Anthropogenic activities decrease functional richness over time, but not other functional aspects of the fish community in a tropical bay

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ABSTRACT

Functional diversity can be used to help understanding the processes shaping biological communities and the effects of human disturbances on the ecosystem's services. Untangling these biological processes is crucial to apply effective policies aiming the biodiversity conservation. Temporal changes (1983–1985, 1999–2001 and 2017–2019) in five functional indices (functional richness, divergence, evenness, specialization and originality) of the fish communities in two zones (inner and outer) of shallow areas in a tropical bay heavily human-impacted in recent decades were evaluated. The tested hypothesis was that functional indices decrease over time, because of the environmental degradation. A substantial decrease in species richness and abundance in the more recent periods (1999–2002 and 2017–2019) compared to 1983–1985 was observed. However, the functional structure remained relatively stable, with the exception of functional richness that showed a significant decrease over time. The apparent stability in the other functional indices may be due to the presence of dominant and functionally redundant species over time that compensates for the loss of species while maintaining similar functions. It was also discovered that decreases in fish richness in shallow bay areas result in losses in functional richness, with resident fish and benthivorous species being the most affected by environmental degradation. By employing a comprehensive approach that integrates the use of functional indices and taxonomic diversity to assess temporal changes in the fish community, it provides a broader understanding of ecological processes. Such insights could prove invaluable in guiding the implementation of conservation strategies.

1. Introduction

The growing increase of anthropic actions in coastal areas around the world results in major impacts on the structure and biological composition of aquatic communities (Loreau et al., 2001), with increasing losses of biodiversity and/or changes in the species abundance, which may be accompanied by the loss of important ecological functions (Hughes et al., 2003; Costanza et al., 2014; Gomes-Gonçalves et al., 2020). Effective methods for monitoring and identification of changes in biodiversity are needed. Biological diversity at a local scale is more than evaluating species richness, since assemblages with the same richness can present different species compositions and also different functions (Harper and Hawksworth, 1994). For example, when an environmental disturbance occurs, it is likely to have direct effects on local fauna, and even if the number of species increases or remains stable, the composition may be continually changing (Lindholm et al., 2020; Pawluk et al.,

2022), also affecting ecosystem functions and the way species are established in time and space. It is essential that these changes are monitored and understood with a view to conserving biodiversity.

There has been a growing interest in studies on functional diversity in the recent decades (Villeger et al., 2010; Brandl et al., 2016; Zhou et al., 2019; Fontrodona-Eslava et al., 2021) aiming to investigate how communities are organized, based on a set of traits relevant to the responses of species to the ecosystem (Dias et al., 2013). It is possible to ordinate species in a multidimensional space according to the functional attributes and, from the species distribution patterns to access the different functional groups.

How species relate to each other as competitors or members of a network of interactions, and how they face similar environmental constraints, can be assessed through functional indices (Mouillot et al., 2007). These approaches that group fish into different functional groups that exploit similar environmental resources (Root, 1967) increase the

understanding on the use of environments by fish because it is directly linked to the functions that the species play in the environment in which they are part (Diaz and Cabido, 2001; Flynn et al., 2011; Mouillot et al., 2011).

Functional diversity is key to linking community composition to ecosystem processes, and this is particularly important when disturbances occur that alter the structure of communities (Mammola et al., 2021). Functional diversity indices can behave independently with distinct responses to environmental changes, indicating that each measure is a unique dimension of fish diversity and that environmental changes may affect them differently (Edie et al., 2018). Stability in functional indices usually occurs when there is a redundancy of functional traits, with several species performing similar functions. However, when the communities present high functional originality, with several species with sets of unique traits, the taxonomic loss may reflect in losses of important ecosystem functions (Brandl et al., 2016). Furthermore, differential distributions of species composition, where competitive (low disturbance) and disturbance tolerant (high disturbance) traits dominate, are good indicators of ecosystem stability (Biswas and Mallik, 2011).

Estuarine environments are among the most productive ecosystems in the world, having a great biodiversity and density of organisms (Costanza et al., 1997). These transitional environments present high fluctuations in environmental conditions with the biodiversity being composed of some central and persistent species and some regular or occasional visitors (Magurran and Henderson, 2003). Despite their enormous ecological value, these areas have been suffering strong impacts, mainly due to the disorderly growth of human activities in their surroundings (Lotze et al., 2006). In this context, the Sepetiba Bay is an estuarine environment located about 60 km of the city of Rio de Janeiro, Brazil, which has suffering major impacts in recent decades, when anthropogenic activities such as agriculture and fishing were replaced by industrial enterprises. These human activities along the bay surroundings result in decreases in the environmental quality, such as habitat degradation, and pollution due to the release of effluents directly into the Bay through rivers and drainage channels (Pellegatti et al., 2001; Araújo et al., 2017a). Increased urban and industrial growth contributed to increase degradation in this coastal area with an estimated substantial loss of approximately 26% of mangrove area over three decades, thus affecting the local biodiversity (Araújo et al., 2017a).

The ichthyofauna of Sepetiba Bay has been monitored in recent decades, with both biological and environmental data covering more than 30 years (Araújo et al., 2016, 2017b). Decreasing changes in fish richness and abundance have been reported and associated with changes in environmental conditions (Araújo et al., 2016, 2017b). In addition, more recent studies have identified changes in the functional diversity of the ichthyofauna (Gomes-Gonçalves et al., 2020; Gomes-Gonçalves and Araujo, 2024). Gomes-Gonçalves et al. (2020) observed significant changes in functional originality, which is particularly concerning, as losses of species with unique characteristics also represents losses of their functions in the ecosystem.

The ichthyofauna that uses shallow waters are close to the influences of impacted areas at the bay shoreline, being more subject to changes in taxonomic composition and structure. Changes in species composition and functional diversity in fish communities in two bay zones (inner, outer) in three periods over three decades (1983–1985; 1999–2001; 2017–99) were investigated. The aim was to assess whether taxonomic and functional changes occurred during these periods of intense human activity in the bay surroundings, i.e., considering that previous studies indicated a decline in the fish taxonomic richness in this area, it is expected that functional diversity indices have also changed over time as a result of environmental degradation. The tested hypothesis was that the ichthyofauna functional diversity indices decreased over time, except in the case of high functional redundancy. Biological patterns that can provide a basis for decision-making on prioritizing habitats for

ichthyofauna conservation in coastal ecosystems of great ecological and economic value are hoped to be identified.

2. Materials and methods

2.1. Study area

Sepetiba Bay (22°54′–23°40′S; 43°34′–44°10′ W) has an area of 450 km^2 and encompasses a wide range of habitats, including mangroves, sandbanks and small estuarine areas. The bay is is located at ~60 km of Rio de Janeiro City, south-eastern Brazil. The bay has two different zones (Fig. 1) according to depth and salinity gradients, and level of human influences (Araújo et al., 2016, 2017b). The inner zone is influenced by discharges from perennial small rivers characterized by comparatively higher turbidity and temperature (22–32 °C) and lower salinity (average ≈ 28), depth is mostly <10 m and substrate are predominated by muddy (Araújo et al., 2002; Leal Neto et al., 2006). The outer zone is near the sea connection and has contrasting environmental conditions with substrate mainly sandy, comparatively lower temperature (20–29 °C) and higher salinity (≈33) and transparency, maximum depth is ca. 28 m (Araújo et al., 2017b).

The bay supports a rich and diversified fish fauna and is used as rearing grounds by several coastal fish species (Araújo et al., 2018). The Port of Sepetiba, active since 1982 (Leal Neto et al., 2006), was initially built with a single pier to provide a bulk import terminal for coal and alumina. Since 1998, the port has a new wharf used for the import and export of various cargoes, including rolled steel, vehicles, containers and sulphur products (Clarke et al., 2004). The company Mercantil Industrial Ingá was responsible for one of the most severe environmental disasters in the region. Located in the middle bay shoreline, this industry had as its main activity the processing of ore to produce Zn with high purity, generating a large number of toxic residues, mainly Zn and Cd that were accidently discharged into the bay (Gomes et al., 2009).

From the 1970s onwards, there was a rapid industrial expansion in which chemical and metallurgical industries were installed and started to operate around the bay, bringing a series of potentially harmful influences (Barcellos and Lacerda, 1994; Molisani et al., 2004; Cunha et al., 2006). In 1982, the beginning of the activities of the Port of Sepetiba provided a great economic and industrial development of the region (Clarke et al., 2004). However, physical changes (habitat destruction) occurred on the shores of Sepetiba Bay, and these activities also brought numerous sources of pollutants, responsible for chemical contamination, such as Cd and Zn. In 1996, due to heavy rains, a tailings containment dam burst, releasing large loads of toxic waste into Sepetiba Bay (Gomes et al., 2009). In addition, other impact activities also took place in a more recent periods, such as the expansion of the Port of Sepetiba by dredging the access channel to a depth of 20 m, aiming at the operation of larger ships (Araújo et al., 2017a). Also, the Submarine Development Program (PROSUB), created with the proposal to expand the national defense structure, for the manufacture of four conventional submarines and one with nuclear propulsion, was implemented in the area. This program encompasses the construction of a Metallic Structures Manufacturing Unit, two Shipyards, a Radiological Complex and a Naval Base. Currently, Sepetiba Bay is facing serious environmental damage, such as deforestation of its surrounding areas and water warming, resulting from the installation of Thermal Power Plants in the inner bay area.

Anthropogenic activities favored the disorderly urbanization growth (Table 1), with an increment in population densities around the bay, which increased from approximately 60,000 in 1980 to an estimated 2 million people in 2000 (Leal Neto et al., 2006). This resulted in coastal habitat alteration, contributing to shoreline degradation, impoverishing of natural habitats, and increasing pollutants loads into the bay destroying shelters and rearing grounds for aquatic organisms, in particular of the local ichthyofauna (Ribeiro et al., 2013; Araújo et al., 2016, 2017b).
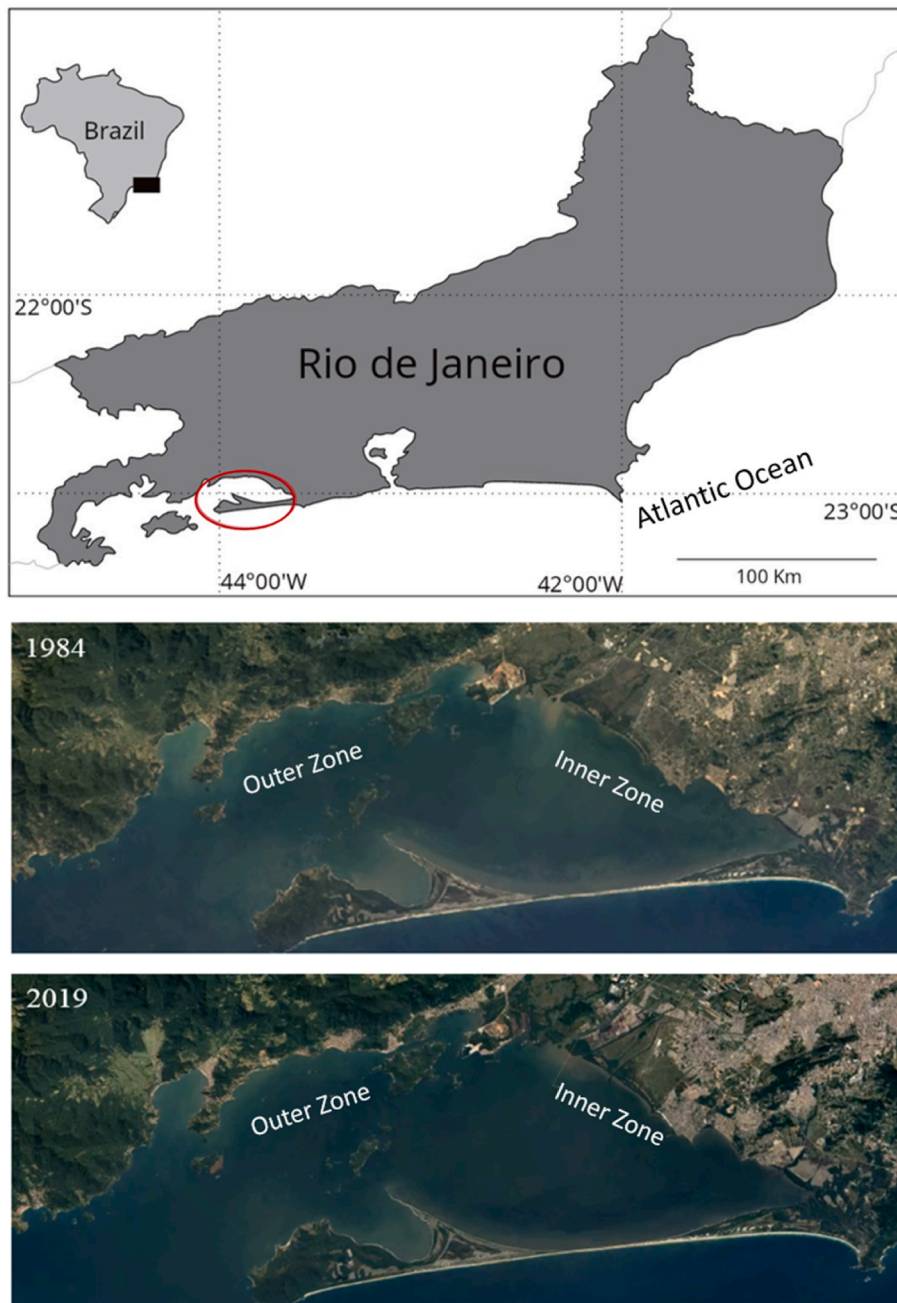
Fig. 1. Study area, Sepetiba Bay, indicating the two studied zones (inner, IZ; and outer, OZ). The changes in land use by human activities and urbanization (gray areas) can be observed in the lowest two maps, representing the increase in environmental changes between 1984 and 2019. Source: Google Earth (2021).

2.2. Fishing procedure

Samplings were carried out at three sites of two zones (inner, outer) with four replicates in two seasons (summer and winter) during three periods (1983–1985, 1999–2001 and 2017–2019), resulting in a total of 144 samples (3 sites × 2 zones × 4 replicates × 2 seasons × 3 periods). Fish were collected with a beach seine net (12 m long × 2.5 m high, with 5-mm mesh size at the wings), provided with 30 m ropes to facilitate the trawls, which were carried out perpendicularly to the shoreline from a depth of approximately 1.5 m. The hauls were carried out by two people, one at each end of the rope, covering swept area of approximately 300 m². The collected fish were fixed in 10% formalin, and after 48h, preserved in 70% ethanol. The fish were identified to the lowest possible taxonomic level, and part of the material was deposited in the Ichthyological Collection of the Laboratory of Fish Ecology at the Federal Rural University of Rio de Janeiro.

2.3. Data analyses

Species richness and fish abundance were compared among the sampling periods and two zones by using a Permutational Analysis of Variance (PERMANOVA) type I (sequential), with 999 permutations to calculate the p-values. The fish abundance and fish richness were the response variable and the periods (1983–1985; 1999–2001; 2017–2019) and zone (nested in the periods) were the fixed factors. To determine the typical species of each period, i.e., those that most contributed to the within-group average similarity, a percentage similarity analysis (SIMPER) was used (Clarke and Warwick, 2001). Prior to the analyses, the abundance data were square-root transformed to reduce the weight of the most abundant species. For the calculation of the functional

Table 1

Synthesis of the main anthropic events that occurred in Sepetiba Bay in recent decades.

Period	Event	Reference
1980–2000	Increase in population density from 60,000 to 2 million people.	Leal Neto et al. (2006)
1982	Start of activities at Sepetiba Port	Clarke et al. (2004)
1985–2015	Loss of approximately 26% of mangrove area	Araújo et al., 2017a, 2017b
1987–2013	Decrease in richness and abundance of fish species.	Araújo et al., 2016, 2017b
1996	Accidental discharges of Cd and Zn in the Bay	Ribeiro et al. (2013)
1998	Port expansion	Clarke et al. (2004)
1999	New contamination by Cd and Zn	Gomes et al. (2009)
2009	New widening of the port access channel	Gomes et al. (2009)
2010	Beginning of activities of the steel company TKCSA	Ribeiro et al. (2013)
2013	Construction of a terminal/shipyard for submarines	Araújo et al. (2016)
2022	Installation of Thermal Power Plants in the inner bay area	This study

diversity indices, a trait matrix was used, based on information on the life history for each species. The traits (Tables S1 and S2 in the Supplementary Material) were assigned based on the primary data, published journals and information available in the Fishbase (Froese and Pauly, 2021). Species traits encompassed fish characteristics associated to reproductive and trophic strategies and habitat use (Table 2). To represent groups of species that share similar functional characteristics (functional groups), a principal coordinate analysis on the trait matrix was employed.

To assess functional changes in ichthyofauna, five functional diversity indices were used: functional richness (Fric), functional divergence (Fdiv), functional evenness (Feve), functional specialization (Fspe) and functional originality (Fori). Functional richness (Fric) represents the amount of the functional space filled by species (based on their functional attributes) in a community. According to Villeger et al.

Table 2

Description of the functional guilds (reproductive, trophic and habitat use) adapted from Elliott et al. (2007)

Guilds	Description
Trophic	
Planktivore	Predominant feeding on zooplankton and phytoplankton in the water column.
Detritivore	Predominant feeding on debris and/or microphytobenthos
Piscivore	Predominant diet of small fish, which may include nektonic invertebrates.
Bentophague	Predominant feeding on invertebrates associated with the substrate
Hyperbentophague	Predominant feeding on invertebrates that live just above the sediment (hyperbenthos).
Herbivore	Predominant diet of macroalgae, macrophytes and/or phytoplankton
Opportunistic	Food made up of a wide variety of foods.
Reproductive	
Bearers	Species carry embryos, and sometimes juveniles, externally or internally.
Guarders	Species that have parental care, care for eggs and embryos until hatching occurs and often extends to the larval stages.
Nonguarders	Species without parental care
Habitat use	
Resident	Species capable of completing their entire life cycle within the estuarine environment.
Marine migrant	Species that spawn in the sea and often enter the estuary in large numbers, particularly as juveniles.
Marine straggler	Species that spawn in the sea and normally enter the estuary only in small numbers and occur more frequently in the lower regions, where salinities are approximately 35.
Semianadromous	Species whose spawning occurs in less saline regions of the estuary, carrying out reproductive migrations from the sea to higher reaches of the estuaries.

(2008), this index is influenced by the identity of the species and, more particularly, by the most extreme species in terms of functional traits that delimit the convex hull. Functional Evennes (Feve) indice measures the regularity with which each functional space is occupied by species, weighted by their abundance (Villeger et al., 2008; 2010). According to Villeger et al. (2008), this index reaches higher values when the abundance is distributed proportionally between species or when the distance in space between species is regular.

Functional divergence (Fdiv) reflects how abundances are distributed within the volume of functional attributes occupied by species (Villeger et al., 2008), reaching low values when the most abundant species have functional attributes close to the center, while higher values occur when the most abundant species have extreme functional attributes (Villeger et al., 2008; Mouillot et al., 2013). Functional specialization (Fspe) represents the distance between each species and the centroid of the functional space occupied by the assemblage, indicating how much generalist (closer to the center of the functional space) or specialists (further from the center of the functional space) the species are within the group under analysis (Mouillot et al., 2013). In contrast, functional originality (Fori) is expressed as the average distance between each species and its closest neighbor in the functional space, reflecting the degree of uniqueness of the functional attributes in the community (Mouillot et al., 2013). This last index can also be used as a metric of functional redundancy, so that the lower the originality, the greater the functional redundancy (Mouillot et al., 2013).

These indices directly measure the distribution of species in multivariate functional trait space, where species are plotted along axes capturing combined characteristics. According to Maire et al. (2015), multidimensional spaces perform better in representing the diversity of functional trait combinations, as the distance between species in low-dimensional spaces often fails to accurately reflect dissimilarity in trait values among species, potentially leading to biased estimates of functional diversity. To assess the quality of functional spaces, the R function 'quality_funct_space' was utilized, which calculates the quality of all potential spaces. This prevents the computation of functional indices in low-quality functional spaces and assists in selecting of optimal functional space. Functional diversity indices were computed using the "mFD" package (Magneville et al., 2022). For calculating and visualization of the functional indices, the followed the tutorial provided by Camille Magneville et al. (2022). The tutorial can be accessed at the following link: https://cmlmagneville.github.io/mFD/.

A Permutation Analysis of Variance (PERMANOVA) based on Euclidean distance was performed to compare indices between periods and zones. In this study, seasonal differences (summer *versus* winter) of the different indices were not compared because previous studies showed that seasonal effects on the fish communities are negligible (Araújo et al., 2016; Camara et al., 2019). Collections were conducted in both periods, only to cover a most comprehensive possibility of evaluating the three periods with samples in similar locations and wide sampling period. Significant differences among the levels of the fixed factors ($P < 0.05$) were followed by pairwise comparison tests. This analysis was performed using the PRIMER version 6.1.13 & PERMANOVA + version 1.0.3 software (Anderson et al., 2008).

3. Results

3.1. Fish composition

A total of 18,742 individuals was recorded, distributed among 86 species (Table S3 in the Supplementary Material). In 1983–1985, a total of 11,121 individuals distributed among 69 species was observed, whereas in 1999–2001, 3660 individuals were recorded distributed among 40 species, and in 2017–2019, 3961 individuals were recorded and distributed among 38 species. Species richness per sample was significantly higher in the period 1983–1985 compared to more recent periods (Pseudo-$F_{(2; 138)} = 19.55$; $P = 0.001$) and, between-zones

comparisons also indicated significant differences in species richness with higher values to the outer zones compared to the inner zone (Pseudo-$F_{(1; 139)} = 3.4$; $P = 0.02$) (Table 3). The number of individuals per sample also showed significant differences between the periods (Pseudo-$F_{(2; 138)} = 7.56$; $P = 0.001$), however, we did not observe significant differences in fish abundance between the zones (nested in the periods) (Pseudo-$F_{(1; 139)} = 1.07$; $P = 0.34$) (Table 3).

Over the three studied periods, 30 species were identified as occurring in at least 10% of the samples in a given period (Table S3 in the Supplementary Material). Of these, 11 species remained in the three periods, three species occurred only in 1983–1985 and 1999–2001, while two species occurred in 1983–1985 and 2017–2019. In addition, 10 species occurred only in 1983–1985.

Average similarity of fish species within each period were relatively low (<22%), indicating a great variability in fish species composition among the samples (Table 4). The silverside *Atherinella brasiliensis* (Quoy and Gaimard, 1825) was the species that most contributed to the within-group average similarity in all periods according to SIMPER analysis (Table 4). Others species that also had significant contribution to the average similarity in each period were the anchovies *Anchoa januaria* (Steindachner, 1879) and *A. tricolor* (Spix and Agassiz, 1829), which contributed most to the average similarity in 1983–1985 (Table 4). A high contribution to within-group average similarity of *A. tricolor* was also observed in the period 1999–2001 and of *A. januaria* in the period 2017–2019 (Table 4). The mullet *Mugil liza* Valenciennes, 1836 had high contribution in 1999–2001, and the pompano *Trachinotus carolinus* (Linnaeus, 1766) in 1999–2001 and 2017–2019 (Table 4).

The predominant taxa that occurred in all three periods but were more abundant in 1983–1985 were *Atherinella brasiliensis* followed by *Micropogonias furnieri* (Desmarest, 1823) (Table 4). The anchovies *Anchoa januaria* and *A tricolor* and the mojarra *Eucinostomus argenteus* Baird and Girard, 1855 were also more abundant in 1983–1985. The kingcroaker *Menticirrhus americanus* (Linnaeus, 1758) and *Mugil liza* were abundant in 1999–2001 only. Species with high average abundance in 2017–2019 were *T. carolinus,* the marine catfish *Genidens genidens* (Cuvier, 1829) and the leatherjacket *Oligoplites palometa* (Cuvier, 1832) and *Oligoplites saurus* (Bloch and Schneider, 1801) (Table 4).

3.2. Habitat use and trophic guilds

During 1983–1985, the ichthyofauna was composed by a greater number of resident species, followed by marine visitors and marine migrants (Fig. 2A). In the following periods, a sharp decline in these three fish groups was recorded, mainly for the resident species (Fig. 2A). In 1999–2001 and 2017–2019, the greatest number of fish species were marine migrants (Fig. 2A).

Regarding to the trophic groups, the highest species richness was of benthophagous and hyperbentophagous fish recorded in 1983–1985 compared to 1999–2001 and 2017–2019. These two groups had a high decrease in the last two periods, while the other trophic groups did not change much in species richness among the three periods. (Fig. 2B).

Table 3

Means and standard deviation of the number of species and the number of individuals for each sampling period in the two zones of the Sepetiba Bay. Letters indicate significant different/equality among the sampling periods for each zone.

Period	Zone	Number of Species	Number of Individuals
1983–1985	Inner	6.3 ± 2.7[a,b]	175 ± 212[b]
	Outer	7.8 ± 4.6[a]	287 ± 430[a]
1999–2001	Inner	4.2 ± 2.5[b]	83 ± 71[b]
	Outer	3.5 ± 1.9[c]	68 ± 128[b]
2017–2019	Inner	3.1 ± 1.9[c]	94 ± 188[b]
	Outer	5.0 ± 1.6[bc]	70 ± 83[b]
Pseudo-F$_{(2;138)}$; P		19.55; 0.001	7.56; 0.001

Table 4

Species that most contributed to the within-group average abundance and average similarity for each period in the Sepetiba Bay, according to SIMPER analyses.

Average abundance in number per sample; average similarity in %.

Species	1983–1985	1999–2001	2017–2019
Average Similarity (%)	(21.16)	(16.59)	(15.42)
Atherinella brasiliensis	1.35; 7.16	0.80; 4.98	0.69; 2.45
Anchoa januaria	1.20; 3.72	–	0.82; 3.17
Anchoa tricolor	1.03; 3.00	0.57; 1.75	–
Eucinostomus argenteus	0.99; 2.02	–	0.32; 0.41
Micropogonias furnieri	0.80; 1.68	0.49; 1.41	0.45; 1.88
Menticirrhus americanus	–	0.31; 0.92	–
Genidens genidens	–	–	0.48; 1.28
Trachinotus carolinus	–	0.31; 1.62	0.56; 2.37
Mugil liza	–	0.9; 3.91	–
Oligoplites palometa	–	–	0.29; 1.10
Oligoplites saurus	–	–	0.38; 0.91

3.3. Functional groups

A principal coordinates analysis (PCoA) on the functional fish traits (Fig. 3) revealed that 37.3% of the total variation was explained by the first axis (PCoA 1) and 23.1% by the second axis (PCoA 2). This analysis allowed us to identify 16 distinct functional groups. The first functional group (Group I) exhibited a positive correlation with the first axis and consisted mainly of pelagic species with elongated body forms, including engraulid species such as *Anchoa januaria* (Steindachner, 1879), *Anchoa tricolor* (Spix and Agassiz, 1829), *Anchoa lyoleps* (Evermann and Marsh, 1900), *Lycengraulis grossidens* (Spix and Agassiz, 1829), and *Cetengraulis edentulus* (Cuvier, 1829), as well as carangid species like *Oligoplites saurus* (Bloch and Schneider, 1801) and *Oligoplites palometa* (Cuvier, 1832), along with *Atherinella brasiliensis* and other species (Fig. 3). The second functional group (Group II) was primarily composed of species with elongated bodies and benthic feeding habits. This group included detritivorous species from the Mugilidae family, as well as benthic feeding species such as the gerreid species *Eucinostomus argenteus* Baird and Girard, 1855 and *Eucinostomus gula* (Quoy and Gaimard, 1824), among others like *Menticirrhus littoralis* (Holbrook, 1847), *Menticirrhus americanus* (Linnaeus, 1758), *Micropogonias furnieri* (Desmarest, 1823), *Umbrina coroides* Cuvier, 1830, *Cynoscion leiarchus* (Cuvier, 1830), among others (Fig. 3).

The group III comprised pelagic species with high mobility and laterally flattened with symmetrical body shapes. This group included carangid species such as *Trachinotus carolinus* (Linnaeus, 1766), *Trachinotus falcatus* (Linnaeus, 1758), *Trachinotus goodei* Jordan and Evermann, 1896, *Selene vomer* (Linnaeus, 1758), and *Chloroscombrus chrysurus* (Linnaeus, 1766), as well as species like *Chaetodipterus faber* (Broussonet, 1782) and *Brevoortia aurea* (Spix and Agassiz, 1829). Group IV was composed of benthic species with a close association with the substrate, low mobility, and laterally flattened, asymmetric body shapes, including flounders such as *Citharichthys spilopterus* Gunther, 1862 and *Symphurus plagusia* (Bloch and Schneider, 1801), as well as *Citharichthys cornutus* (Günther, 1880), *Citharichthys arenaceus* Evermann and Marsh, 1900, *Etropus longimanus* Norman, 1933, and *Etropus crossotus* Jordan and Gilbert, 1882. Group V encompassed species such as marine catfishes with dorsoventrally flattened bodies, likes *Genidens genidens* (Cuvier, 1829), *Genidens barbus* (Lacepède, 1803), *Aspistor luniscutis* (Valenciennes, 1840), and *Cathorops spixii* (Agassiz, 1829), along with *Prionotus punctatus* (Bloch, 1793).

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 and Gilbert, 1882), as well as *Dactyloscopus crossotus* Starks, 1913. Group VII consisted of mobile benthic species
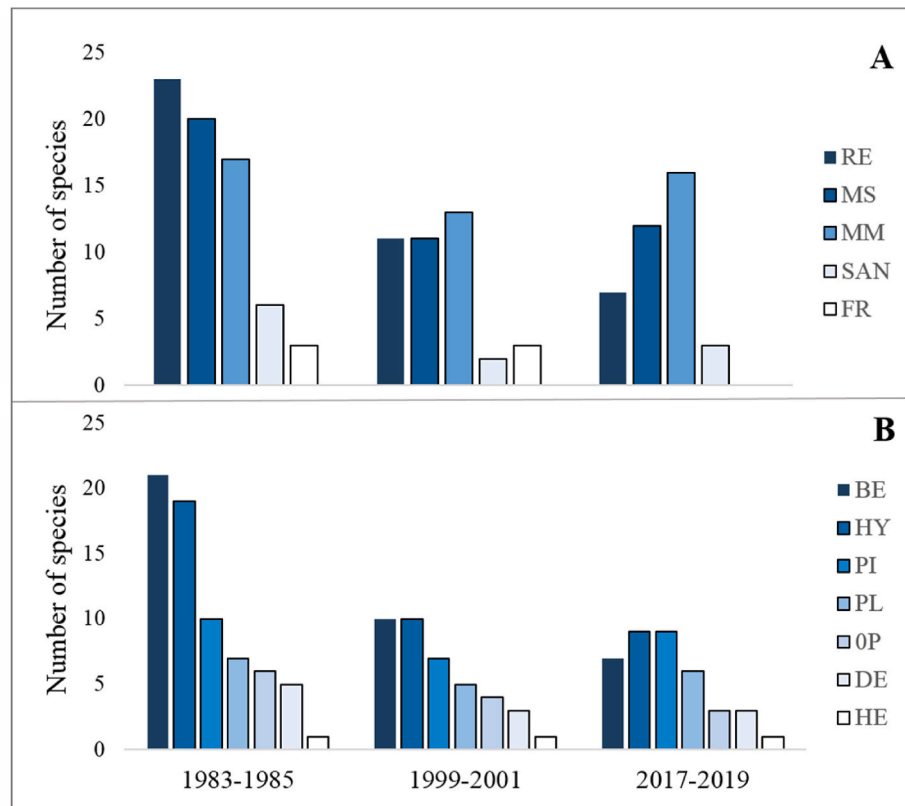
Fig. 2. Composition in number of species of guilds (habitat use, A; and trophic, B) in the three periods in the Sepetiba Bay, RJ, Brazil. Habitat use: RE, resident species; MM, marine migrants; MS, marine visitors; SAN, semi-anadromous species; FR, freshwater species; Trophic guilds: HE, herbivores; PI, piscivoress; BE, benthophagess; HY, hyperbenthophages; PL, planktivores; DE, detritivores; OP, opportunist.

characterized by elongated bodies, including species like *Odontoscion dentex* (Cuvier, 1830), *Bardiella ronchus* (Cuvier, 1830), among others (Fig. 3). Group VIII consisted of resident benthic species inhabiting inner bay areas, with low mobility and laterally flattened, asymmetric body shapes. This group included flounders such as *Achirus declivis* (Linnaeus, 1758), *A. lineatus* (Linnaeus, 1758), and *Trinectes microphthalmus* (Chabanaud, 1928), while Group IX was characterized by dorsoventrally flattened resident fishes, such as species *Sphoeroides greeleyi* Gilbert, 1900, and *S. testudineus* (Linnaeus, 1758).

Group X consisted of elongated pelagic species, including *Strongylura timucu* (Walbaum, 1792) and *S. marina* (Walbaum, 1792), as well as species like *Hyporhamphus unifaciatus* (Ranzani, 1841) and *H. brasiliensis* (Linnaeus, 1758). Group XI consisted of only one species and occurred only in the first evaluated period (*Gymnothorax ocellatus* Agassiz, 1831), which is a less mobile resident species. Group XII was also composed of only one species (*Syngnathus folletti* Herald, 1942), a cylindrical fish with male parental care (carrying the eggs). Group XIII comprised two species, *Poecilia vivipara* Bloch and Schneider, 1801, and *Jenynsia lineata* (Jenyns, 1842), which exhibit male parental care (carrying the eggs). Additionally, Group XIV was composed of only one species (*Cosmocampus elucens* (Poey, 1868)), which also exhibits parental care (carrying the eggs). The species *Monacanthus ciliatus* (Mitchill, 1818) represented Group XV, and the species *Tilapia rendalli* (Boulenger, 1897) (Group XVI) occurred only in the period from 1999 to 2001.

3.4. Functional diversity

A significant decrease in the fish functional richness was observed, with higher values in 1983–1985 (Pseudo-F = 6.09; P = 0.012) compared to the other two periods (Fig. 4, Table 5). A decrease in the functional volume filled by the ichthyofauna was observed (blue area of the convex hull, Fig. 4) between 1983 and 1985 and the two subsequent

periods, with the greatest volume in the outer zone compared to the inner zone in this first period (Fig. 4). The area of the global convex hull (demarcated by the small crosses outline; clear areas encompassing the blue areas) also indicates less loss of species (and functions) in the first period, especially in the outer zone. Regarding to the other functional diversity indices, no significant changes among the periods were detected (Table 5; Figs. S1–S6 in the Supplementary Material).

4. Discussion

A significant reduction in the number of species, impacting the functional richness, between the period 1983–1985 and the other two periods (1999–2001 and 2017–2019), whereas all other functional indices remained stable over time was indicated by the present study. This demonstrates that the large decline in the number of species that occurred after the 1980s had a considerable impact on the volume of the functional space occupied by the local community, so that some of the resources potentially available to the community are no longer used. Decreases in the functional richness reflected in the area of the convex hull indicate decreases in species with most particular traits that used to explore a given part of the resources that is no longer used by the community (Mason et al., 2005; Mouchet et al., 2010). Functional richness is a very sensitive index to changes in species richness, since the probability of occupying a wider functional space increases with the number of species (Mouchet et al., 2010).

The apparent stability of the other functional indices may be due to the effect of dominant and functionally similar species that occurred over the sampled periods. Differently from functional richness, functional divergence and evenness indices consider not only the functional identity of the species, but also their abundances (Villeger et al., 2008). In fact, the majority of the most abundant species were present in all evaluated periods, which can also be confirmed by the SIMPER analysis,
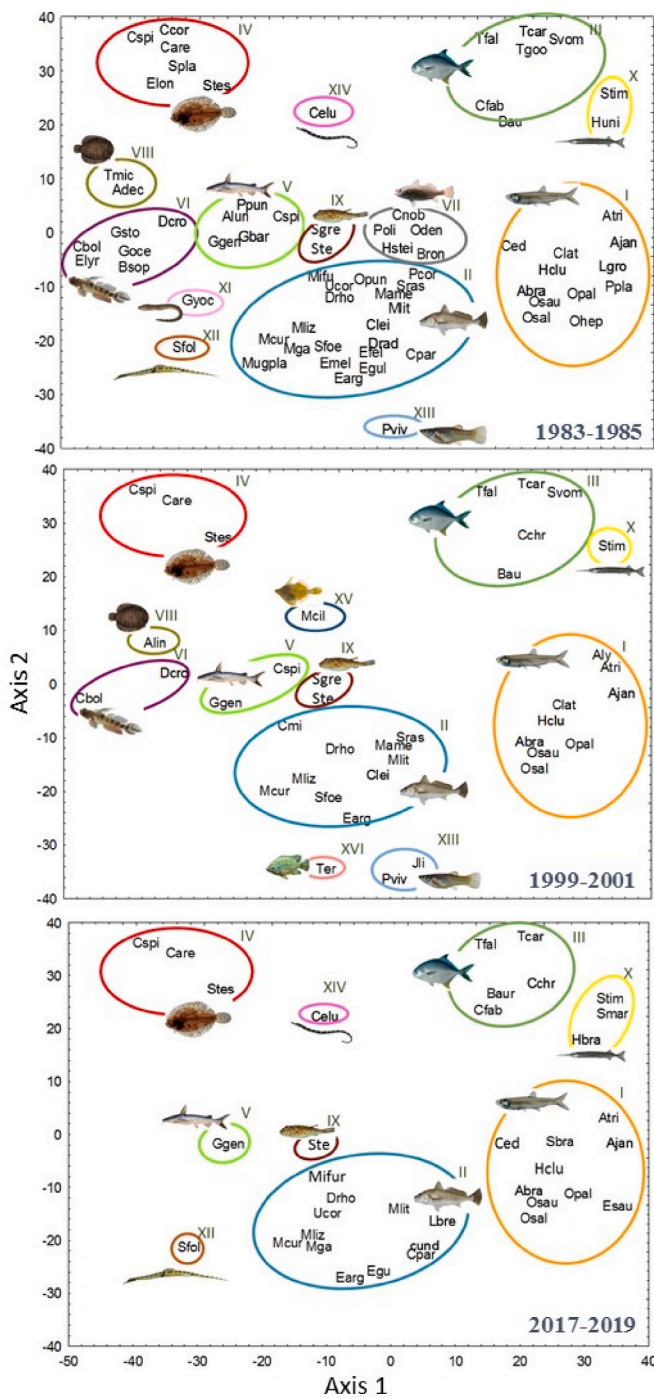
Fig. 3. Functional groups obtained through the Principal Coordinates Analysis from the functional traits of the species in each period. Species code indicated in Table S2 in the Supplementary Information.

which pointed out several species, some belonging to the same functional group, with a high contribution to similarity within each evaluated period.

While this study did not find significant differences in the specialization and functional originality indices among the studied periods, relatively low values for these indices was observed in the most recent period. Functional originality reflects the degree of exclusivity of the functional attributes (Mouillot et al., 2013), so that low values of this index may indicate redundancy of functional characteristics. Environments containing groups with functionally redundant species have a certain functional stability, since several species perform similar

functions and, if there is a disturbance that leads to a decrease in the abundance and species richness, other species that remain in the system can compensate for the loss of richness, because they perform the similar functions (Rosenfeld, 2002; Rice et al., 2013; Carmona et al., 2016). Habitat degradation has the potential to alter the relationship between species richness and functional diversity, causing a decline in these indices and increasing functional biotic homogenization (Ibarra and Martin, 2015). Decreasing in fish richness and abundance in deeper zones of the Sepetiba Bay were reported by Araújo et al. (2016), with most changes recorded in the inner zone, due to its proximity to the impact sources. Gomes-Gonçalves et al. (2020) performed a temporal comparison (1993–1995 versus 2012–2015) of fish functional indices in deep areas of the Sepetiba Bay, and reported a significant decrease in the functional originality over time. This is particularly worrying, since the loss of species with unique traits can result in losses of functions, which can represent declines in the ecosystem functions.

In the period 1983–1985, most species that occurred in Sepetiba Bay were residents. After the 1990s, a decrease in this habitat use guild was observed, suggesting that, although adapted to estuarine environments, local conditions would not be satisfactory for many resident estuarine species. Temporal changes in abiotic variables in Sepetiba Bay as temperature increases over the last few decades have coincided with decreases in fish species richness and density in this system (Araújo et al., 2016, 2017b). Estuarine environments are characterized by their marked environmental gradient, selecting species that tolerate the stressful conditions of this type of environment. Thus, if there is a significant change in the initial environmental conditions, the species may not be tolerant to this new change, not being able to maintain itself in the environment (Whitfield, 1999). In addition to changes in natural environmental conditions, human influences in the bay shoreline may also have negatively influenced the ichthyofauna, especially in relation to the resident species, which are more dependent on the habitat, since they spend their entire life cycle in the estuarine environment (Elliott et al., 2007). Also, periodic dredging of the access channel to the Port of Sepetiba is a likely impact that can affect resident species more directly than visitors or migrants, as the former are constantly present in the area, while the others occupy the environment only in certain periods of time.

The trophic fish guilds that feed on benthic organisms showed a sharp decline after the first studied period (1983–1985). This decrease in this fish species may be related to the decrease in the availability of habitats and their main food resources (benthic organisms) due to the general degradation of the habitat, which has a significant impact on the local fauna. Sediment deposition rates in the bay that range from 5 to 50 mg cm^{-2}.day^{-1} (Barcellos et al., 1997) may be hampering the development of benthic organisms. Also, increases in anthropic activities as the establishment of large industrial enterprises discharging effluents into the bay are probably contributing to the scarcity or disappearance of several benthic organisms and of fish that depend on these organisms for feeding. In addition, dredgings of the navigable channel to access the Sepetiba Port cause a series of harmful effects to the environment, such as the removal of benthic individuals with the sediment, changes in sediment properties and re-suspension of nutrients and pollutants stored in the sediment (Ponti et al., 2009). Barletta et al. (2016) investigated the effects before, during and after dredging operations on the demersal fish fauna in a tropical estuary, and observed significant changes in the ichthyofauna and in the characteristics of the environment, in which, after dredging, the fish community resumed to a biomass similar to the previous period before dredging, but with altered composition. In this study, a fish guild composed of species closely associated with the bottom, such as the flounders *Achirus declivis, Citharichthys spilopterus* and *Symphurus plagusia*, formed an important functional group in the 1983–1985. Changes in the bay sediment and other bottom characteristics probably had a relevant impact on this functional group that have very specialized traits.

The stability of the functional structure seems to be related to the
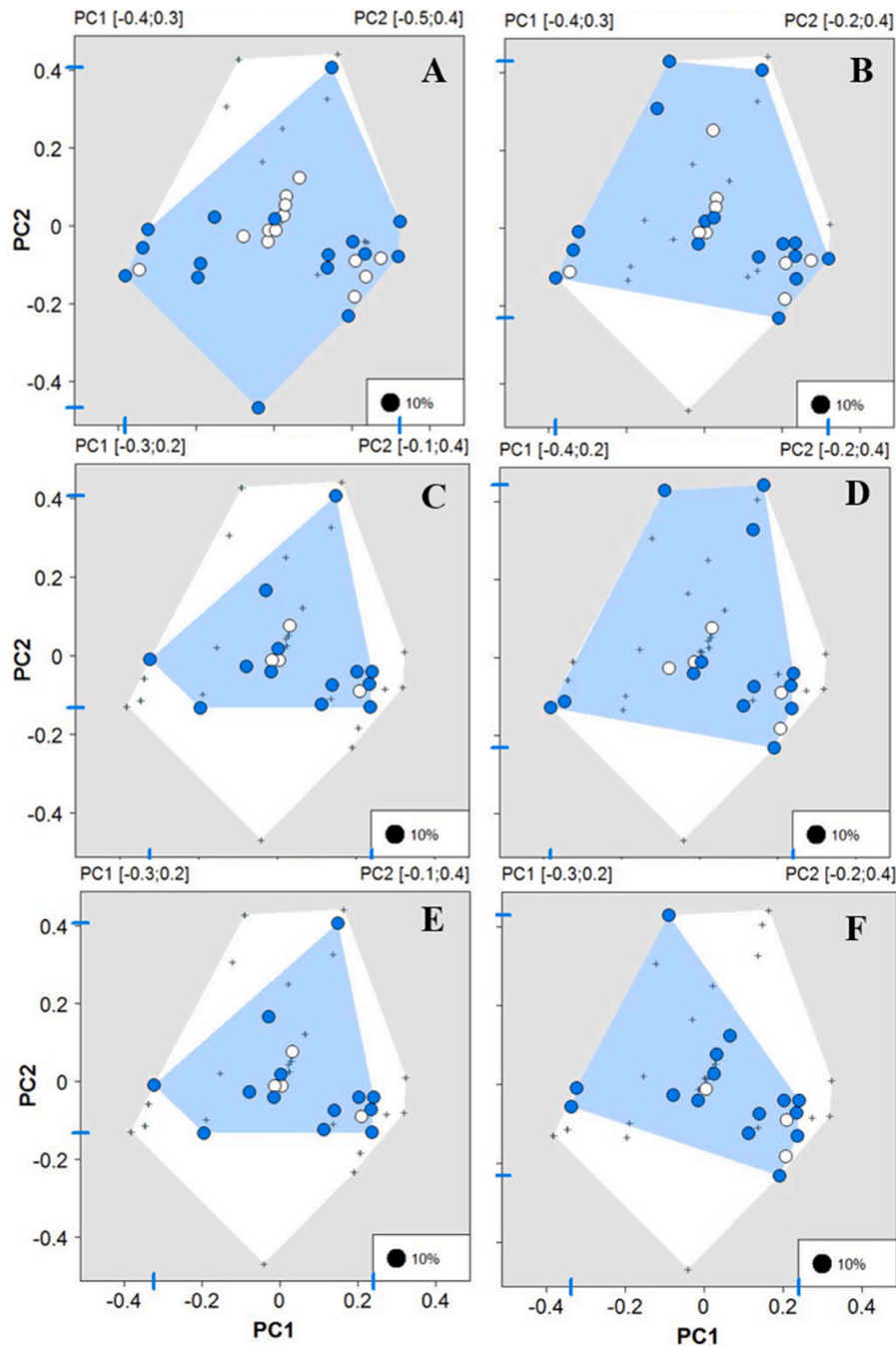
Fig. 4. Graphic representation of the functional richness indices calculated for each zone in each period. A and B, period 1983–1985, outer and inner zones, respectively; C and D, 1999–2001, outer and inner zones, respectively; E and F, 2017–2019, outer and inner zones, respectively. Blue circles, species present in the zone/period, but may be present in other zones/periods; white circles, species exclusively present in that region and absent in other zones/period; crosses, species absent in the community, but present in some other zone/period in the species pool. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

dominance of functional groups within which species are replaced according to their responses to environmental impacts. Environmental filtering as structuring mechanism of communities in Sepetiba Bay was found by Gomes-Gonçalves et al. (2020) evaluating temporal changes in fish taxonomic and functional diversity in deep areas of the Sepetiba Bay between 1993–1995 and 2012–2015. They reported that, similarly to richness and abundance, the taxonomic distinction also decreased over time with losses in functional originality. Teichert et al. (2017) investigated ichthyofauna functional redundancy in different environments and observed that, in estuarine environments, environmental conditions act as a filter, so that diversity seems to be associated with

environmental heterogeneity and physiological constraints, which contribute to the several species distributed along the estuaries share similar functional attributes.

Anthropogenic activities represent serious threats to the coastal ecosystems, altering the habitat structure and causing a series of impairments to the local communities, creating current conditions different from the previous ones. Changes in the estuarine environment can alter natural flows and lead to habitat losses, impairing fish movement between previously linked habitats, especially in early ontogenetic phases (Barletta and Lima, 2019). These changes are better assessed in medium- and long-term comparisons, which provide a picture of

Table 5
Functional diversity indices for each zone, between the studied periods. FRic, functional richness; FDiv, functional divergence; FEve, functional evenness; FSpe, functional specialization; Fori, functional originality.

Períodos	Zones	FRic	FDiv	FEve	FSpe	FOri
1983–1985	**outer**	0.531	0.602	0.282	0.394	0.024
	inner	0.646	0.876	0.305	0.580	0.047
1999–2001	**outer**	0.200	0.830	0.300	0.501	0.057
	inner	0.261	0.559	0.368	0.406	0.022
2017–2019	**outer**	0.217	0.910	0.270	0.511	0.088
	inner	0.240	0.703	0.358	0.483	0.015

biological communities, allowing more robust and assertive information on the eventual changes in species distribution and in functional traits over time (Villeger et al., 2010; Törnroos et al., 2019; Gomes-Gonçalves et al., 2020). Lindenmayer et al. (2012) listed a number of benefits of long-term studies, such as quantifying the organism responses to changes in ecosystems, allowing a better understanding of long-term ecological processes and providing essential data that can be used for the development of ecological models for ecosystem management. Thus, studies on environmental quality and fish relationships with focus on long-term approaches should be encouraged. In addition, science-based monitoring of risks to ichthyofauna in all types of coastal developments aiming to predict and minimize the impacts of these activities on biodiversity are of paramout importance (Barletta and Lima, 2019).

The effectiveness of protected areas relies not solely on access to species richness, but also on comprehending the functions that species fulfill within ecosystems. Therefore, both taxonomic and functional community descriptors are indispensable components in biological studies, essential for a comprehensive understanding of biodiversity dynamics and the formulation of effective biodiversity conservation policies. (Li et al., 2023). Substantial loss of fish species richness and abundance over the last decades in deep areas of the Sepetiba Bay (Araújo et al., 2016, 2017b) resulted in decreases in the fish functional originality (Gomes-Gonçalves et al., 2020). In the present study, it was discovered that decreases in fish richness in shallow bay areas result in losses in functional richness, with resident fish and benthivorous species being the most affected by environmental degradation. This finding contributes to our understanding of functional diversity by employing an approach that integrates functional indices and taxonomic diversity to assess temporal changes in the fish community. It offers valuable insights that could guide the implementation of conservation strategies. The aim was to continue the studies of environmental monitoring to mitigate local human impacts on tropical semi-enclosed ecosystems andprovide foundational information for conservation practices.

Ethical approval

All applicable international, national, and/or institutional guidelines for the care and use of animals were followed, including Universidade Federal Rural do Rio de Janeiro, Brazil, Animal Care Protocol (# 11874).

Compliance with ethical standards

Authors state that the research was conducted according to ethical standards.

CRediT authorship contribution statement

Rafaela de Sousa Gomes-Gonçalves: Writing – original draft, Methodology, Investigation, Formal analysis. **Laryssa Cordeiro da Silva Ferreira:** Formal analysis, Methodology. **Francisco Gerson Araújo:** Writing – review & editing, Funding acquisition, Formal analysis, Conceptualization.

Declaration of competing interest

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

Data availability

Data will be made available on request.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2024.108818.

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