REGULAR ARTICLE

Interdecadal changes in ichthyofauna in a tropical bay with high anthropogenic influences: functional stability despite turnover predominance

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Abstract

Functional characteristics of species are of great importance for understanding their roles in ecosystems and can be used to detect long-term chances in the environment. We evaluated temporal changes (1983–1985 and 2017–2019) in taxonomic and functional indices of the fish fauna in shallow areas of a tropical bay heavily impacted by anthropogenic activities in recent decades. The hypothesis that functional indices change over time as a result of environmental degradation was tested. Our results showed a significant decrease in species richness and abundance over time, and in functional richness, while others functional diversity indices (divergency, evenness, and originality) remained stable. Thirteen functional groups were detected, some of which contained only one species, raising concerns about the loss of ecosystem functions due to ongoing changes. We also observed an increase in beta diversity over time, which may be the result of a decrease in local richness without leading to regional extinctions. Turnover was the most important process in structuring the fish fauna at the evaluated time scale. The relative stability of the functional structure and the higher levels of turnover seem to be related to the dominance of functional groups, within which species replace each other according to their responses to environmental filters that select for specific functional traits. Incorporating functional diversity indices and beta diversity variations in the fish community helped to enhance the existing information about this coastal system by offering improved estimates of biological diversity through diverse approaches. The predominance of turnover identified in the preset study suggests a dynamic and fluctuating species composition within the habitat. In this sense, habitat preservation should prioritize the protection of diverse habitats to accommodate a broad spectrum of species.

KEYWORDS
beta diversity, estuarine fish, functional traits, temporal comparisons

1 | INTRODUCTION

Understanding the dynamics of biological communities through the lens of species' functional diversity and ecological patterns is crucial for gaining deeper insights into ecosystem organization and functioning. This becomes especially significant in the present scenario, where human activities pose an escalating threat to biodiversity, often resulting in irreversible consequences. Estuarine zones stand as some of the most productive ecosystems on a global scale, generating a diverse array of goods and services while supporting rich biological

diversity. This can be attributed to their exceptional productivity and abundant food resources, which in turn provide refuge and serve as crucial spawning sites for numerous species (Islam et al., 2006; Macário et al., 2021). Despite their ecological value, these ecosystems are among the most affected due to the rapid expansion of human populations in their surroundings, causing significant impacts on biodiversity (Lotze et al., 2006). Hence, there is an urgent need to evaluate the spatiotemporal patterns of biodiversity, investigating environmental changes and the resilience of biodiversity. Alterations in these ecosystems can disrupt natural flows and result in habitat loss, impeding the movement of organisms between previously interconnected habitats (Barletta & Lima, 2019). These changes manifest more prominently in the long run, enabling a comprehensive understanding of biological communities and offering more robust and informative insights into the distribution and functional significance of species over time (e.g., Gomes-Gonçalves et al., 2020; Törnroos et al., 2019; Villeger et al., 2010).

Gaining an understanding of the patterns observed in various diversity indices and assessing their agreement can provide valuable insights into the relative importance of different processes that shape biological assemblages. This knowledge is essential for evaluating the impacts of global changes on ecosystem functions and services (Violle et al., 2014). Functional diversity indices, which take into account the functional traits of different aspects of each species, have the potential to significantly enhance our understanding of how biodiversity responds to environmental changes (e.g., Brandl et al., 2016; Gomes-Gonçalves et al., 2020; Villeger et al., 2010). The utilization of functional traits to characterize communities offers several advantages. First, it possesses higher predictive potential compared to purely taxonomic approaches that fail to consider the ecological roles played by species in the environment (Flynn et al., 2011; Mouillot et al., 2011). Additionally, it enables the evaluation of species interrelationships, such as competition or membership in an interaction network, and how species respond to similar environmental constraints (Mouillot et al., 2007). Furthermore, these indices are particularly useful when comparing ecosystems, as they can assess similar functions shared by ecosystems without necessarily relying on taxonomic overlap (Mindel et al., 2016).

Beta-diversity is equally important for comparing the changes in species composition across different locations and over time within an ecological system (Heino et al., 2015; Jankowski et al., 2009; Whittaker, 1972). This approach consists of two components: species turnover and nestedness. Species turnover happens when there is a substitution of species, meaning that some species present in one location are not found in another location but are instead replaced by different species that are absent at the first location (Baselga, 2010). Turnover often prevails as a result of species replacement along environmental gradients (Peláez et al., 2017). On the other hand, nestedness occurs when certain species are absent in one location compared to another, without being replaced by different species (Baselga, 2010). Therefore, customizing conservation efforts to the unique ecological context and the drivers of turnover and nestedness in an ecosystem is crucial. An adaptive, holistic approach, accounting for the dynamic environment, is vital for effective conservation.

Sepetiba Bay is a coastal system located in the southern part of the state of Rio de Janeiro. Despite its high ecological value, the bay has faced significant anthropic activities in recent decades, resulting in substantial habitat transformations. These activities involve the destruction of coastline habitats and the conversion of the area for cultivation, industrialization, and human settlement. Consequently, a significant amount of organic and industrial effluents is released into the bay, leading to increased pollution and habitat degradation (Cunha et al., 2009; Magalhães et al., 2003; Molisani et al., 2006). These activities have a detrimental effect on the local fish fauna (Araujo et al., 2016; Araújo et al., 2017). Monitoring of the fish fauna in Sepetiba Bay has been conducted for several decades, generating a wealth of biological and environmental data spanning over 30 years. In a study by Araujo et al. (2016), temporal changes in the fish fauna of the bay's deep areas were evaluated, revealing a notable decline in fish richness and abundance. Most of these changes were observed in the inner zone, as it is closer to the sources of impact. Furthermore, Gomes-Gonçalves et al. (2020) conducted a temporal assessment (1993–1995 and 2012–2015) of the fish fauna in the deep areas of the bay. Their findings revealed a significant decrease in functional originality, which is particularly concerning because the loss of species with unique characteristics also signifies the loss of their ecological functions. This loss can have detrimental effects on the ecosystem, leading to a series of damage.

The fish fauna in the shallow zones, located in close proximity to the impact areas, has also shown temporal changes in taxonomic composition (Araujo et al., 2016). Considering previous studies indicating a decline in fish fauna richness, we sought to test the hypothesis that functional diversity indices would also be affected over time due to environmental degradation. To investigate this, we examined the variation in beta diversity and functional indices during two distinct periods (1983–1985 and 2017–2019), aiming to address two key questions: (1) Have there been changes in the functional diversity indices of the fish fauna between the two evaluated periods? (2) Based on the beta diversity indices, is there a prevalence of species turnover or are there indications of species loss (higher nestedness)? Our objective was to identify biological patterns that can inform decision-making regarding the prioritization of habitats for the conservation of fish fauna, thereby aiding efforts to prevent or minimize adverse effects on biodiversity in ecosystems.

2 | MATERIALS AND METHODS

2.1 | Study area

Sepetiba Bay (Figure 1) is a semi-open coastal area covering approximately 450 km^2 located in the southern region of Rio de Janeiro State. It lies between latitudes 22°54'S and 23°04'S, and longitudes 43°34'W and 44°10'W. This bay is partially open to the ocean, with a board opening at its western end and a narrow channel to the east. To the south, it is demarcated by the extensive Marambaia sandbank, which stretches for a length of 40 km. The Marambaia sandbank
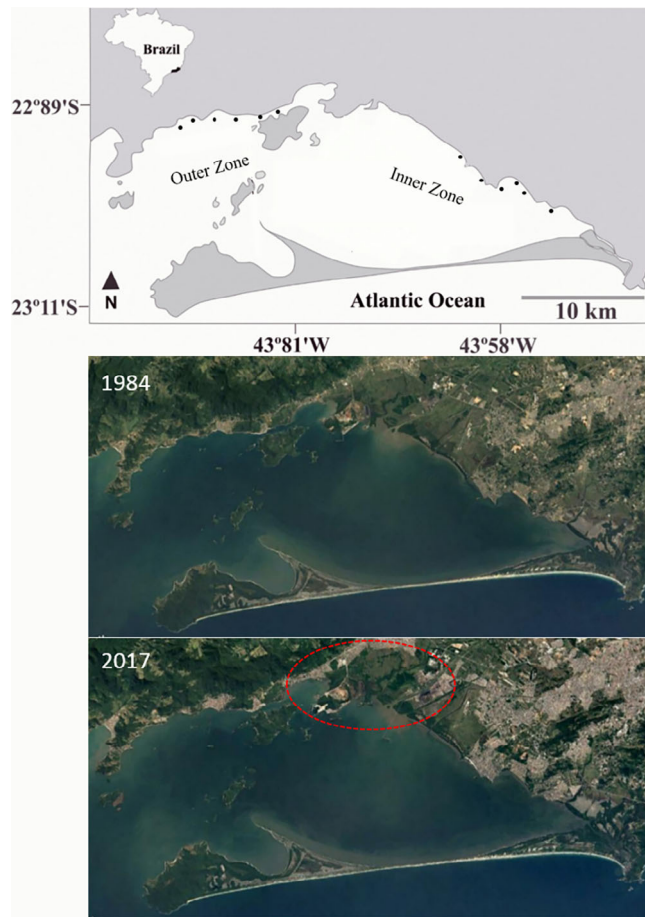
FIGURE 1 Study area Sepetiba Bay, with the two sampling zones (inner and outer) marked. The expansion of land use due to anthropogenic activities and urbanization (represented by gray areas) can be observed in the two maps depicting the years 1984 and 2017. The main industrial areas are located within the region outlined by the red ellipse and the black dots represent the sampled locations. Google Earth (2021).

serves as a natural barrier, partitioning the waters of Sepetiba Bay from the Atlantic Ocean (Fundação Rio-Águas, 2020). Several rivers, streams, and drainage channels flow into the bay, contributing to the dilution of the waters and carrying effluents from residential and industrial areas on the mainland. The average water temperature in the bay ranges from 21.5°C during the winter to 26.5°C in the summer. The salinity ranges from 29 in the inner bay to 33 in the outer bay. This microtidal system exhibits a tidal range of approximately 1 m (Mahiques et al., 2010). The prevailing northeasterly and southwesterly winds induce thermal currents that facilitate the exchange of water between the bay and the ocean.

In recent decades, Sepetiba Bay has undergone substantial transformations. Traditional economic activities like agriculture and fishing have been supplanted by industrial expansion. These human-induced developments in the bay's vicinity have led to environmental deterioration, with pollutants being discharged directly into the bay or indirectly via rivers and drainage channels, leading to water pollution (Pellegatti et al., 2001). Moreover, human activities have fueled

unchecked urbanization, resulting in a surge in population densities around the bay. In 1980, the population stood at around 60,000, but by 2000, it had surged to an estimated 2 million people (Leal Neto et al., 2006).

Since the 1970s, there has been a rapid industrial expansion in the area, with the establishment and operation of chemical and metallurgical industries in the bay. This expansion, in turn, has inadvertently encouraged various other potentially detrimental activities, including overfishing and increased construction (Barcellos & Lacerda, 1994; Molisani et al., 2004; Cunha et al., 2006). The commencement of operations at the Port of Sepetiba in 1982 triggered substantial economic and industrial growth in the region (Clarke et al., 2004). Alongside the visible transformations along the shores of Sepetiba Bay, these developments also brought about multiple sources of pollution, resulting in instances of chemical contamination, including cadmium (Cd) and zinc (Zn). In addition, more recent activities have also had a significant impact. These include the expansion of the port through the dredging of the access channel to a depth of 20 m, allowing for the accommodation of larger vessels (Azevedo et al., 2007). Furthermore, the implementation of the Submarine Development Program (PROSUB) was introduced with the goal of strengthening the national defense infrastructure by constructing conventional submarines and one nuclear-powered submarine. This program encompasses the construction of a metal structure manufacturing unit, two shipyards, a radiological complex, and a naval base. Such extensive urban and industrial expansion has been associated with the degradation of this coastal area. This development has led to a significant loss of approximately 26% of mangrove areas over three decades, resulting in alterations in the local environment that have impacted the resident fauna (Araújo et al., 2017).

2.2 | Database

For this study, we utilized a database compiled from fish samplings conducted with the same equipment and methodological procedures in shallow areas during two distinct periods of 1983–1985 and 2017–2019. The study was carried out in two zones of Sepetiba Bay: the inner zone and the outer zone.

The inner zone is located in the innermost part of the bay and includes mangrove formations, along with several rivers and drainage channels that carry industrial and urban effluents into the bay (Figure 1). In this zone, discharges from small perennial rivers play a crucial role, leading to increased water turbidity and temperature, and decreased salinity (Leal Neto et al., 2006). Additionally, it is the zone mostly heavily affected by the presence of industrial plants along the shoreline (Barcellos et al., 1997; Borges & Nittrouer, 2016; Gonçalves et al., 2020). On the other hand, the outer zone is located near to the bay's wide connection with the ocean and has comparatively lower levels of anthropogenic activities, leading to more consistent and stable environmental conditions.

Our sampling efforts encompassed both bay zones (inner and outer) during two seasons (summer and winter) and across two

separated time periods (1983–1985 and 2017–2019). This study primarily assesses the impact of human-induced changes on the ichthyofauna between these two periods. Given the negligible seasonal effects (Araujo et al., 2016; Camara et al., 2019), data were collected in both seasons solely to ensure a thorough evaluation of these two periods. This comprehensive approach resulted in a total of 144 samples, involving six sites, three replicates, two seasons, two zones, and two periods.

2.3 | Fish sampling

The fish were collected using a beach seine net (12 × 2.5 m, 5-mm mesh size), equipped with 30 m ropes attached to each side of the net to facilitate hauling. The net was dragged perpendicular to the coastline from a depth of approximately 1.5 m. The hauls were performed by two people, one at each end of the rope, covering an area of approximately 300 m^2.

Immediately after collection, the fish were anesthetized with benzocaine hydrochloride (50 mg L^{-1}), then fixed in 10% formaldehyde–seawater solution and, after 48 hours, preserved in 70% ethanol. The fish were identified at the lowest possible taxonomic level, and vouchers were deposited in the Ichthyological Collection of the Fish Ecology Laboratory at the Universidade Federal Rural do Rio de Janeiro. In the most recent study period, the care of fishes was conducted in strict compliance with the ICMBio (Brazilian Environmental Agency) animal welfare laws, as outlined in the Brazilian Federal Law 11.794 of 8 October 2008. Additionally, all procedures followed the guidelines and policies approved by the Ethics Council of Animal Use (CEUA/ICBS/UFRRJ) under permission number 12179.

2.4 | Data analyses

The study involved a comparison of the fish abundance, measured by the number of individuals and the species richness per 100 m^2 (sampling unit) across different zones and periods. To accomplish this, a permutational multivariate analysis of variance (PERMANOVA) was employed, with 999 permutations used to calculate p values. In this analysis, fish abundance and richness were used as the response variables, while periods (1983–1985, 2017–2019) and zones (nested within periods) were treated as fixed factors. Paired comparisons using PERMANOVA were conducted to examine the variations between periods and zones. Prior to conducting these analyses, the fish richness and abundance were subjected to a square-root transformation. The transformation of biological data aimed to mitigate the impact of the most abundant species while retaining information about their relative abundance.

Temporal changes in fish community composition were assessed through the application of beta-diversity, comparing the two analyzed periods. This approach involved the calculation of overall dissimilarity (Sørensen dissimilarity, βsor), as well as turnover (βsim) and nestedness (βsne) components of variation (Baselga, 2010). Furthermore, the

spatial dispersion patterns of each sample were evaluated using multivariate dispersion permutation analysis (PERMDISP) using a distance matrix based on the Sørensen dissimilarity (presence and absence) for fish data. This method estimates the average distance between samples and the group centroid in a multivariate space, serving as an indicator of spatial beta diversity (Anderson et al., 2006). The PERMDISP results can be interpreted as a measure of taxonomic beta diversity, providing insights into the variability in species composition at a specific spatial scale (Anderson et al., 2006). For these statistical analyses, the software packages PRIMER 6 + PERMANOVA and the betapart package (Baselga & Orme, 2012) in the R software were utilized to perform the PERMANOVA and PERMDISP analyses, respectively.

To analyze functional diversity, we gathered data on the life history and ecological traits of each species. Specifically, six categories of functional traits were chosen (Table 1, S1, and S2). We complied this information from previously published sources and available data from Fishbase (Froese & Pauly, 2021). The species were categorized based on their body shape, vertical distribution in the water column, mobility, and their reproductive, trophic, and habitat use guilds (Tables 1 and S1). To depict groups of species that shared similar functional characteristics, a principal coordinate analysis was employed on a fish trait matrix.

TABLE 1 Functional traits categories selected for this study, with the traits code.

Trait groups	Category	Code
Vertical distribution (VD)	Pelagic	P
	Benthonic	B
Body shape (BS)	Filiform	FILI
	Elongated	ELON
	Flattened laterally symmetrical	FLS
	Flattened laterally asymmetric	FLASS
	Flattened dorsoventrally	FDV
	Cilindric	CLI
Mobility (MOB)	Mobile	MO
	Sedentary	SE
Feeding guild (FG)	Planctivore	PL
	Detritivore	DE
	Piscivore	PIS
	Benthophague	BEM
	Hyperbenthophague	HYP
	Herbivore	HE
	Opportunistic	0PP
Reproductive guild (RG)	Bearers	BEAR
	Nonguarders	NOGUAR
Habitat use (HU)	Resident	RE
	Marine migrant	MM
	Marine straggler	MS

TABLE 2   Functional diversity measures.

Index	Code	Description	Formula
Functional richness	FRic	Convex-hull volume occupied by species based on the distance among their functional traits. It quantifies the amount of niche space occupied by the species within a community.	$FRic = \Sigma [Dij]$
Functional evenness	FEve	Measures the regularity with which each functional space is occupied by species, accounting for their respective abundances. It quantifies the proportion of the total abundance supported by the species with the most extreme trait values.	$FEve = (\Sigma [Dij])/[2 \times FRic]$
Functional divergence	FDiv	Species deviance from the mean distance to the center of gravity weighted by relative abundance. It quantifies the variance of the species' functions and the position of their clusters in trait space.	$FDiv = \Sigma [Pij \times |Dij|]$
Functional originality	FOri	The weighted mean distance to the nearest species from the global species pool. It quantifies the mean distance in the functional trait space to all other species that occur in a given assemblage. To assess how unique a species is based on its rarity or its proportion in relation to other species in the ecosystem.	$FOri = 1 - (\Sigma[P^2])$

Note: Definitions are aligned with those provided in Villeger et al. (2008) and Mouillot et al. (2013). Dij represents the difference/dissimilarity in trait values between species *i* and *j*; D*ij* represents the difference in trait values between species *i* and *j*; FRic is the functional richness; P*ij* is the relative abundance of species *i* and *j*; D*ij* is the difference in trait values between species *i* and *j*; P represents the proportion (or relative abundance) of the species traits in the ecosystem.

To assess the functional changes within the fish community, four indices of functional diversity were utilized: functional richness (FRic), functional divergence (FDiv), functional evenness (FEve), and functional originality (FOri). FRic quantifies the extent of functional space occupied by species based on their specific functional traits within a community (Table 2). As indicated by Villeger et al. (2008), FRic is influenced by the identity of species, particularly by those species with the most extreme functional traits that define the convex hull. FEve measures the regularity with which each functional space is occupied by species, accounting for their respective abundances (Villeger et al., 2008, 2010). According to Villeger et al. (2008), this index attains higher values when species abundance is evenly distributed or when the spatial distance between species is consistent. FDiv provides insights into the distribution of abundances within the range of functional traits exhibited by species (Villeger et al., 2008). This index tends to have lower values when the most abundant species possess functional traits that are closer to the center of the trait space. Conversely, higher values of FDiv indicate that the most abundant species exhibit extreme functional traits (Mouillot et al., 2013; Villeger et al., 2008). FOri is defined as the average distance between each species and its nearest neighbor within the functional space, illustrating the level of distinctiveness or exclusivity of functional traits within the community (Mouillot et al., 2013).

The functional diversity indices were computed using the "dbFD" function from the FD package within the R environment (R Development Core Team, 2022). To generate graphical representations of these indices, we followed the "R script" example provided by Sébastien Villéger, which can be accessed at http://villeger.sebastien. free.fr. To compare the indices across zones and periods, a PERMANOVA was conducted using Euclidean distance as the dissimilarity measure. This analysis was performed using PRIMER software version 6 (Anderson et al., 2008).

3 | RESULTS

3.1 | Fish composition

This study recorded a total count of 15,082 individuals belonging to 78 fish species (Table S3). Specifically, for the period 1983–1985, there were 11,121 individuals across 68 species, while during 2017–2019, there were 3961 individuals belonging to 38 species. A significant decrease in average fish densities (PERMANOVA: pseudo-$F = 7.81$, $p = 0.002$) and richness (pseudo-$F = 23.86$, $p = 0.001$) was found with comparatively lower values in the more recent period (Table 3). Furthermore, significant differences were observed in fish densities between the nested zones within each period (pseudo-$F = 4.07$, $p = 0.02$). The outer zone exhibited higher species richness per sample, and it was significantly different from the inner zone during the more recent period (pseudo-$F = 3.79$, $p = 0.002$).

The outer zone exhibited a higher number of recorded species, with a total of 64 species (Table S3). Among these, 39 species were exclusively observed during the period 1983–1985, eight species were exclusive to the 2017–2019 period, and 17 species were observed in both periods. On the other hand, the inner zone recorded 56 species, with 32 species exclusive to the 1983–1985 period,

TABLE 3 Means and standard error of the number of species and the number of individuals per sampling (100 m^2) for each period in the two zones of the Sepetiba Bay.

Period	Zone	Number of species	Number of individuals
1983–1985	Inner	6.3 ± 0.55	175 ± 43.28
	Outer	7.8 ± 0.03	287 ± 87.80
2017–2019	Inner	3.1 ± 0.40	94 ± 38.35
	Outer	5.0 ± 0.33	70 ± 17.12
Pseudo-F; p		23.86; 0.001	7.81; 0.002

Note: Pseudo-F and p values for periods comparisons also shown.

10 species exclusive to the 2017–2019 period, and 14 species observed in both periods.

A decrease in densities was detected for the following dominant species: *Anchoa januaria* (Steindachner, 1879) and *Atherinella brasiliensis* (Quoy & Gaimard, 1825), primarily in the inner zone, as well as *Eucinostomus argenteus* Baird & Girard, 1855, and *Micropogonias furnieri* (Desmarest, 1823), primarily in the outer zone (Figure 2). On the other hand, the densities of *Mugil liza* Valenciennes, 1836, and *Genidens genidens* (Cuvier, 1829) increased overtime (Figure 2).

Temporal taxonomic beta-diversity analysis unveiled that species turnover (βsim = 0.64 ± 0.3) played a dominant role in shaping local communities, while the nestedness component (βsne) had a comparatively smaller impact (βsne = 0.14 ± 0.2) (Figure 3). Additionally, the PERMDISP analysis indicated significant differences in beta diversity between the two periods, with lower values recorded for the 1983–1985 period (53.78 ± 1.78), while the 2017–2019 period exhibited a value of 58.4 (±1.23).

3.2 | Functional groups

The principal coordinates analysis (PCoA) conducted on functional traits (Figure 4) revealed that the first axis (PCoA 1) accounted for 37.3% of the total variation, while the second axis (PCoA 2) explained 23.05% of the variation. This analysis allowed us to identify 13 distinct functional groups.

The first functional group (group I) displayed a positive correlation with the first axis and consisted primarily of pelagic species with elongated body, including engraulid species such as *Anchoa januaria*, *Anchoa tricolor* (Spix & Agassiz, 1829), *Lycengraulis grossidens* (Spix & Agassiz, 1829), and *Cetengraulis edentulus* (Cuvier, 1829), as well as carangid species like *Oligoplites saurus* (Bloch & Schneider, 1801) and *Oligoplites palometa* (Cuvier, 1832), along with *Atherinella brasiliensis* and other species (Figure 4).

The second functional group (group II) primarily comprised migratory marine species with elongated body shapes and benthic feeding habits. This group included detritivorous species from the Mugilidae family, as well as benthic-feeding species such as the sciaenids *Menticirrhus littoralis* (Holbrook, 1847), *Menticirrhus americanus* (Linnaeus, 1758), *Micropogonias furnieri* (Desmarest, 1823), *Umbrina coroides* Cuvier, 1830, *Cynoscion leiarchus* (Cuvier, 1830), among others.

Group III comprised pelagic species with deep bodies, high mobility, and laterally flattened and symmetric body shape. This group included the marine stragglers carangid species such as *Trachinotus carolinus* (Linnaeus, 1766), *Trachinotus falcatus* (Linnaeus, 1758), *Trachinotus goodei* Jordan & Evermann, 1896, *Selene vomer* (Linnaeus, 1758), and *Chloroscombrus chrysurus* (Linnaeus, 1766), as well as the Atlantic spadefish *Chaetodipterus faber* (Broussonet, 1782) and the clupeid *Brevoortia aurea* (Spix & Agassiz, 1829).

Group IV comprised benthic species with a close association to the substrate, exhibiting low mobility and a laterally flattened, asymmetric body shape. This group included flatfishes *Citharichthys spilopterus* Gunther, 1862 and *Symphurus plagusia* (Bloch & Schneider, 1801), as well as *Citharichthys cornutus* (Günther, 1880), *Citharichthys arenaceus* Evermann & Marsh, 1900, *Etropus longimanus* Norman, 1933, and *Etropus crossotus* Jordan & Gilbert, 1882.

Group V consisted of the Ariidae family, which encompassed species such as the marine catfishes *Genidens genidens*, *Genidens barbus* (Lacepède, 1803), *Aspistor luniscutis* (Valenciennes, 1840), and *Cathorops spixii* (Agassiz, 1829), and the bluewing searobin *Prionotus punctatus* (Bloch, 1793). These species have dorsoventrally flattened bodies and are omnivorous benthic fish.

Group VI was composed of resident species with dorsoventrally flattened bodies, including gobies such as *Gobionellus stomatus* Starks, 1913, *Gobionellus oceanicus* (Pallas, 1770), *Evorthodus lyricus* (Girard, 1858), *Bathygobius soporator* (Valenciennes, 1837), and *Ctenogobius boleosoma* (Jordan & Gilbert, 1882), and the bigeye stargazer *Dactyloscopus crossotus* Starks, 1913.

Group VII was composed of mobile benthic species, characterized by elongated bodies and feed preferably on the hyperbenthos. This group included species such as the sciaenids *Odontoscion dentex* (Cuvier, 1830) and *Bardiella ronchus* (Cuvier, 1830), the grunts *Haemulon steindachneri* (Jordan & Gilbert, 1882) and *Conodon nobilis* (Linnaeus, 1758), and the threadfin *Polydactylus oligodon* (Günther, 1860).

Group VIII consisted of resident benthic species that dwell inner bay zones, with low mobility and a laterally flattened asymmetric body shape. This group included soles such as *Achirus declivis* (Linnaeus, 1758) and *Trinectes microphthalmus* (Chabanaud, 1928), while Group IX was characterized by resident flattened dorsoventrally round pufferfish *Sphoeroides greeleyi* Gilbert, 1900 *and Sphoeroides testudineus* (Linnaeus, 1758).
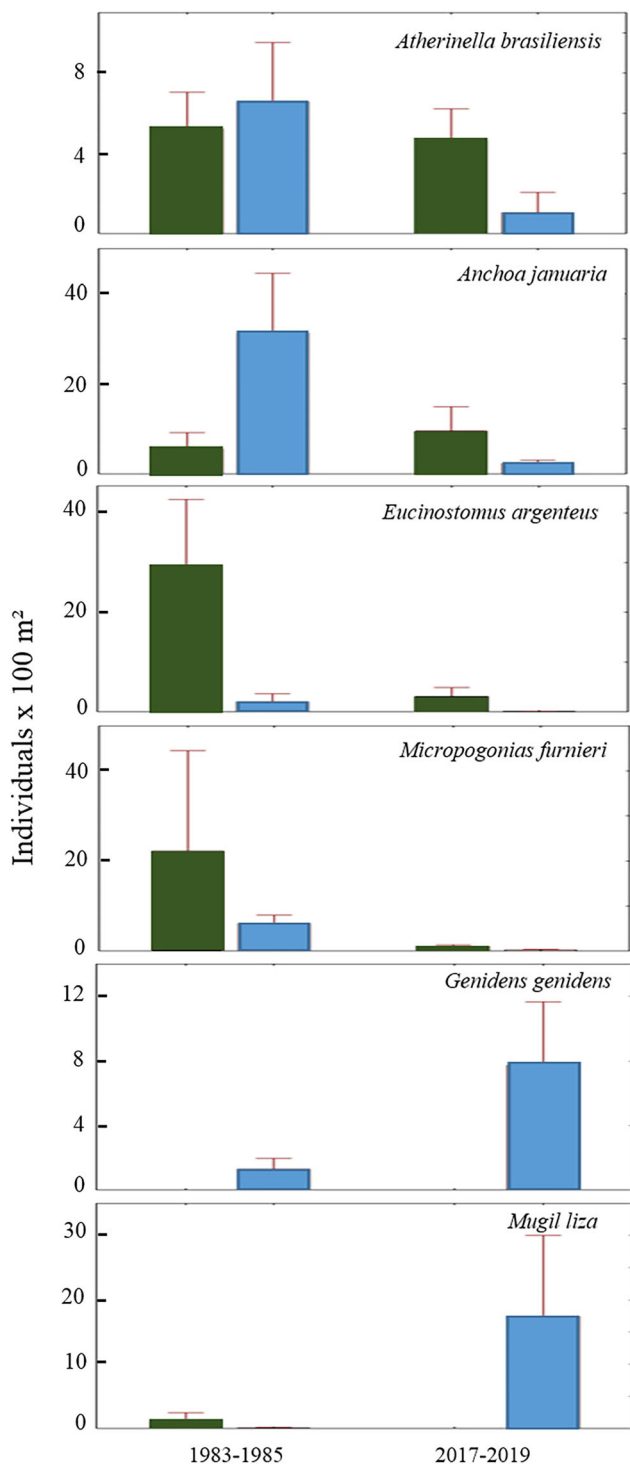
FIGURE 2 Means + 1 standard error (whiskers) of the density (individuals per 100 square meter) for selected abundant fish species in the inner (blue bars) and the outer (green bars) zones of Sepetiba Bay in the two time periods (1983–1985 and 2017–2019).

Group X was composed of pelagic elongated needlefish species, including *Strongylura timucu* (Walbaum, 1792) and *Strongylura marina* (Walbaum, 1792), as well as halfbeak species *Hyporhamphus unifaciatus* (Ranzani, 1841) and *Hyporhamphus brasiliensis* (Linnaeus, 1758). Furthermore, four groups were represented by single elongated
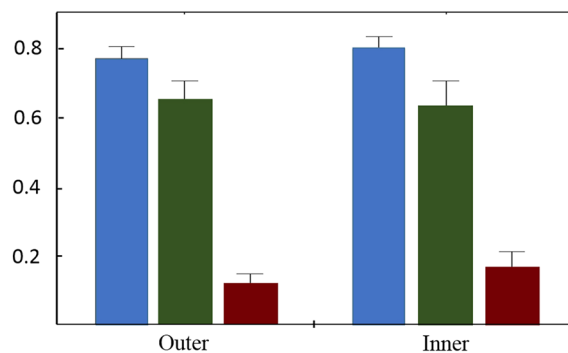
FIGURE 3 Means + 1 standard error (whiskers) for the beta-diversity in the two time periods (1983–1985 and 2017–2019). Overall dissimilarity (Sørensen dissimilarity, β_{sor}, blue bars), and its turnover (β_{sim}, green bars) and nestedness (β_{sne}, red bars) components.
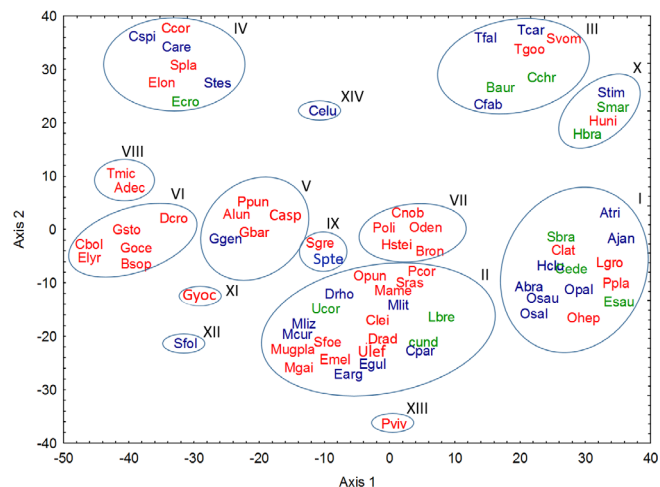

FIGURE 4 The functional groups depicted by the principal coordinates analysis on the functional traits. Species in blue are observed in both periods, those in red were only present in 1983–1985, and those in green were only observed in 2017–2019. The species codes are indicated in Table S2.

species: XI (*Gymnothorax ocellatus* Agassiz, 1831), a less mobile resident species; XII (*Syngnathus folletti* Herald, 1942), a cylindrical fish with paternal reproductive care (bearing the eggs); XIII (*Poecilia vivipara* Bloch & Schneider, 1801), an elongated species with paternal reproductive care (bearing the eggs); and XIV (*Cosmocampus elucens* [Poey, 1868]), a marine straggler species with reproductive care (bearing the eggs).

3.3 | Functional diversity

A significant decline in species' functional richness was observed, with higher values recorded during the 1983–1985 period (PERMANOVA: pseudo-$F = 23.86$, $p = 0.001$) (Figure 5, Table 4). Additionally, a reduction in the filled functional volume by the fish community was

noted, as represented by the blue area of the convex hull in Figure 5, particularly in the more recent period. Furthermore, the outer zone exhibited a larger functional volume in comparison to the inner zone during the first period (Figure 5). The area of the global convex hull, indicated by the gray contour, suggested a lower loss of species (and associated functions) in the first evaluated period, particularly noticeable in the outer zone (Figure 5).

Regarding other functional diversity indices, no significant changes were identified between the periods, indicating that despite species turnover, the functional structure of the fish community appears to remain stable, except for functional richness (Table 4).
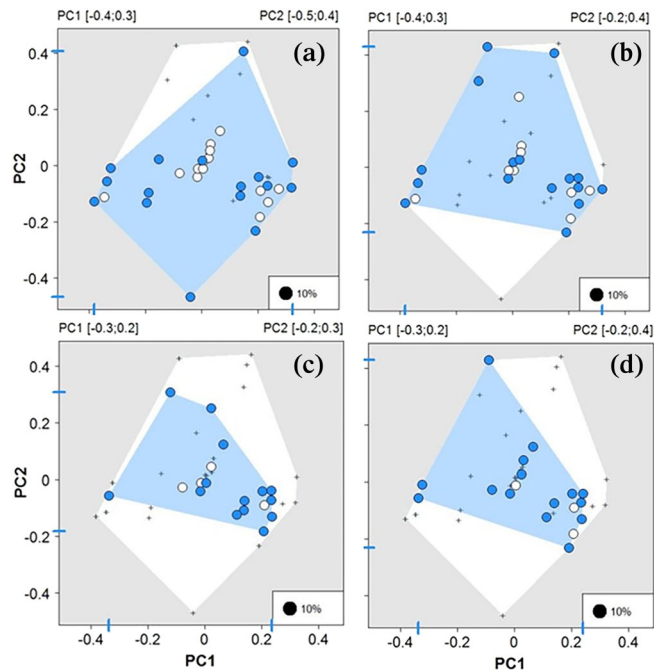

FIGURE 5 Graphical representation showing the functional richness indices calculated for each zone in each period. (a) and (b) represent the outer and inner zones for the 1983–1985 period, while (c) and (d) depict the outer and inner zones for the 2017–2019 period. Blue circles, species present in the zone/period but may also be present in other zones/periods; white circles, species exclusively present in that zone and absent in other zones/periods; crosses, species absent in the community but present in some other zone/period within the species pool.

4 | DISCUSSION

In this study, we investigated the spatiotemporal dynamics of functional diversity and beta diversity within the fish community of Sepetiba Bay, a coastal marine system that has been susceptible to environmental changes due to rapid urban and industrial expansion in recent decades. Our findings reveal a significant temporal decline in fish species richness and abundance. However, Sepetiba Bay exhibited consistent patterns of functional diversity across both time periods, suggesting that species perform similar ecological functions. Notably, there was a significant decrease in functional richness during the most recent period, which aligns with expectations as this index is particularly sensitive to changes in species numbers (Mouchet et al., 2010). The stability observed in the other functional indices could be attributed to the presence of dominant species that exhibit functional similarities across the sampled periods (Villéger et al., 2008), which is further supported by the low level of functional originality. This particular index serves as a measure of redundancy, where lower originality values indicate higher functional redundancy (Mouillot et al., 2013). Consequently, in cases where there is a decline in the abundance and/or richness of species with specific functional attributes, other functionally redundant species can compensate for this loss by performing similar functions (Biggs et al., 2020; Carmona et al., 2016; Rice et al., 2013; Rosenfeld, 2002).

Turnover has emerged as the predominant factor shaping the structure of the ichthyofauna over the assessed time frame, indicating a relatively dynamic composition over time. The majority of species exhibited low occurrence frequencies, potentially associated with their seasonal nature, as they inhabit specific locations during particular seasons or life stages (Langer et al., 2016), thus contributing to the observed high species turnover. Moreover, the high turnover within this ecosystem could be attributed to distinct local environmental conditions, where the environmental gradient acts as a selective filter, favoring species with specific physiological requirements (Baselga, 2010). Despite the decline in species richness, we observed an increase in beta diversity during the most recent period, which could be attributed to the ongoing degradation processes that the ecosystem has experienced in recent decades. This finding can be explained by the local species loss and population isolation resulting from adverse environmental conditions, leading to initially higher beta diversity (Moreno & Halffter, 2001). This pattern has been observed in previous studies, such as Socolar et al. (2016), who investigated the

TABLE 4 Functional diversity indices for each zone for the studied periods.

Period and zones	Functional richness	Functional divergence	Functional evenness	Functional originality
1983–1985				
Outer	0.531	0.602	0.282	0.024
Inner	0.646	0.876	0.305	0.047
2017–2019				
Outer	0.217	0.910	0.270	0.057
Inner	0.240	0.703	0.358	0.022

response of beta diversity to various environmental factors and noted that climate change appears to enhance beta diversity in certain systems. This increase may be a consequence of reduced local richness without resulting in regional extinctions.

Our findings reveal a significant decline in the number of resident species, indicating that, despite their adaptation to estuarine environments, local conditions were insufficient to support the persistence of many of these species. This appears to be the case for some flatfish species, such as the soles *Trinectes microphthalmus* and *Achirus declivis*, which were exclusively observed in the earlier period. These benthic species have a strong affinity to the substrate, utilizing camouflage as a feeding tactic and for protection against predators (Gibson, 2005). Changes in these environments can have a profound impact on these species due to their intimate relationship with the substrate. For instance, the dredging of the access channel in Sepetiba Bay in 2009, which increased the depth to 20 m to accommodate larger ships, has the potential to disrupt these species and their habitats. While necessary for maintaining navigable channels at suitable depths, this activity carries a range of detrimental effects. Dredging has the potential to dislodge benthic organisms along with sediments, altering sediment properties and suspending sediment, nutrients, and pollutants stored within (Barletta et al., 2016; Ponti et al., 2009). Moreover, the displacement and extraction of sediment during dredging can lead to the burial and mortality of eggs and larvae, as well as disrupt the migration of eggs, larvae, and adult organisms (Soinski et al., 2022).

The stability of functional structure and the pronounced species turnover observed in ecosystems with extensive environmental gradients, such as Sepetiba Bay, seem to be linked to the prevalence of dominant functional groups. Within these groups, species are replaced based on their respective responses to environmental filters that favor specific functional traits. As noted by Baselga (2010), species dispersal across different locations primarily occurs through turnover when environmental filtering serves as the key mechanism shaping the faunal structure, favoring species with distinct physiological requirements. Teichert et al. (2017) conducted a study examining functional redundancy patterns within the ichthyofauna across different environments. In estuarine environments, Teichert et al. (2017) discovered that environmental conditions act as a filtering mechanism, closely linking diversity to both environmental heterogeneity and physiological limitations. This phenomenon leads to numerous species distributed throughout the estuaries sharing similar functional attributes. Similar patterns have been observed in previous studies conducted in Sepetiba Bay, such as the research by Gomes-Gonçalves et al. (2020). In their study, they assessed the taxonomic and functional diversity patterns of the ichthyofauna in deep bay areas over two time periods (1993–1995 and 2012–2015). They noted a decline in taxonomic distinctiveness, consistent with the patterns of species richness and abundance observed. These findings suggest an environmental filtering process, wherein species from unique taxonomic groups are replaced by species from existing taxonomic groups that possess similar functional characteristics and environmental preferences (Gomes-Gonçalves et al., 2020).

Human activities pose significant threats to coastal ecosystems, resulting in habitat structure alterations and causing extensive damage to local communities. These modifications may have created current conditions that differ from previous ones, potentially explaining the high species turnover observed. Although our study did not specifically assess temporal changes in the environmental parameters, previous research has demonstrated the detrimental impacts of industrialization, particularly since the 1980s, leading to significant contamination by heavy metals (Costa et al., 2011; Cunha et al., 2009; Gonçalves et al., 2020). This contamination could have imposed limitations on the presence of many species. Additionally, the Baía de Sepetiba experienced a major environmental incident in 1996, often referred to as an environmental "tragedy," when the containment dam of the Mercantil Industrial Ingá company ruptured, causing pollution by Zn and Cd (Pinto, 2005). This event could have played a role in the considerable reduction of fish species' richness and abundance within the bay.

This study revealed consistent findings when comparing two periods (1983–85 and 2017–2020) spanning over three decades. During this time, significant changes occurred in the bay due to the influence and expansion of anthropogenic activities in its vicinity. In this interim period, there was a marked decrease in functional richness, not reflected in the other three examined functional indices (evenness, divergence, and originality). Thirteen functional groups were detected, some of which contained only one species, raising concerns about the loss of ecosystem functions due to ongoing changes, with a predominance of turnover over nestedness. This highlights the necessity to protect multiple areas to safeguard a greater diversity of species.

The method utilized, beach trawling, along with its consistent application over time, enabled spatial–temporal comparisons and proved to be effective in achieving these results. However, like all sampling methods, trawling exhibits selectivity and is less efficient in capturing larger and faster-swimming fish. It is crucial to acknowledge the limitations of this method. However, despite these limitations, its efficiency is evident, as demonstrated by the reported results above. According to Margules and Pressey (2000), it is not feasible to document every species within a given area, therefore it is imperative to acknowledge this inherent knowledge gap and embrace progressively more effective methods to encompass the maximum possible biological diversity. Since all fish sampling methods exhibit selectivity (Franco et al., 2012), the selection of a sampling methodology should consider the objectives and characteristics of the habitat under investigation. To conduct this study, we utilized a 30-year-old database, employing consistently similar sampling techniques involving a 5 mm mesh beach seine net. This approach enabled a precise comparison of ichthyofauna composition between the two assessed time periods, despite the method's inherent limitations.

Our findings underscore the importance of utilizing various diversity indices to conduct a comprehensive assessment of biological communities. We observed that the turnover component made the most significant contribution to beta diversity, indicating the need for conservation efforts to prioritize the establishment of different

protected areas (Langer et al., 2016), since turnover dominance implies an approach considering multiple sites within a region, to ensure the protection of the greatest number of species (Socolar et al., 2016; Wright & Reeves, 1992). Considering the significant loss of species richness and abundance, along with the decline in functional richness identified in our study, it becomes imperative to continue environmental monitoring efforts, engage in discussions to mitigate local anthropogenic impacts, and establish conservation practices to ensure the preservation of biodiversity. By employing functional diversity and beta diversity indices to examine spatial–temporal variations, we aimed to enhance the existing understanding of this coastal system. The adoption of diverse approaches facilitates a more holistic estimation of biological diversity and complements the existing information.

CONFLICT OF INTEREST STATEMENT

The authors declare that they have no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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REFERENCES

Anderson, M. J., Ellingsen, K. E., & McArdle, B. H. (2006). Multivariate dispersion as a measure of beta diversity. *Ecology Letters*, 9, 683–693. https://doi.org/10.1111/j.1461-0248.2006.00926.x

Anderson, M. J., Gorley, R. N., & Clarke, K. R. (2008). PERMANOVA for PRIMER: guide to software and statistical methods. Plymouth: primer-E. Available on internet at http://www.primer-e.com

Araújo, D. F., Peres, L. G. M., Yepez, S., Mulholland, D. S., Machado, W., Tonha, M., & Garnier, J. (2017). Assessing man-induced environmental changes in the Sepetiba Bay (southeastern Brazil) with geochemical and satellite data. *Comptes Rendus Geoscience*, 349, 290–298. https://doi.org/10.1016/j.crte.2017.09.007

Araujo, F. G., Azevedo, M. C. C., & Guedes, A. P. P. (2016). Inter-decadal changes in fish communities of a tropical bay in southeastern Brazil. *Regional Studies in Marine Science*, 3, 107–118. https://doi.org/10.1016/j.rsma.2015.06.001

Azevedo, M. C. C., Araújo, F. G., Cruz Filho, A. G., Pessanha, A. L. M., Silva, M. A., & Guedes, A. P. P. (2007). Demersal fishes in a tropical bay in southeastern Brazil: Partitioning the spatial, temporal and environmental components of ecological variation. *Estuarine, Coastal and Shelf Science*, 75, 468–480. https://doi.org/10.1016/j.ecss.2007.05.027

Barcellos, C., & Lacerda, L. D. (1994). Cadmium and Zinc source assessment in the Sepetiba Bay and basin region. *Environmental Monitoring and Assessment*, 29, 183–199. https://doi.org/10.1007/bf00546874

Barcellos, C., Lacerda, L. D., & Ceradini, S. (1997). Sediment origin and budget in Sepetiba Bay (Brazil) — An approach based on multielemental analysis. *Environmental Geology*, 32(3), 203–209. https://doi.org/10.1007/s002540050208

Barletta, M., Cysneiros, F. J. A., & Lima, A. R. A. (2016). Effects of dredging operations on the demersal fish fauna of a south American tropical-subtropical transition estuary. *Journal of Fish Biology*, 89, 890–920. https://doi.org/10.1111/jfb.12999

Barletta, M., & Lima, A. R. A. (2019). Systematic review of fish ecology and anthropogenic impacts in south American estuaries: Setting priorities for ecosystem conservation. *Frontiers in Marine Science*, 6, 237. https://doi.org/10.3389/fmars.2019.00237

Baselga, A. (2010). Partitioning the turnover and nestedness components of beta diversity. *Global Ecology and Biogeography*, 19, 134–143. https://doi.org/10.1111/j.1466-8238.2009.00490.x

Baselga, A., & Orme, C. D. L. (2012). Betapart: An R package for the study of beta diversity. *Methods in Ecology and Evolution*, 3, 808–812. https://doi.org/10.1111/j.2041-210X.2012.00224.x

Biggs, C., Yeager, L., Bolser, D., Bonsell, C., Dichiera, A., Hou, Z., Keyser, S., Khursigara, A., Lu, K., Muth, A., Jr., & Erisman, B. (2020). Does functional redundancy affect ecological stability and resilience? A review and meta-analysis. *Ecosphere*, 11, e03184. https://doi.org/10.1002/ecs2.3184

Borges, H. V., & Nittrouer, C. A. (2016). Sediment accumulation in Sepetiba Bay (Brazil) during the holocene: A reflex of the human influence. *Journal of Sedimentary Environments*, 1(1), 90–106. https://doi.org/10.12957/jse.2016.21868

Brandl, S. J., Emslie, J. M. J., Ceccarelli, D. M., & Richards, Z. T. (2016). Habitat degradation increases functional originality in highly diverse coral reef fish assemblages. *Ecosphere*, 7, e01557. https://doi.org/10.1002/ecs2.1557

Camara, E. M., Azevedo, M. C. C., Franco, T. P., & Araújo, F. G. (2019). Hierarchical partitioning of fish diversity and scale-dependent environmental effects in tropical coastal ecosystems. *Marine Environmental Research*, 148, 26–38. https://doi.org/10.1016/j.marenvres.2019.05.006

Carmona, C. P., Bello, F., Manson, N., & Leps, J. (2016). Traits without borders: Integrating functional diversity across scales. *Trends in Ecology and Evolution*, 31(5), 382–394. https://doi.org/10.1016/j.tree.2016.02.003

Clarke, C., Hilliard, R., Junqueira, A. O. R., Neto, A. C. L., Polglaze, J., & Raaymarkers, S. (2004). Ballast water risk assessment, port of Sepetiba, Federal Republic of Brazil. *GloBallast Monograph Series*, 14, 63. https://books.google.com.br/books/about/Ballast_Water_Risk_Assessment.html?id=PVQzvgAACAAJ&redir_esc=y

Costa, L. C., Ferreira, A. P., & Neves, E. B. (2011). Aplicação do sistema de projeção de poluição industrial (Modelo IPPS) na bacia hidrográfica da

Baía de Sepetiba (Rio de Janeiro, Brasil): estudo de caso. *Caderno Saúde Coletiva*, 19(1), 66–73.

Cunha, B. C. A., Rocha, D. S., Geraldes, M. C., Pereira, S. D., & Almeida, A. C. (2009). Pb isotopic signatures in sediments of a subtropical coastal lagoon: Anthropogenic source for metal contamination in the Sepetiba Bay (SE – Brazil). International coastal symposium. *Journal of Coastal Research*, 56, 797–801 (Special Issue). https://doi.org/10.4236/ijg.2015.67052

Cunha, C. L. N., Rosman, P. C. C., Ferreira, A. P., & Monteiro, T. C. N. (2006). Hydrodynamics and water quality models applied to Sepetiba Bay. *Continental Shelf Research*, 26, 1940–1953. https://doi.org/10.1016/j.csr.2006.06.010

Flynn, D. F. B., Mirotchnick, N., Jain, M., Palmer, M. I., & Naeem, S. (2011). Functional and phylogenetic diversity as predictors of biodiversity-ecosystem-function relationships. *Ecology*, 92, 1573–1581. https://doi.org/10.1890/10-1245.1

Franco, A., Pérez-Ruzafa, A., Drouineau, H., Franzoi, P., Koutrakis, E. T., Lepage, M., Verdiell-Cubedo, D., Bouchoucha, M., López-Capel, A., Riccato, F., Sapounidis, A., Marcos, C., Oliva-Paterna, F. J., Torralva-Forero, M., & Torricelli, P. (2012). Assessment of fish assemblages in coastal lagoon habitats: Effect of sampling method. *Estuarine, Coastal and Shelf Science*, 112, 115–125. https://doi.org/10.1016/j.ecss.2011.08.015

Froese, R. D., & Pauly, D. (Eds.). (2021). FishBase. World Wide Web electronic publication. www.fishbase.org

Fundação Rio-Águas. (2020). Rios de Janeiro: Um manual dos rios, canais e corpos hídricos da cidade do Rio de Janeiro. Disponível em. http://www.rio.rj.gov.br/

Gibson, R. N. (2005). *Flatfishes: Biology and exploitation*. Blackwell Publishing.

Gomes-Gonçalves, R. S., Aguiar, F. S., Azevedo, M. C. C., & Araújo, F. G. (2020). Functional stability despite anthropogenic influences on the ichthyofauna of a tropical bay. *Marine Environmental Research*, 159, 1–10. https://doi.org/10.1016/j.marenvres.2020.105016

Gonçalves, R., Oliveira, D. F., Rezende, C. E., Almeida, P., Lacerda, L. D., Gama, B. A. P., & Godoy, J. M. (2020). Spatial and temporal effects of decommissioning a zinc smelter on the sediment quality of an estuary system: Sepetiba Bay, Rio de Janeiro, Brazil. *Journal of the Brazilian Chemical Society*, 31(4), 683–693. https://doi.org/10.21577/0103-5053.20190232

Heino, J., Soininen, J., Alahuhta, J., Lappalainen, J., & Virtanen, R. (2015). A comparative analysis of metacommunity types in the freshwater realm. *Ecology and Evolution*, 5(7), 1525–1537. https://doi.org/10.1002/ece3.1460

Islam, S., Hibino, M., & Tanaka, M. (2006). Distribution and diets of larval and juvenile fishes: Influence of salinity gradient and turbidity maximum in a temperate estuary in upper Ariake Bay, Japan. *Estuarine, Coastal and Shelf Science*, 68, 62–74. https://doi.org/10.1016/j.ecss.2006.01.010

Jankowski, J., Ciecka, A., Meyer, N., & Rabenold, K. (2009). Beta diversity along environmental gradients: Implications of habitat specialization in tropical montane landscapes. *The Journal of Animal Ecology*, 78, 315–327. https://doi.org/10.1111/j.1365-2656.2008.01487.x

Langer, T., Murry, B., Pangle, K., & Uzarski, D. (2016). Species turnover drives β-diversity patterns across multiple spatial and temporal scales in Great Lake coastal wetland communities. *Hydrobiologia*, 777, 55–66. https://doi.org/10.1007/s10750-016-2762-2

Leal Neto, A. C., Legey, L. F. L., González-Araya, M. C., & Jablonski, S. (2006). A system dynamics model for the environmental management of the Sepetiba Bay watershed, Brazil. *Environmental Management*, 38(5), 879–888. https://doi.org/10.1007/s00267-005-0211-5

Lotze, H., Lenihan, H., Bourque, B., Bradbury, R., Cooke, R., Kay, M., Kidwell, S., Kirby, M., Peterson, C., & Jackson, J. (2006). Depletion, degradation and recovery potential of estuaries and coastal seas. *Science*, 312, 1806–1809. https://doi.org/10.1126/science.1128035

Macário, B. S., Olímpio, M. S., Sales, N. S., & Pessanha, A. L. M. (2021). The effect of habitat structure and the interactions between four juvenile fishes and zooplankton-prey in a tropical estuary. *Estuarine, Coastal and Shelf Science*, 261, 107528. https://doi.org/10.1016/j.ecss.2021.107528

Magalhães, M. M., Marinho, P., Domingos, A. C., Oliveira, S. M., Costa, L. O., & Azevedo, S. F. O. (2003). Microcystins (*Cyanobacteria hepatotoxins*) bioaccumulation in fish and crustaceans from Sepetiba Bay, (Brazil, RJ). *Toxicon*, 42, 289–295. https://doi.org/10.1016/s0041-0101(03)00144-2

Mahiques, M. M., Sousa, S. H. M., Furtado, V. V., Tessler, M. G., Toledo, F. A. L., Burone, L., Figueira, R. C. L., Klein, D. A., Martins, C. C., & Alves, D. P. V. (2010). The southern Brazilian shelf: General characteristics, quaternary evolution and sediment distribution. *Brazilian Journal of Oceanography*, 58, 25–34. https://doi.org/10.1590/S1679-87592010000600004

Margules, C. R., & Pressey, R. L. (2000). Systematic conservation planning. *Nature*, 405, 243–253. https://doi.org/10.1038/35012251

Mindel, B. L., Neat, F. C., & Trueman, C. N. (2016). Functional, size and taxonomic diversity of fish along a depth gradient in the deep sea. *PeerJ*, 4, e2387. https://doi.org/10.7717/peerj.2387

Molisani, M. M., Kjerfve, B., Silva, A. P., & Lacerda, L. D. (2006). Water discharge and sediment load to Sepetiba Bay from an anthropogenically-altered drainage basin, SE Brazil. *Journal of Hydrology*, 331(3–4), 425–433. https://doi.org/10.1016/j.jhydrol.2006.05.038

Molisani, M. M., Marins, R. V., Machado, W., Paraquetti, H. H. M., Bidone, E. D., & Lacerda, L. D. (2004). Environmental changes in Sepetiba bay, SE Brazil. *Regional Environmental Change*, 4, 17–27. https://doi.org/10.1007/s10113-003-0060-9

Moreno, C. E., & Halffter, G. (2001). Spatial and temporal analysis of the alpha. Beta a gamma diversity of bats in a fragmented landscape. *Biodiversity and Conservation*, 10, 367–382. https://doi.org/10.1023/a:1016614510040

Mouchet, M., Villéger, S., Mason, N., & Mouillot, D. (2010). Functional diversity measures: An overview of their redundancy and their ability to discriminate community assembly rules. *Functional Ecology*, 24, 867–876. https://doi.org/10.1111/j.1365-2435.2010.01695.x

Mouillot, D., Dumay, O., & Tomasini, J. A. (2007). Limiting similarity, niche filtering and functional diversity in coastal lagoon fish communities. *Estuarine, Coastal and Shelf Science*, 71, 443–456. https://doi.org/10.1016/j.ecss.2006.08.022

Mouillot, D., Graham, N. A. J., Villeger, S., Manson, N. W. H., & Bellwood, D. R. (2013). A functional approach reveals community responses to disturbances. *Trends in Ecology and Evolution*, 28(3), 167–177. https://doi.org/10.1016/j.tree.2012.10.004

Mouillot, D., Villéger, S., Scherer-Lorenzen, M., & Mason, N. W. (2011). Functional structure of biological communities predicts ecosystem multifunctionality. *PLoS One*, 6, e17476. https://doi.org/10.1371/journal.pone.0017476

Peláez, O. E., Azevedo, F. M., & Pavanelli, C. S. (2017). Environmental heterogeneity explains species turnover but not nestedness in fish assemblages of a Neotropical basin. *Acta Limnologica Brasiliensia*, 29, c117. https://doi.org/10.1590/S2179-975X8616

Pellegatti, F., Figueiredo, A. M. G., & Wasserman, J. C. (2001). Neutron activation analysis applied to determination of heavy metals and other trace elements in sediments from Sepetiba bay (RJ), Brazil. *Geostandards Newsletter – The Journal of Geostandards and Geoanalysis*, 25, 307–315. https://doi.org/10.1111/j.1751-908X.2001.tb00607.x

Pinto, L. M. O. (2005). Implicações da contaminação por metais pesados no meio ambiente da Baía de Sepetiba e entorno: o caso da Cia. Mercantil Ingá. Dissertação apresentada ao Curso de Mestrado em Sistemas de Gestão da Universidade Federal Fluminense.

Ponti, M., Pasteris, A., Guerra, R., & Abbiati, M. (2009). Impacts of maintenance channel dredging in a northern Adriatic coastal lagoon. II: Effects on macrobenthic assemblages in channels and ponds. *Estuarine*

Coastal and Shelf Science, 85, 143–150. https://doi.org/10.1016/j.ecss.2009.06.027

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

Rice, J., Niels, D., Gislason, H., & Pope, J. (2013). Does functional redundancy stabilize fish communities? *ICES Journal of Marine Science*, 70, 734–742. https://doi.org/10.1093/icesjms/fst071

Rosenfeld, J. S. (2002). Functional redundancy in ecology and conservation. *Oikos*, 98(1), 156–162. https://doi.org/10.1034/j.1600-0706.2002.980116.x

Socolar, J., Gilroy, J., Kunin, W., & Edwards, D. (2016). How should beta-diversity inform biodiversity conservation? *Trends in Ecology and Evolution*, 31, 67–80. https://doi.org/10.1016/j.tree.2015.11.005

Soinski, T., Gato, C., Matsumoto, A., Brazão, M., & Smith, W. (2022). Dredging on water bodies and margin interventions: Effects on fish. *Boletim Do Instituto de Pesca*, 48, 9. https://doi.org/10.20950/1678-2305/bip.2022.48.e641

Teichert, N., Lepage, M., Sagouis, A., Borja, A., Chust, G., Ferreira, M., Pasquaud, S., Schinegger, R., Segurado, P., & Argillier, C. (2017). Functional redundancy and sensitivity of fish assemblages in European rivers. Lakes and estuarine ecosystems. *Scientific Reports*, 7, 17611. https://doi.org/10.1038/s41598-017-17975-x

Törnroos, A., Pecuchet, L., Olsson, J., Gårdmark, A., Blomqvist, M., Lindegren, M., & Bonsdorff, E. (2019). Four decades of functional community change reveals gradual trends and low interlinkage across trophic groups in a large marine ecosystem. *Global Change Biology*, 25(4), 1235–1246. https://doi.org/10.1111/gcb.14552

Villeger, S., Mason, N. W. H., & Mouillot, D. (2008). New multidimensional functional diversity indices for a multifaceted framework in functional ecology. *Ecology*, 89, 2290–2301. https://doi.org/10.1890/07-1206.1

Villeger, S., Miranda, J., Flores-Hernandez, D., & Mouillot, D. (2010). Contrasting changes in taxonomic vs. functional diversity of tropical fish communities after habitat degradation. *Ecological Applications*, 20(6), 1512–1522. https://doi.org/10.1890/09-1310.1

Violle, C. P. B., Pacala, S. W., Enquist, P. J., & Kattge, J. (2014). The emergence and promise of functional biogeography. *Proceedings of the National Academy of Sciences*, 111(38), 13690–13696. https://doi.org/10.1073/pnas.1415442111

Whittaker, R. H. (1972). Evolution and measuremen t of species diversity. *Taxon*, 21, 213–251. https://doi.org/10.2307/1218190

Wright, D. H., & Reeves, J. H. (1992). On the meaning and measurement of nestedness of species assemblages. *Oecologia*, 92, 416–428. https://doi.org/10.1007/BF00317469

SUPPORTING INFORMATION

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