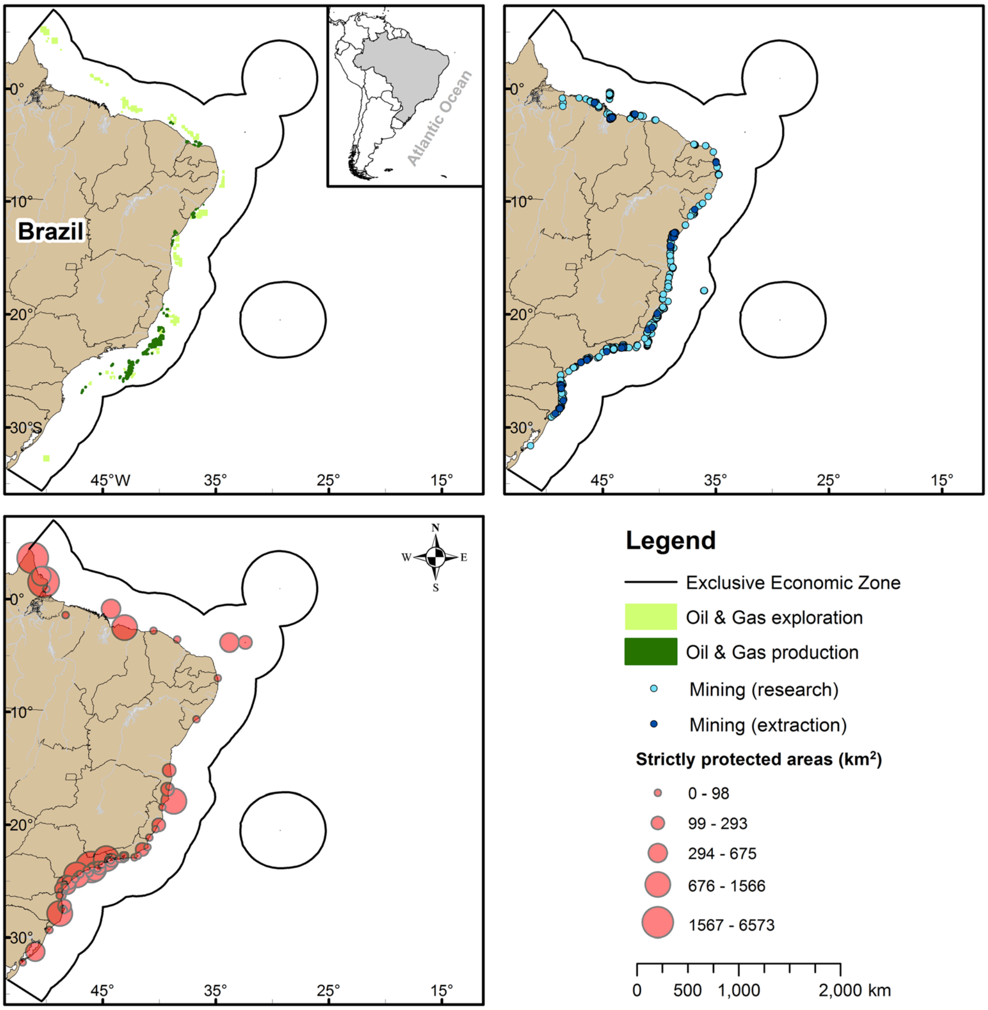

FIGURE 3 Offshore areas leased for the oil and gas industry, mining areas, and the strictly protected areas established (as of February 2018) in Brazilian jurisdictional waters

for the range size and the weight set for features (Di Minin et al., 2014). Priority ranking was determined via a reverse heuristic algorithm that sequentially removes single cells based on the criterion of minimization of marginal loss (i.e. the relative importance of a cell compared with all others). The cells with the smallest marginal loss are removed first, and cells with high marginal loss (i.e. top-ranked areas for conservation) are removed last. The additive-benefit function, which aggregates conservation value for all species (or other conservation features) within a cell, was used to calculate the marginal loss. The function, therefore, tends to prioritize cells with the greater number of species, maximizing the average coverage over all species. This option is indicated when the data are considered to be a surrogate for biodiversity as a whole (see Lehtomäki & Moilanen, 2013 for details).

Feature weighting allows Zonation to maintain a balance among features in analyses outcomes. In the present work, both positive (for conservation features) and negative (for socio-economic features) weights were applied in order to produce a spatial priority ranking that accounts for the needs of multiple seascape uses, and thus avoid potential conflicts between them (Moilanen et al., 2011). All groups of species (turtles, birds, mammals, and fishes) received the same aggregate weight (W_j = 12.5). Within each group each species weighted the *n*th part of the total (e.g. for turtles, each of the five species weighted 12.5/5 = 2.5) (Table 1). Ecoregions received an aggregate weight equivalent to that of the four species groups summed (W_j = 50). Large and small-scale fishing also were weighted equally (W_j = –25), whereas exploration blocks (W_j = –8.33333) received half the weight of production fields (W_j = –16.6667). Mining areas at research phase (W_j = –8.33333) also received half the weight of mining areas at concession/extraction phases (W_j = –16.6667). The sum of the weights for conservation (W_j = 100) and economic (W_j = –100) features was zero (as in Faleiro & Loyola, 2013; Moilanen et al., 2011).

Prioritization analyses were undertaken considering six spatial planning scenarios in a way that allows examining trade-offs between conservation and socio-economic features. First, the software algorithm was free to determine which cells had the best conservation return based solely on biodiversity data and weights given to them, without considering any human activity ('unconstrained' scenario; socio-economic activities layers were included in this analysis with a zero weight). Next, the large-scale fishing effort ('large-scale fishing' scenario), the small-scale fishing effort ('small-scale fishing' scenario), the oil and gas activities ('oil and gas' scenario) and the mining activities ('mining' scenario) were incorporated separately into the prioritization analyses. Finally, an inclusive multicriteria prioritization incorporating all conservation

TABLE 1 List of biodiversity and socio-economic features included in the prioritization scenarios and their respective weights assigned in Zonation. The plus sign indicates that the feature was included in the scenario

		Scenario					
Feature	**Weight**	**Unconstrained**	**Large-scale fishing**	**Small-scale fishing**	**Oil and gas**	**Mining**	**All activities**
Biodiversity							
Fishes	+12.5	+	+	+	+	+	+
Mammals	+12.5	+	+	+	+	+	+
Seabirds	+12.5	+	+	+	+	+	+
Turtles	+12.5	+	+	+	+	+	+
Ecoregions	+50	+	+	+	+	+	+
Socio-economic							
Large-scale fishing	−25		+				+
Small-scale fishing	−25			+			+
Oil and gas areas	−25				+		+
Mining areas	−25					+	+

and socio-economic features was undertaken to identify areas where several objectives could be achieved simultaneously ('all-activities' scenario) (Table 1).

In all scenarios, cells that overlap with existing strictly protected areas (Figure 2) were treated as top priorities for conservation. Multiple-use MPAs were not considered given the uncertainty around their effectiveness for biodiversity conservation (Sala et al., 2018). Thus, results indicated broad priority areas to expand established MPAs in Brazil by considering the scale of representation of conservation features in sites that are already protected. All strictly protected areas (i.e. IUCN categories I–III or integral protection units, as defined by Brasil, 2000) that include any marine space and were officially established prior to the period of the analyses (December 2017), were included.

Recently, two oceanic MPAs, covering about 116,418.5 km^2, were declared that moved Brazil closer to meeting the quantitative area requirement of Target 11 (Giglio et al., 2018; Magris & Pressey, 2018). Therefore, we investigated the overlap of the best 10% of cells under each conservation scenario with each of these MPAs: the Natural Monuments of Trindade and Martin Vaz Islands and Columbia seamount (TMV-MPA) and of St Peter and St Paul Archipelago (SPSP-MPA) (Brasil, 2018a, 2018b; Appendix A). Shapefiles of protected areas were acquired from the Brazilian Ministry of the Environment (http://mapas.mma.gov.br/i3geo/datadownload.htm).

2.5 | Scenario comparisons

Results from the prioritization analyses were compared considering the top-ranked 10% of seascape, in line with the Aichi biodiversity Target 11 (CBD, 2010). Comparisons were based on: (1) the spatial overlap among grid cells prioritized in different scenarios; (2) performance curves showing the level of representation of biodiversity features and the percentage of each socio-economic activity displaced as a function of geographic protection of seascape under each prioritization scenario; and (3) the number of non-protected species under each prioritization scenario. Finally, the proportion of species' distributions represented in the prioritization scenarios was tested for normality by the Shapiro–Wilk test, which showed that data were not normally distributed ($P < 0.01$). In this case, the significance of differences in species distribution's coverage among planning scenarios was determined using non-parametric Friedman tests, followed by pairwise Wilcoxon signed rank tests when the null hypothesis of equal coverage was rejected.

3 | RESULTS

Considering the potential for synergies between CBD Aichi Targets 11 and 12, high-priority areas for conservation were identified from the northern equatorial coast (2°N) to the subtropical latitudes at Brazil's southern boundary (33°S), and encompassed both coastal and oceanic areas (Figure 4a–f). The scenario including only threatened species and ecoregions (i.e. the unconstrained scenario) identified priorities mainly in neritic waters and around oceanic islands (Figure 4a). In contrast, when all socio-economic features were incorporated into the analysis (i.e. the all-activities scenario), spatial priorities partially shifted to areas off the coast, mainly in the southern part of the country (Figure 4f).

The spatial overlap averaged 76.9% across the prioritization scenarios, varying from 61.9% (unconstrained scenario vs. all-activities scenario) to 93.1% (unconstrained scenario vs. oil and gas scenario) (Table 2). Large areas of the coast of Maranhão, along the northern coast of Bahia, and on, or close to, the Abrolhos Bank, surrounding the Rocas Atoll and some oceanic islands (Fernando de Noronha, Trindade and Martin Vaz), were prioritized under all scenarios

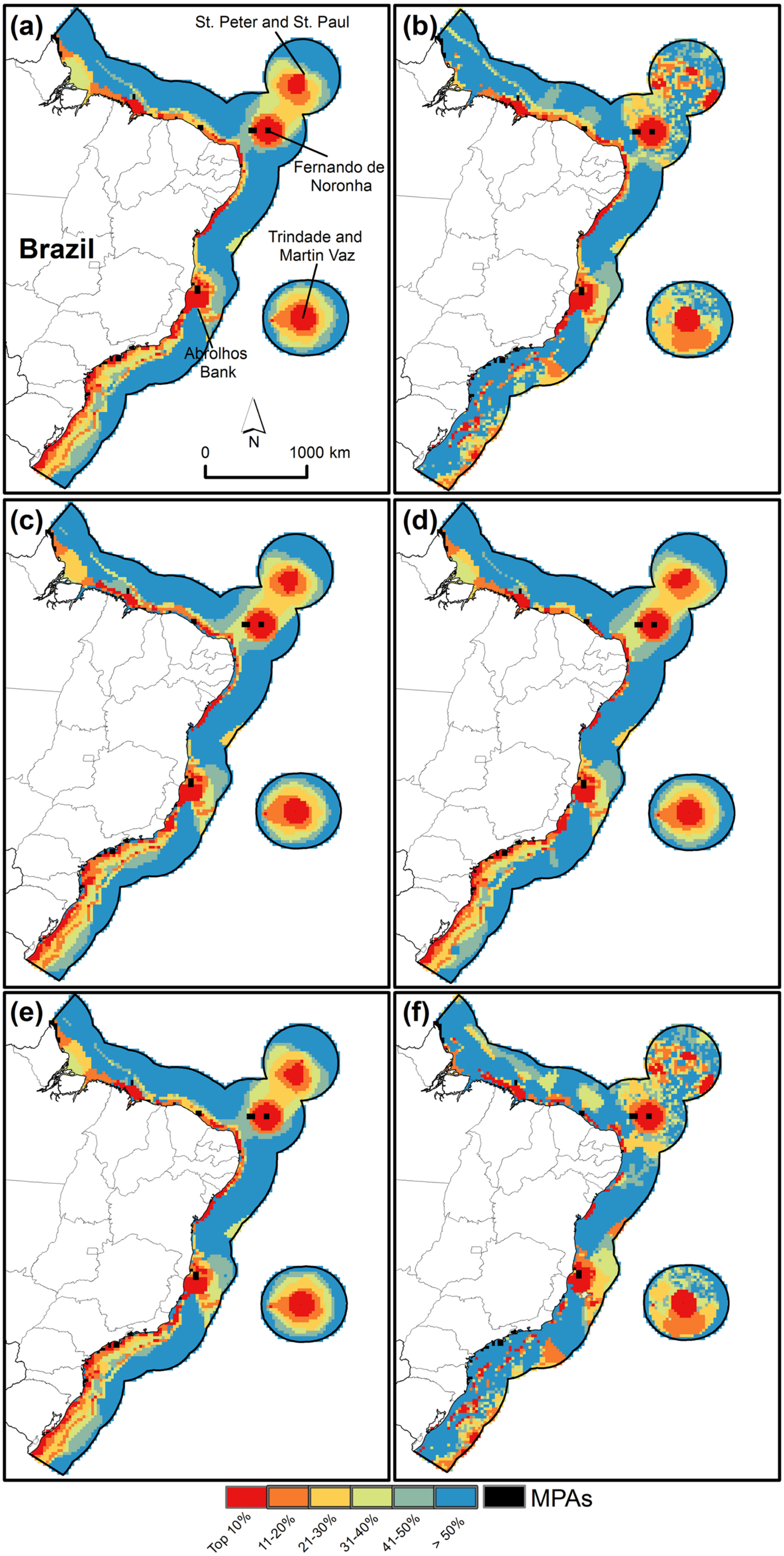

FIGURE 4 Hierarchical ranking of conservation priorities determined for the scenarios (a) unconstrained, (b) large-scale fishing, (c) small-scale fishing, (d) oil and gas, (e) mining, and (f) all activities. Areas in red are priorities for marine protected areas (MPAs) expansion under the Convention on Biological Diversity Aichi target of 10%. Strict MPAs included in prioritization analyses are also shown

TABLE 2 Percentage of priority cells for conservation shared between pairs of prioritization scenarios

	UN	LF	SF	OG	MI
LF	74.0				
SF	86.7	69.5			
OG	93.1	71.4	83.5		
MI	91.8	71.7	87.5	89.5	
All	61.9	79.3	65.3	64.7	64.5

UN: unconstrained scenario; LF: large-scale fishing scenario; SF: small-scale fishing scenario; OG: oil and gas scenario; MI: mining scenario; All: all activities scenario.

(Figure 4a–f). The priority cells for conservation common to all scenarios corresponded to 5.8% of the study area (Figure 5).

The unconstrained scenario covered on average 40% of conservation features' distribution (median: 29.7%). The addition of socio-economic data in the analysis reduced the mean representation of conservation features in all scenarios relative to the unconstrained scenario (Figure 6a–f). However, this reduction varied widely: −2.2% in the oil and gas scenario, −4.5% in the mining scenario, −7.2% in the small-scale fishing scenario, −19.7% in the large-scale fishing scenario, and as much as −31.5% in the all-activities scenario. There were significant differences among scenarios in the representation of bird (Friedman test: Q = 11.6, P = 0.040), fish (Q = 90.8, P < 0.001), and turtle species (Q = 19.4, P = 0.002), but not for mammal species (Q = 5.0, P = 0.126) and ecoregions (Q = 2.5, P = 0.767). Pairwise Wilcoxon tests indicated that the unconstrained scenario provided a significantly greater coverage for bird species than the oil and gas scenario did, and for fish species when compared with all other scenarios (except the oil and gas scenario; Figure 7). Also, the difference in representation between the unconstrained and other scenarios for turtle species was marginally non-significant (P = 0.068 for all pairwise

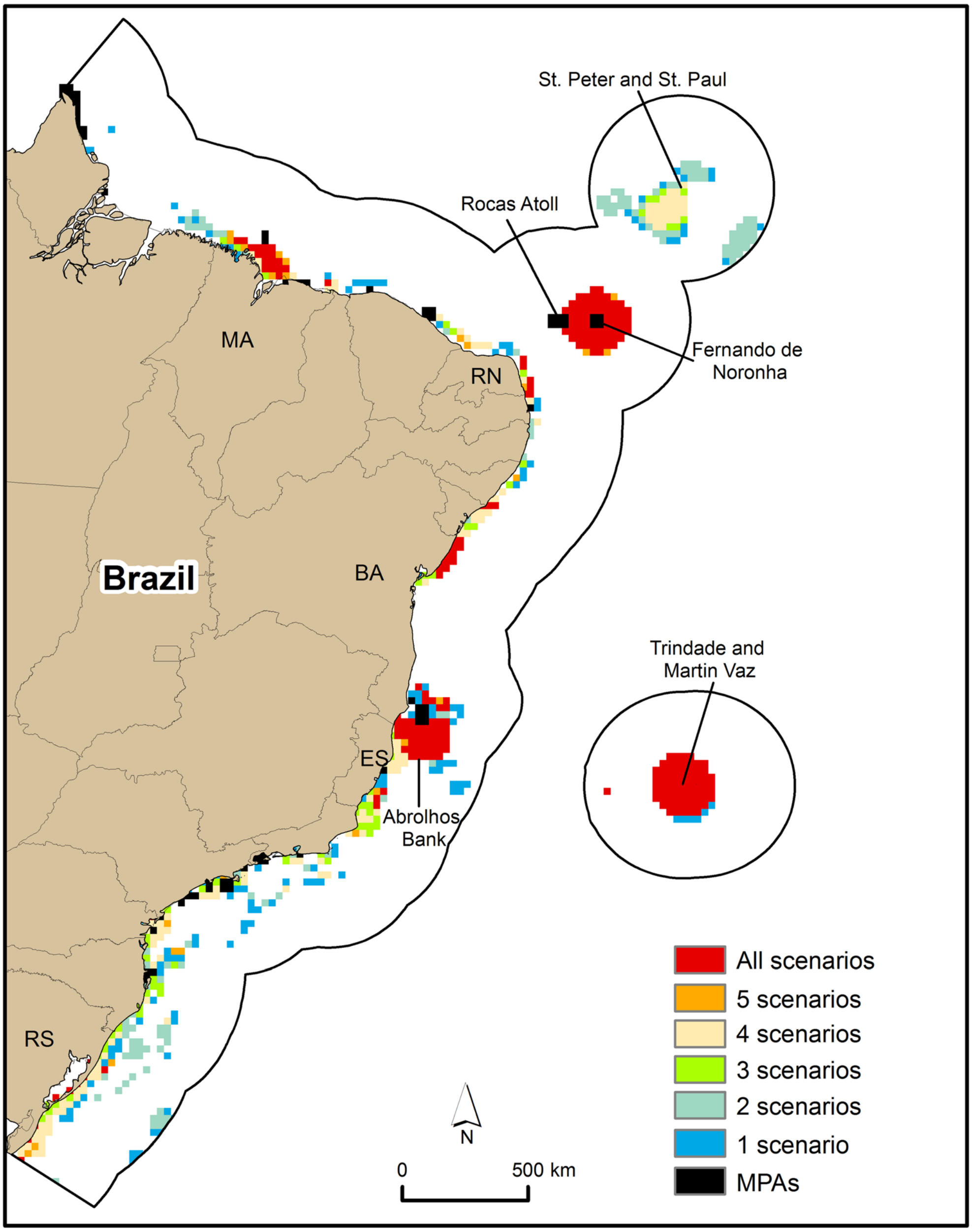

FIGURE 5 Spatial overlap in the top 10% of cells for conservation selected under the six prioritization scenarios

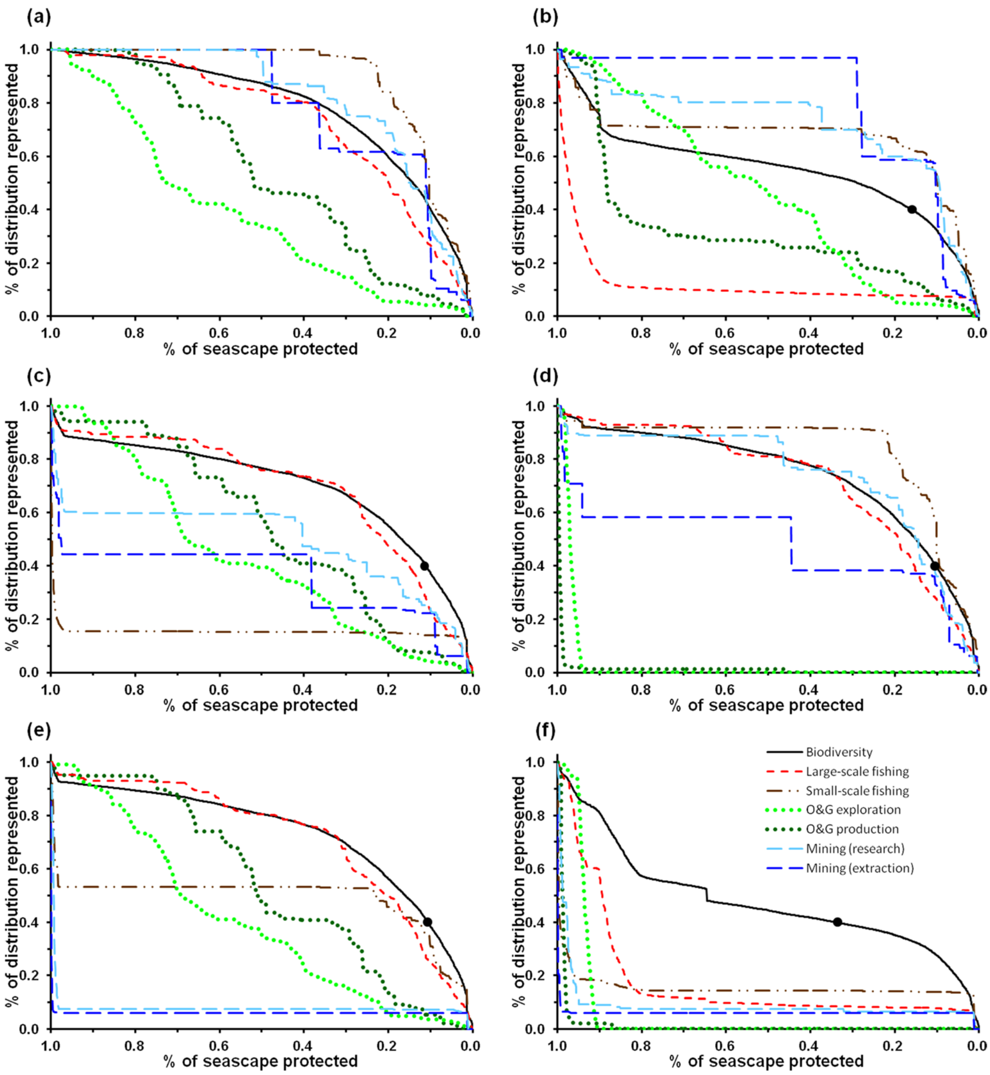

FIGURE 6 Performance curves showing the proportion of biodiversity features represented and the proportion of each socio-economic activity displaced at any fraction of seascape protected under the scenarios (a) unconstrained, (b) large-scale fishing, (c) small-scale fishing, (d) oil and gas, (e) mining, and (f) all activities. The black dot shows the mean representation of biodiversity features (i.e. 40%) attained under the unconstrained scenario at 10% of seascape (shown in vibrant red on Figure 4a)

comparisons, except that with the commercial fishing scenario, $P = 0.317$; Figure 7).

From the 118 threatened species considered, five and seven fish species would not be protected under the large-scale fishing and the all-activities scenarios respectively. These are the St Paul's Rocks or Mid-Atlantic Ridge endemics oblique butterflyfish (*Prognathodes obliquus*; VU), Saint Paul's gregory (*Stegastes sanctipauli*; VU), red scorpionfish (*Scorpaenodes insularis*; VU), salmon-spotted jewelfish (*Choranthias salmopunctatus*; VU), and Smith's triplefin (*Enneanectes smithi*, VU), under both the commercial fishing and the all-activities scenarios, and additionally the sea catfish (*Paragenidens* aff. *grandoculis*; CR) and the striped wormfish (*Cerdale fasciata*; EN) under the all-activities scenario. All species would be at least partially protected under the other four scenarios.

On the other hand, there were major differences among scenarios in their opportunity costs (Figure 6a–f). The inclusion of all socio-economic features in the all-activities scenario reduced the representation of large (−18.9%) and small-scale (−28.8%) fishing effort, oil and gas exploration (−4.2%) and production areas (−7.8%), and mineral research (−30.9%) and production areas (−22.7%), relative to the unconstrained scenario. When each socio-economic feature was considered individually (Figure 6b–e), their representation levels were not substantially different from that achieved under the all-activities scenario (Figure 6f). Large-scale fishing was the main constraint to representation of threatened species and ecoregions and produced the lowest spatial overlap with the unconstrained scenario (Table 2). The second-largest constraint was small-scale fishing (Table 2).

To achieve the same mean representation of conservation features that the unconstrained scenario would protect at the best 10% of seascape (Figure 6a), it would be necessary to protect about 16% of Brazil's EEZ under the large-scale fishing scenario (Figure 6b), 11% under the small-scale fishing scenario (Figure 6c), 10.3% under the oil and gas scenario (Figure 6d), 10.7% under the mining scenario (Figure 6e), and 34% under the all-activities scenario (Figure 6f). Notably, the proportion of socio-economic features captured at the 34% seascape target for conservation, under the all-activities scenario, was lower than the proportion represented at the 10% seascape target, under the unconstrained scenario: 8.5% for large-scale fishing (vs. 26.6% under the unconstrained scenario), 14.2% for small-scale fishing (vs. 42.6%), 7.4% for mineral research areas (vs. 37.4%), 6.1%

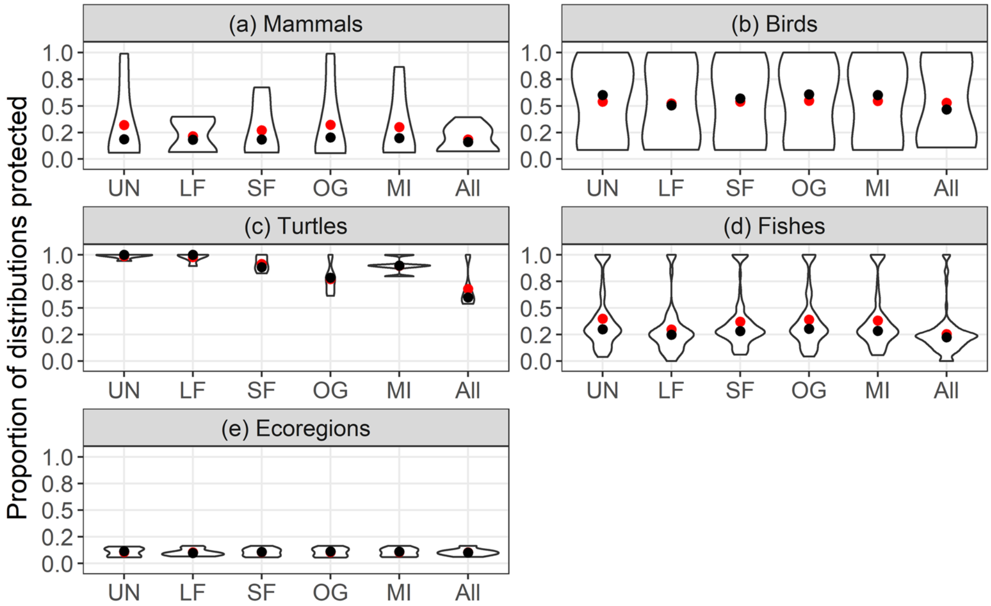

FIGURE 7 Proportion of the distribution of (a) eight mammals, (b) eight seabirds, (c) five marine turtles, (d) 97 fish species, and (e) eight ecoregions protected in the top 10% of cells for conservation under the six prioritization scenarios. The red dot indicates the mean and the black dot the median. UN: unconstrained scenario; LF: large-scale fishing scenario; SF: small-scale fishing scenario; OG: oil and gas scenario; MI: mining scenario; All: all activities scenario

for mineral production areas (vs. 28.8%), and 0% for both oil and gas exploration and production areas (vs. 4.2% and 7.8% respectively).

There was a substantial discrepancy between our spatial priorities and the recently created TMV-MPA and SPSP-MPA (Appendix 1). Each MPA individually protects 8.1% or less of the prioritized (top 10%) areas under each scenario (Table 3a). The SPSP-MPA covers a slightly greater proportion of the best areas for protection based on the unconstrained scenario than the TMV-MPA does, whereas the latter includes a greater proportion of the prioritized areas under the all-activities scenario than the former does. Together, the new MPAs protect from 4.5% (for the all-activities scenario) to 12.5% (for the mining scenario) of the best areas for conservation identified in the present work. The proportion of the SPSP-MPA overlapped by the best 10% of the seascape ranged from 0% (by the large-scale-fishing scenario) to 58.9% (by the mining scenario). For the TMV-MPA, the overlap varied little and ranged between 22.3% (by five scenarios) and 23.4% (by the all-activities scenario). Considering both MPAs, the coverage varied from 13.2% (by the large-scale fishing scenario) to 37.4% (by the mining scenario) (Table 3b).

TABLE 3 Percentage of priority areas for conservation (top 10%) under each prioritization scenario (a) overlapping and (b) overlapped by the St Peter and St Paul (SPSP) and Trindade and Martin Vaz (TMV) marine protected areas

Scenario	SPSP	TMV	Total
(a) Overlapping with			
Unconstrained	6.8	4.5	11.4
Large-scale fishing	0.0	4.6	4.6
Small-scale fishing	7.3	4.3	11.7
Oil and gas	6.9	4.5	11.5
Mining	8.1	4.4	12.5
All activities	0.01	4.5	4.5
(b) Overlapped by			
Unconstrained	48.2	22.3	33.1
Large-scale fishing	0.0	22.3	13.2
Small-scale fishing	54.5	22.3	35.6
Oil and gas	49.5	22.3	33.6
Mining	58.8	22.3	37.4
All activities	0.05	23.3	13.8

4 | DISCUSSION

Although there is a clear conflict between extractive sectors and the achievement of global commitments for biodiversity, the results revealed that there is scope for reconciling solutions, demonstrating that decision-makers do not need to design MPAs in areas of low importance for conservation in their pursuit of policy targets (Magris & Pressey, 2018; Spalding et al., 2016). For instance, when socio-economic activities were integrated into the scenarios, the areas with highest priority for conservation changed only partially towards lower cost areas (e.g. offshore areas in the south-east and south Brazilian coast), where both small and large-scale fishing effort is relatively lower and there is no seabed mining (Figures 2, 3). Such displacement would substantially decrease opportunity costs and alleviate conservation conflicts with all resource users, but especially with one of the most widespread and important extractive activities worldwide: fishing (Kroodsma et al., 2018). The incorporation of socio-economic information would thus help to achieve a balance between development and conservation goals, as evidenced by studies in other countries with broad dependency on sea resources (Jumin et al., 2017; Li et al., 2019; Mazor et al., 2014). However, explicitly including the broad range of qualitative elements of the Aichi Target 11 (Rees, Foster, Langmead, Pittman, & Johnson, 2018) could create additional trade-offs, not addressed here, among planning objectives.

In Brazil, as in many countries around the world (Klein et al., 2008; Mazor et al., 2014), large and small-scale fisheries are the major constraints for protecting high-priority areas. Given the growing geographical expansion of marine fisheries (Swartz, Sala, Tracey, Watson, & Pauly, 2010), the barriers imposed by the fishing sector to marine protection should increase in the future. Mining operations are also nationally widespread and had the second greatest influence on the representation of conservation features, possibly because of their concentration in nearshore waters, which are rich in restricted-range species (Pinheiro et al., 2018). The oil and gas sector, on the other hand, caused only a small loss in representation of threatened species, owing to the relatively broad distributions of most species overlapping with concession areas. This result is consistent with previous findings (Vilar et al., 2015) and reinforces that it is possible to expand the country's system of MPAs with minimal interference on the oil and gas sector—responsible for about 12% of the national gross domestic product in 2011 (Confederação Nacional da Indústria, 2012). However, with the expected lease of new oil and gas blocks, the impacts from this sector would worsen in the near future, in particular on vulnerable deep-sea ecosystems (Almada & Bernardino, 2017).

Some have de facto advocated the consideration of economic activities in spatial conservation planning to reduce opportunity-costs and conflicts (e.g. Mazor et al., 2014; Naidoo et al., 2006). This rationale rests on the implicit assumption that there is considerable spatial homogeneity in species composition and, hence, that both areas being and not being used for extractive purposes provide equivalent conservation benefits. The present work, however, shows that avoiding tensions and costs would have some serious biological consequences, such as the decline in conservation features representation under all conservation scenarios that include extractive activities (Figure 6a–f). The reduced performance of scenarios respecting socio-economic uses in comparison with the unconstrained scenario can be explained by the spatial overlap between priority sites for conservation actions and areas currently used for extractive purposes (i.e. most importantly fishing, but also seabed mining and oil and gas production). The inclusion of socio-economic data in the planning process may, therefore, favour relatively cheap areas to the exclusion of costly but biologically relevant sites (Jenkins & Van Houtan, 2017; Pressey, Weeks, & Gurney, 2017).

Areas of high relevance for conservation, which are currently under severe (and low) anthropogenic threats, were revealed by comparing the output from the unconstrained scenario with those obtained when socio-economic activities were integrated. For example, St Peter and St Paul's Archipelago, which harbours a high number of endemic and globally threatened species (Pinheiro et al., 2018), became unselected for conservation after including large-scale fishing in the prioritization process (Figure 4a,f). This site is irreplaceable and has already suffered fisheries-induced functional extinctions (Luiz & Edwards, 2011). Therefore, a ban of fishing activities and effective enforcement are required (Giglio et al., 2018). Similarly, nearshore areas of the south-east and south Brazilian coast that were prioritized under the unconstrained scenario and in other studies (MMA, 2007; Vilar et al., 2015; Vilar, Joyeux, & Spach, 2017) were replaced by offshore areas under the all-activities scenario. This shift occurred because the small and large-scale fishing efforts are considerably lower in these offshore areas (Figure 2) that are also free from mineral extractive activities (Figure 3). Although these areas are important for the conservation of deep-sea benthic ecosystems and pelagic species (Almada & Bernardino, 2017; Krüger et al., 2017), they are currently far less impacted by extractive activities than nearshore areas (Figures 2, 3; Halpern et al., 2015). Protecting more vulnerable areas close to the coast, rather than offshore areas with few anthropogenic uses, would then be most effective to mitigate expanding threats for imperilled species (Pressey et al., 2017; Pressey, Visconti, & Ferraro, 2015).

Importantly, over half of the top-ranked 10% of cells for conservation were selected under the six prioritization scenarios, suggesting a potential win–win situation in these cost-effective cells (Figure 5). They correspond to spatial mismatches between extractive activities and biologically valuable areas and highlight opportunities to unify socio-economic and conservation goals. These areas include large and remote portions around Rocas Atoll, Fernando de Noronha, and Trindade and Martin Vaz, where there are many vulnerable species and relatively fewer threats (Halpern et al., 2015; Pinheiro et al., 2018). Emerging evidence shows that reefs with low levels of human disturbance can support high functional diversity and biomass of fishes, representing refuges for sharks and other predators (D'agata et al., 2016; Juhel et al., 2019; Letessier et al., 2019). The implementation of proactive, large-scale initiatives such as the establishment of large no-take MPAs could safeguard these remaining near-pristine marine areas, benefiting even highly mobile marine species (Graham & McClanahan, 2013; Juhel et al., 2019; Speed et al., 2018; White et al., 2017). But the protection of the more threatened areas (including island shelves and slopes) by strictly protected areas should remain a priority, potentially leading to larger positive effects on biodiversity.

The need to minimize negative impacts on socio-economic activities implies that synergies between conservation targets must be identified and secured (Di Marco et al., 2016). However, important gaps in knowledge limit planners' ability to incorporate a broad range of biodiversity aspects in spatial prioritization strategies (Appeltans et al., 2012; Mora, Tittensor, Adl, Simpson, & Worm, 2011). Here, threatened species of vertebrates were selected as surrogates because they are targeted by national (MMA, 2014a) and international conservation policies (CBD, 2010), include key contributors to ecosystems functions (e.g. prey regulation, biocontrol of invasive species, and nutrient cycling), and are in imminent risk of extinction and will probably be lost if no action is taken. Their effectiveness as surrogates for biodiversity, however, is uncertain. We stress the importance of taking a more comprehensive view and including other biodiversity components in the conservation prioritization to more fully realize conservation targets. Among other aspects that could influence the location of the priority areas and exacerbate the conflict between conservation objectives and socio-economic activities are ecological connectivity and the effects of global warming (Magris, Pressey, Weeks, & Ban, 2014). Recognizing that some elements of biodiversity require more protection than others will also make the achievement of conservation objectives more difficult. Future research incorporating these and other relevant factors might help identify more effective MPA networks for the persistence of species.

Brazilian MPAs traditionally have been established without taking into account the basic principles of spatial conservation prioritization (Magris & Pressey, 2018). The race to achieve Aichi targets marks a new opportunity to change this pattern. However, the small overlap between the top ranked areas for conservation in this study and the two new oceanic no-take MPAs (Table 3) supports previous criticisms of the lack of scientific input in the decision-making process (Giglio et al., 2018; Magris & Pressey, 2018). Despite having expanded the country's marine area under legal (but not strict) protection from 1.5% to about 26%, Brazil still needs to increase its MPA system to meet Aichi Targets 11 and 12. Among other conditions, to meet Target 11 the areas protected should 'be ecologically representative', 'including at least 10% of each ecoregion within the country' (CBD, 2010). With 50% (four out of eight) of the country's marine ecoregions still poorly protected (UN Environment Programme World Conservation Monitoring Centre & IUCN, 2018), Brazil would need to establish new MPAs to fulfil its commitments. Our study aiming at maximizing the conservation of threatened marine vertebrates under different scenarios of anthropic use offers a glimpse into the opportunities, and choices, that all maritime countries face and will need to consider to fully achieve Aichi biodiversity targets by 2020 and beyond.

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REFERENCES

Almada, G. V. M. B., & Bernardino, A. F. (2017). Conservation of deep-sea ecosystems within offshore oil fields on the Brazilian margin, SW Atlantic. *Biological Conservation*, *206*, 92–101. https://doi.org/10.1016/j.biocon.2016.12.026

Álvarez-Romero, J. G., Mills, M., Adams, V. M., Gurney, G. G., Pressey, R. L., Weeks, R., … Storlie, C. J. (2018). Research advances and gaps in marine planning: Towards a global database in systematic conservation planning. *Biological Conservation*, *227*, 369–382. https://doi.org/10.1016/j.biocon.2018.06.027

Appeltans, W., Ahyong, S. T., Anderson, G., Angel, M. V., Artois, T., Bailly, N., & Costello, M. J. (2012). The magnitude of global marine species diversity. *Current Biology*, *22*, 2189–2202. https://doi.org/10.1016/j.cub.2012.09.036

Asmus, M. L. (1998). A planície costeira e a Lagoa dos Patos. In U. Seelinger, C. Odebrecht, & J. P. Castello (Eds.), *Os ecossistemas marinho e costeiro do extremo sul do Brasil* (pp. 9–12). Rio Grande, Brazil: Ecoscientia.

Baillie, J., & Zhang, Y. P. (2018). Space for nature. *Science*, *361*, 1051. https://doi.org/10.1126/science.aau1397

Ban, N. C., & Klein, C. J. (2009). Spatial socioeconomic data as a cost in systematic marine conservation planning. *Conservation Letters*, *2*, 206–215. https://doi.org/10.1111/j.1755-263X.2009.00071.x

Becker, J. J., Sandwell, D. T., Smith, W. H. F., Braud, J., Binder, B., Depner, J., … Weatherall, P. (2009). Global bathymetry and elevation data at 30 arc seconds resolution: SRTM30_PLUS. *Marine Geodesy*, *32*, 355–371. https://doi.org/10.1080/01490410903297766

Brasil. (2000). Lei no 9.985, de 18 de julho de 2000. Regulamenta o art. 225, § 1°, incisos I, II, III e VII da Constituição Federal, institui o Sistema Nacional de Unidades de Conservação da Natureza e dá outras providências. *Diário Oficial da União—Seção 1*, *138*, 1–6.

Brasil. (2018a). Decreto no 9.312, de 19 de março de 2018. Cria a Área de Proteção Ambiental do Arquipélago de Trindade e Martim Vaz e o Monumento Natural das Ilhas de Trindade e Martim Vaz e do Monte Columbia. *Diário Oficial da União—Seção 1*, *1*, 1–3.

Brasil. (2018b). Decreto no 9.313, de 19 de março de 2018. Cria a Área de Proteção Ambiental do Arquipélago de São Pedro e São Paulo e o Monumento Natural do Arquipélago de São Pedro e São Paulo. *Diário Oficial da União—Seção 1*, *1*, 3–4.

Butchart, S. H. M., Clarke, M., Smith, R. J., Sykes, R. E., Scharlemann, J. P. W., Harfoot, M., … Burgess, N. D. (2015). Shortfalls and solutions for meeting national and global conservation area targets. *Conservation Letters*, *8*, 329–337. https://doi.org/10.1111/conl.12158

Convention on Biological Diversity. (2010). *COP decision X/2. Strategic plan for biodiversity 2011–2020*. Montreal, QC, Canada: Convention on Biological Diversity.

Confederação Nacional da Indústria. (2012). *A contribuição do setor brasileiro de petróleo, gás e biocombustíveis para o desenvolvimento sustentável no país*. Brasília, DF, Brazil: Confederação Nacional da Indústria.

Collette, B., Amorim, A. F., Boustany, A., Carpenter, K. E., de Oliveira Leite, Jr. N., Di Natale, A., … Uozumi, Y. (2011). *Thunnus thynnus*. The IUCN Red List of Threatened Species 2011: E. T21860A9331546. https://doi.org/10.2305/IUCN.UK.2011-2.RLTS.T21860A9331546.en

D'agata, S., Mouillot, D., Wantiez, L., Friedlander, A. M., Kulbicki, M., & Vigliola, L. (2016). Marine reserves lag behind wilderness in the conservation of key functional roles. *Nature Communications*, *7*, 12000. https://doi.org/10.1038/ncomms12000

Davidson, L. N. K., & Dulvy, N. K. (2017). Global marine protected areas to prevent extinctions. *Nature Ecology & Evolution*, *1*, 0040. https://doi.org/10.1038/s41559-016-0040

De Santo, E. M. (2013). Missing marine protected area (MPA) targets: How the push for quantity over quality undermines sustainability and social justice. *Journal of Environmental Management*, *124*, 137–146. https://doi.org/10.1016/j.jenvman.2013.01.033

Devillers, R., Pressey, R. L., Grech, A., Kittinger, J. N., Edgar, G. J., Ward, T., & Watson, R. (2015). Reinventing residual reserves in the sea: Are we favouring ease of establishment over need for protection? *Aquatic Conservation: Marine and Freshwater Ecosystems*, *25*, 480–504. https://doi.org/10.1002/aqc.2445

Di Marco, M., Butchart, S. H. M., Visconi, P., Buchanan, G. M., Ficetola, G. F., & Rondinini, C. (2016). Synergies and trade-offs in achieving global biodiversity targets. *Conservation Biology*, *30*, 189–195. https://doi.org/10.1111/cobi.12559

Di Minin, E., MacMillan, D. C., Goodman, P. S., Escott, B., Slotow, R., & Moilanen, A. (2013). Conservation businesses and conservation planning in a biological diversity hotspot. *Conservation Biology*, *27*, 808–820. https://doi.org/10.1111/cobi.12048

Di Minin, E., Veach, V., Lehtomäki, J., Montesino-Pouzols, F., & Moilanen, A. (2014). *A quick introduction to Zonation*. Helsinki, Finland: University of Helsinki.

Dinerstein, E., Vynne, C., Sala, E., Joshi, A. R., Fernando, S., Lovejoy, T., … Wikramanayake, E. (2019). A global deal for nature: Guiding principles, milestones, and targets. *Science Advances*, *5*, eaaw2869. https://doi.org/10.1126/sciadv.aaw2869

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*, 216–220. https://doi.org/10.1038/nature13022

Faleiro, F. V., & Loyola, R. D. (2013). Socioeconomic and political trade-offs in biodiversity conservation: A case study of the Cerrado Biodiversity Hotspot, Brazil. *Diversity and Distributions*, *19*, 977–987. https://doi.org/10.1111/ddi.12072

Food and Agriculture Organization of the United Nations. (2014). *The state of world fisheries and aquaculture—Opportunities and challenges*. Rome, Italy: Food and Agriculture Organization of the United Nations.

Ferrier, S., & Wintle, B. A. (2009). Quantitative approaches to spatial conservation prioritization: Matching the solution to the need. In A. Moilanen, K. A. Wilson, & H. P. Possingham (Eds.), *Spatial conservation prioritization: Quantitative methods and computational tools* (pp. 1–15). Oxford, UK: Oxford University Press.

Fox, E., Miller-Henson, M., Ugoretz, J., Weber, M., Gleason, M., Kirlin, J., … Mastrup, S. (2013). Enabling conditions to support marine protected area network planning: California's Marine Life Protection Act Initiative as a case study. *Ocean and Coastal Management*, *74*, 14–23. https://doi.org/10.1016/j.ocecoaman.2012.07.005

Frédou, T., Ferreira, B. P., & Letourneur, Y. (2009). Assessing the stocks of the primary snappers caught in Northeastern Brazilian reef systems. 1: Traditional modelling approaches. *Fisheries Research*, *99*, 90–96. https://doi.org/10.1016/j.fishres.2009.05.008

Froese R., & Pauly, D. (2017). FishBase: World Wide Web electronic publication. Version (01/2017). http://www.fishbase.org./ [17 June 2018].

Giglio, V. J., Pinheiro, H. T., Bender, M. G., Bonaldo, R. M., Costa-Lotufo, L. V., Ferreira, C. E. L., … Francini-Filho, R. B. (2018). Large and remote marine protected areas in the South Atlantic Ocean are flawed and raise concerns: Comments on Soares and Lucas (2018). *Marine Policy*, *96*, 13–17. https://doi.org/10.1016/j.marpol.2018.07.017

Graham, N. A. J., & McClanahan, T. R. (2013). The last call for marine wilderness? *Bioscience*, *63*, 397–402. https://doi.org/10.1525/bio.2013.63.5.13

Halpern, B. S., Frazier, M., Potapenko, J., Casey, K. S., Koenig, K., Longo, C., … Walbridge, S. (2015). Spatial and temporal changes in cumulative human impacts on the world's ocean. *Nature Communications*, *6*, 7615. https://doi.org/10.1038/ncomms8615

Halpern, B. S., Klein, C. J., Brown, C. J., Beger, M., Grantham, H. S., Mangubhai, S., … Possingham, H. P. (2013). Achieving the triple bottom line in the face of inherent trade-offs among social equity, economic return, and conservation. *Proceedings of the National Academy of Sciences of the United States of America*, *110*, 6229–6234. https://doi.org/10.1073/pnas.1217689110

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*, 948–952. https://doi.org/10.1126/science.1149345

Hamel, M. A., Pressey, R. L., Evans, L. S., & Andréfouët, S. (2018). The importance of fishing grounds as perceived by local communities can be undervalued by measures of socioeconomic cost used in conservation planning. *Conservation Letters*, *11*, 1–9. https://doi.org/10.1111/conl.12352

Instituto Brasileiro de Geografia e Estatística. (2011). *Atlas geográfico das zonas costeiras e oceânicas do Brasil*. Rio de Janeiro, Brazil: Instituto Brasileiro de Geografia e Estatística.

Jenkins, C. N., & Van Houtan, K. S. (2017). Mathematical efficiency does not necessarily mean better. *Biological Conservation*, *210*, 351. https://doi.org/10.1016/j.biocon.2017.02.039

Juhel, J. -B., Vigliola, L., Wantiez, L., Letessier, T. B., Meeuwing, J., & Mouillot, D. (2019). Isolation and no-entry marine reserves mitigate anthropogenic impacts on grey reef shark behavior. *Scientific Reports*, *9*, 2897. https://doi.org/10.1038/s41598-018-37145-x

Jumin, R., Binson, A., McGowan, J., Magupin, S., Beger, M., Brown, C., … Klein, C. (2017). From Marxan to management: Ocean zoning with stakeholders for Tun Mustapha Park in Sabah, Malaysia. *Oryx*, 1–12. https://doi.org/10.1017/S0030605316001514

Klein, C. J., Steinback, C., Scholz, A. J., & Possingham, H. P. (2008). Effectiveness of marine reserve networks in representing biodiversity and minimizing impact to fishermen: A comparison of two approaches used in California. *Conservation Letters*, *1*, 44–51. https://doi.org/10.1111/j.1755-263X.2008.00005.x

Knight, A. T., Cowling, R. M., Boshoff, A. F., Wilson, S. L., & Pierce, S. M. (2011). Walking in STEP: Lessons for linking spatial prioritisations to implementation strategies. *Biological Conservation*, *144*, 202–211. https://doi.org/10.1016/j.biocon.2010.08.017

Knight, A. T., Cowling, R. M., Possingham, H. P., & Wilson, K. A. (2009). From theory to practice: Designing and situating spatial prioritization approaches to better implement conservation action. In A. Moilanen, K. A. Wilson, & H. P. Possingham (Eds.), *Spatial conservation prioritization: Quantitative methods and computational tools* (pp. 249–259). Oxford, UK: Oxford University Press.

Knight, A. T., Cowling, R. M., Rouget, M., Balmford, A., Lombard, A. T., & Campbell, B. M. (2008). Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. *Conservation Biology*, *22*, 610–617. https://doi.org/10.1111/j.1523-1739.2008.00914.x

Knoppers, B., Ekau, W., & Figueiredo, A. G. (1999). The coast and shelf of east and northeast Brazil and material transport. *Geo-Marine Letters*, *19*, 171–178. https://doi.org/10.1007/s003670050106

Knoppers, B., Ekau, W., Figueiredo Júnior, A. G., & Soares-Gomes, A. (2002). Zona costeira e plataforma continental do Brasil. In R. C. Pereira, & A. Soares-Gomes (Eds.), *Biologia marinha* (pp. 353–361). Rio de Janeiro, Brazil: Editora Interciência.

Kroodsma, D. A., Mayorga, J., Hochberg, T., Miller, N. A., Boerder, K., Ferretti, F., … Worm, B. (2018). Tracking the global footprint of fisheries. *Science*, *359*, 904–908. https://doi.org/10.1126/science.aao5646

Krüger, L., Ramos, J. A., Xavier, J. C., Grémillet, D., González-Solís, J., Kolbeinsson, Y., … Paiva, V. H. (2017). Identification of candidate pelagic marine protected areas through a seabird seasonal, multispecific and extinction risk-based approach. *Animal Conservation*, *20*, 409–424. https://doi.org/10.1111/acv.12339

Leão, Z. M. A. N., & Kikuchi, R. K. P. (2001). The Abrolhos reefs of Brazil. In U. Seeliger, & B. Kjerfve (Eds.), *Coastal marine ecosystems of Latin America* (pp. 83–96). Berlin, Germany: Springer-Verlag. https://doi.org/10.1007/978-3-662-04482-7_7

Lehtomäki, J., & Moilanen, A. (2013). Methods and workflow for spatial conservation prioritization using Zonation. *Environmental Modelling & Software*, *47*, 128–137. https://doi.org/10.1016/j.envsoft.2013.05.001

Letessier, T. B., Mouillot, D., Bouchet, P. J., Vigliola, L., Fernandes, M. C., Thompson, C., … Meeuwig, J. J. (2019). Remote reefs and seamounts are the last refuges for marine predators across the Indo-Pacific. *PLoS Biology*, *17*, e3000366. https://doi.org/10.1371/journal.pbio.3000366

Li, Y., Zhang, C., Xue, Y., Xu, B., Sun, M., Ren, Y., & Chen, Y. (2019). Developing a marine protected area network with multiple objectives in China. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *29*, 952–963. https://doi.org/10.1002/aqc.3076

Luiz, O. J., & Edwards, A. J. (2011). Extinction of a shark population in the Archipelago of Saint Paul's Rocks (equatorial Atlantic) inferred from the historical record. *Biological Conservation*, *144*, 2873–2881. https://doi.org/10.1016/j.biocon.2011.08.004

Magris, R. A., & Pressey, R. L. (2018). Marine protected areas: Just for show? *Science*, *360*, 723–724. https://doi.org/10.1126/science.aat6215

Magris, R. A., Pressey, R. L., Weeks, R., & Ban, N. C. (2014). Integrating connectivity and climate change into marine conservation planning. *Biological Conservation*, *170*, 207–221. https://doi.org/10.1016/j.biocon.2013.12.032

Marcovaldi, M. A. A. G. D., Santos, A. S., & Sales, G. (2011). *Plano de ação nacional para a conservação das tartarugas marinhas*. Brasília, Brazil: Instituto Chico Mendes de Conservação da Biodiversidade.

Mazor, T., Giakoumi, S., Kark, S., & Possingham, H. P. (2014). Large-scale conservation planning in a multinational marine environment: Cost matters. *Ecological Applications*, *24*, 1115–1130. https://doi.org/10.1890/13-1249.1

Mittermeier, R., Baião, P. C., Barrera, L., Buppert, T., McCullough, J., Langrand, O., … Scarano, F. R. (2010). O protagonismo do Brasil no histórico acordo global de proteção à biodiversidade. *Natureza & Conservação*, *8*, 197–200. https://doi.org/10.4322/natcon.00802017

Ministério do Meio Ambiente. (2007). *Áreas prioritárias para a conservação, uso sustentável e repartição de benefícios da biodiversidade Brasileira: Atualização—Portaria MMA no 9, de 23 de janeiro de 2007*. Brasília, Brazil: Ministério do Meio Ambiente, Secretaria de Biodiversidade e Florestas.

Ministério do Meio Ambiente. (2008). *Macrodiagnóstico da zona costeira e marinha do Brasil*. Brasília, Brazil: Ministério do Meio Ambiente.

Ministério do Meio Ambiente. (2014a). Portaria no 43, de 31 de janeiro de 2014. *Diário Oficial da União—Seção 1*, *25*, 53–54.

Ministério do Meio Ambiente. (2014b). Portaria no 444, de 17 de dezembro de 2014. Lista Nacional Oficial de espécies da fauna ameaçadas de extinção. *Diário Oficial da União–Seção 1*, *245*, 121–126.

Ministério do Meio Ambiente. (2014c). Portaria no 445, de 17 de dezembro de 2014. Lista Nacional Oficial de espécies da fauna ameaçadas de extinção—Peixes e invertebrados aquáticos. *Diário Oficial da União—Seção 1*, *245*, 126–130.

Moilanen, A., Anderson, B. J., Eigenbrod, F., Heinemeyer, A., Roy, D. B., Gillings, S., … Thomas, C. D. (2011). Balancing alternative land uses in conservation prioritization. *Ecological Applications*, *21*, 1419–1426. https://doi.org/10.1890/10-1865.1

Moilanen, A., Franco, A. M. A., Early, R. I., Fox, R., Wintle, B., & Thomas, C. D. (2005). Prioritizing multiple-use landscape for conservation: Methods for large multispecies planning problems. *Proceedings of the Royal Society B*, *272*, 1885–1891. https://doi.org/10.1098/rspb.2005.3164

Mora, C., Tittensor, D. P., Adl, S., Simpson, A. G. B., & Worm, B. (2011). How many species are there on Earth and in the ocean? *PLoS Biology*, *9*, e1001127. https://doi.org/10.1371/journal.pbio.1001127

Ministério da Pesca e Aquicultura. (2012). *Boletim estatístico da pesca e aquicultura—Brasil 2010*. Brasília, Brazil: Ministério da Pesca e Aquicultura.

Naidoo, R., Balmford, A., Ferraro, P. J., Polasky, S., Ricketts, T. H., & Rouget, M. (2006). Integrating economic costs into conservation planning. *Trends in Ecology & Evolution*, *21*, 681–687. https://doi.org/10.1016/j.tree.2006.10.003

Pascelli, C., Riul, P., Riosmena-Rodríguez, R., Scherner, F., Nunes, M., Hall-Spencer, J. M., … Horta, P. (2013). Seasonal and depth-driven changes in rhodolith bed structure and associated macroalgae off Arvoredo island (southeastern Brazil). *Aquatic Botany*, *111*, 62–65. https://doi.org/10.1016/j.aquabot.2013.05.009

Pereira-Filho, G. H., Shintate, G. S. I., Kitahara, M. V., Moura, R. L., Amado-Filho, G. M., Bahia, R. G., … Motta, F. S. (2019). The southernmost Atlantic coral reef is off the subtropical island of Queimada Grande (24°S), Brazil. *Bulletin of Marine Science*, *95*, 277–287. https://doi.org/10.5343/bms.2018.0056

Pinheiro, H. T., Rocha, L. A., Macieira, R. M., Carvalho-Filho, A., Anderson, A. B., Bender, M. G., … Floeter, S. R. (2018). South-western Atlantic reef fishes: Zoogeographical patterns and ecological drivers reveal a secondary biodiversity centre in the Atlantic Ocean. *Diversity and Distributions*, *24*, 951–965. https://doi.org/10.1111/ddi.12729

Pinheiro, H. T., Teixeira, J. B., Francini-Filho, R. B., Soares-Gomes, A., Ferreira, C. E. L., & Rocha, L. A. (2019). Hope and doubt for the world's marine ecosystems. *Perspectives in Ecology and Conservation*, *17*, 19–25. https://doi.org/10.1016/j.pecon.2018.11.001

Pressey, R. L., Visconti, P., & Ferraro, P. J. (2015). Making parks make a difference: Poor alignment of policy, planning and management with protected-area impact, and ways forward. *Proceedings of the Royal Society B*, *370*, 20140280. https://doi.org/10.1098/rstb.2014.0280

Pressey, R. L., Weeks, R., & Gurney, G. G. (2017). From displacement activities to evidence-informed decisions in conservation. *Biological Conservation*, *212*, 337–348. https://doi.org/10.1016/j.biocon.2017.06.009

Rees, S. E., Foster, N. L., Langmead, O., Pittman, S., & Johnson, D. E. (2018). Defining the qualitative elements of Aichi Biodiversity Target 11 with regard to the marine and coastal environment in order to strengthen global efforts for marine biodiversity conservation outlined in the United Nations Sustainable Development Goal 14. *Marine Policy*, *93*, 241–250. https://doi.org/10.1016/j.marpol.2017.05.016

Roberts, C. M. (2019). *30×30: A blueprint for ocean protection—Executive summary*. London, UK: Greenpeace.

Sala, E., Lubchenco, J., Grorud-Colvert, K., Novelli, C., Roberts, C., & Sumaila, U. R. (2018). Assessing real progress towards effective ocean protection. *Marine Policy*, *91*, 11–13. https://doi.org/10.1016/j.marpol.2018.02.004

Soares, M. L. G., Estrada, G. C. D., Fernandez, V., & Tognella, M. M. P. (2012). Southern limit of the western South Atlantic mangroves: Assessment of the potential effects of global warming from a biogeographical perspective. *Estuarine, Coastal and Shelf Science*, *101*, 44–53. https://doi.org/10.1016/j.ecss.2012.02.018

Souza Filho, P. W. M. (2005). Costa de manguezais de macromaré da Amazônia: Cenários morfológicos, mapeamento e quantificação de áreas usando dados de sensores remotos. *Revista Brasileira de Geofísica*, *23*, 427–435. https://doi.org/10.1590/S0102-261X2005000400006

Spalding, M. D., Fox, H. E., Allen, G. R., Davidson, N. C., Ferdaña, Z. A., Finlayson, M., … Robertson, J. (2007). Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. *Bioscience*, *57*, 573–583. https://doi.org/10.1641/B570707

Spalding, M. D., Meliane, I., Bennett, N. J., Dearden, P., Patil, P. G., & Brumbaugh, R. D. (2016). Building towards the marine conservation end-game: Consolidating the role of MPAs in a future ocean. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *26* (Suppl. 2), 185–199. https://doi.org/10.1002/aqc.2686.

Speed, C. W., Cappo, M., & Meekan, M. G. (2018). Evidence for rapid recovery of shark populations within a coral reef marine protected area. *Biological Conservation*, *220*, 308–319. https://doi.org/10.1016/j.biocon.2018.01.010

Swartz, W., Sala, E., Tracey, S., Watson, R., & Pauly, D. (2010). The spatial expansion and ecological footprint of fisheries (1950 to present). *PLoS ONE*, *5*, e15143. https://doi.org/10.1371/journal.pone.0015143

United Nations & Food and Agriculture Organization of the United Nations. (2017). *Commodities and development report 2017: Commodity markets, economic growth and development*. New York, NY: United Nations & Food and Agriculture Organization of the United Nations.

UN Environment Programme World Conservation Monitoring Centre & International Union for Conservation of Nature. (2018). Marine protected planet. http://www.protectedplanet.net/ [10 December 2018].

Vasconcellos, M., Diegues, A. C., & Sales, R. R. (2007). Limites e possibilidades na gestão da pesca artesanal costeira. In Costa (Ed.), *Nas redes da pesca artesanal* (pp. 16–83). Brasília, Brazil: Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis.

Venter, O., Fuller, R. A., Segan, D. B., Carwardine, J., Brooks, T., Butchart, S. H. M., … Watson, J. E. M. (2014). Targeting global protected area expansion for imperiled biodiversity. *PLoS Biology*, *12*, e1001891. https://doi.org/10.1371/journal.pbio.1001891

Vilar, C. C., Joyeux, J. C., Loyola, R., & Spach, H. L. (2015). Setting priorities for the conservation of marine vertebrates in Brazilian waters. *Ocean and Coastal Management*, *107*, 28–36. https://doi.org/10.1016/j.ocecoaman.2015.01.018

Vilar, C. C., Joyeux, J. C., & Spach, H. L. (2017). Geographic variation in species richness, rarity, and the selection of areas for conservation: An integrative approach with Brazilian estuarine fishes. *Estuarine, Coastal and Shelf Science*, *196*, 134–140. https://doi.org/10.1016/j.ecss.2017.06.022

White, T. D., Carlisle, A. B., Kroodsma, D. A., Block, B. A., Casagrandi, R., De Leo, G. A., … McCauley, D. J. (2017). Assessing the effectiveness of a large marine protected area for reef shark conservation. *Biological Conservation*, *207*, 64–71. https://doi.org/10.1016/j.biocon.2017.01.009

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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APPENDIX A

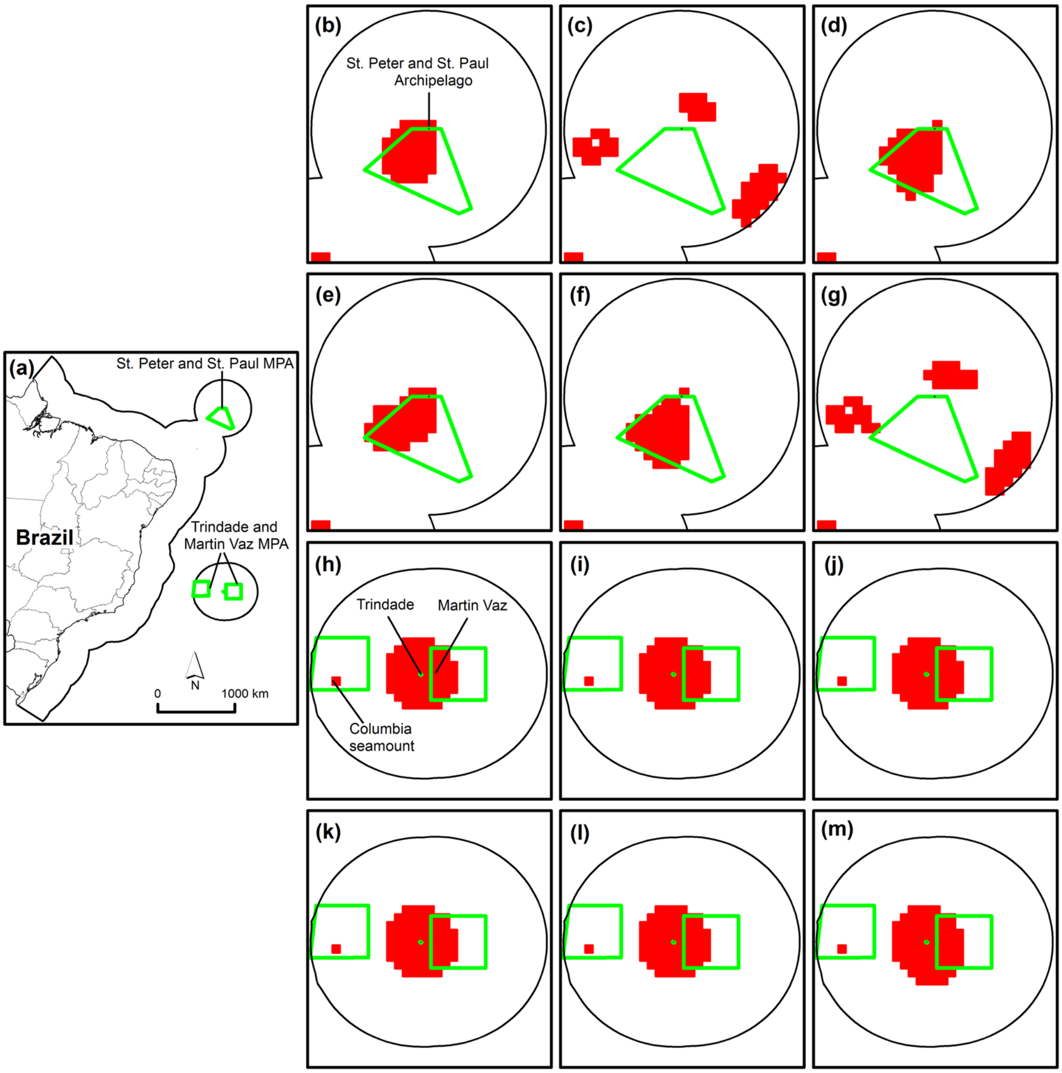

FIGURE A1 (a) Location of the two no-take marine protected areas (MPAs) created in March 2018 in Brazilian jurisdictional waters (green polygons), and overlap of our priority areas for conservation (top 10%, shown in red) under each prioritization scenario with (b–g) the St Peter and St Paul and (h–m) Trindade and Martin Vaz MPAs. Conservation prioritization scenarios: (b, h) unconstrained, (c, i) large-scale fishing, (d, j) small-scale fishing, (e, k) oil and gas, (f, l) mining, and (g, m) all activities. Note that most of the St Peter and St Paul Archipelago and adjacent waters are not within no-take areas and that most shallow habitats around Trindade Island are not within any MPA (for details, see Giglio et al., 2018)