



# Functional stability despite anthropogenic influences on the ichthyofauna of a tropical bay

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## ABSTRACT

Classifying species into groups based on taxonomic relationship and functions are objective approaches to evaluate environmental and anthropogenic influences on coastal fishes. We evaluated temporal (1993–1995 and 2012–2015) changes in the taxonomic and functional indices in three zones of a tropical bay heavily impacted over the recent decades. We tested the hypothesis that both indices decrease over time as result of the environmental degradation. A decrease in the taxonomic richness and abundance was observed mainly of the inner zone. The functional structure remained relatively stable, but the functional originality decreased significantly between the two periods. This functional loss is of particular concern because the loss of species with unique traits may generate a series of ecosystem damage. This information tells us that the use of functional indices is essential to complement taxonomic assessments and to detect a more detailed understanding of the real dimension of biodiversity loss in impacted environments.

## 1. Introduction

One of the key issues in ecology is the evaluation of the patterns and processes that act on the biodiversity (Anderson et al., 2011; Leprieur et al., 2011) and to make predictions aiming to protect ecosystems and/or their rational exploitation. The functional diversity is an essential tool for understanding the functioning of the ecosystem, with great predictive potential, thus complementing the taxonomic approaches (Mouillot et al., 2011; Flynn et al., 2011; Törnroos et al., 2019). The traditional diversity indices have been used for a long time in ecology to quantify changes in communities, however, these approaches do not consider the functional role of species, and their influence on ecosystems (Villéger et al., 2010; Azevedo et al., 2017; Teichert et al., 2018). The functional organization allows evaluating how the species are related to each other, as competitors or as members of a network of interactions, and how they face similar environmental restrictions (Mouillot et al., 2007).

To complement purely taxonomic information with the functional approaches for the analysis of biological communities is an effective way to assess the influences of environmental and anthropogenic impacts on the ecosystems (Azevedo et al., 2017; Silva et al., 2019). A community comprising a group of taxonomically close and functionally related

species may, in a way, be considered less diverse than a community with a similar number of species, but taxonomically distant (Clarke and Warwick, 2001). The phylogenetic diversity measures, which assess the phylogenetic relationship between species, can be expressed as the phylogenetic distance between species according to the topology of phylogenetic trees. Although these measures do not express real evolutionary distances between species, they use similarities between pairs of taxa that are proportional to the number of nodes in the phylogenetic tree (Pillar and Duarte, 2010). Among these indices, the taxonomic distinctness has gained ground in ecology for investigating the phylogenetic heterogeneity of communities (e.g. Leonard et al., 2006; Tweedley et al., 2015).

Temporal investigations that aim to assess various facets of biodiversity are essential for understanding the overall functioning of the ecosystem. Several studies have evaluated the effects of anthropogenic (e.g., pollution, habitat degradation, fisheries) and natural disturbances on ecosystems through functional diversity indices (e.g. Villéger et al., 2010; Barragán et al., 2011) and taxonomic distinctness (e.g. Barjau-Gonzalez et al., 2016; Knapp et al., 2017). In environments with high functional redundancy, habitat degradation, seja ela de origem antropog can reduce the number of species that play similar roles, suggesting a decrease in the capacity that such functions can be performed (Brandl

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et al., 2016). Knapp et al. (2017) evaluated the temporal variation of richness and taxonomic distinctness and observed a decrease in taxonomic distinctness, while species richness increased. Studies conducted with the fish assemblage in drainage basins show that the loss of functional dissimilarity typically exceeds the loss of taxonomic dissimilarity, with assemblages becoming functionally similar to one another (Petchev and Gaston, 2006; Laureto et al., 2015). Changes in the biodiversity can impair the ecological functions and, therefore, it is essential to examine the implications of losses of functions, not only in terms of species conservation, but also in terms of the sustainability of ecosystem services (Moiullot et al., 2008).

Responses of estuarine fish assemblages to human disturbance were evaluated by Teichert et al. (2018) using various taxonomic and functional indices and observed that, despite the significant decrease in taxonomic and functional descriptors, functional richness remained stable, suggesting that the traits of species removed by stressors were maintained by more tolerant species. Environmental gradients can act as biological filters, resulting in the elimination or substantial decreases in the abundance of sensitive species, which may result in convergence of functional traits (Mouillot et al., 2013). In this case, functional originality tends to decrease, since disturbances affect the sensitive species that support a specific functional role in the community. On the other hand, in environments with high functional redundancy, the similarity between vulnerable and persistent species, can guarantee the functions of these groups, despite possible declines in species richness and abundance (Brandl et al., 2016).

Estuarine environments are highly productive, being known as important habitats for many species of fish, especially in the early stages of life, providing abundant food resources and places of refuge and spawning, favoring the survival and growth of species (Azevedo et al., 2007). These ecosystems are sources of wealth for the human population, hosting a wide variety of activities, resulting in anthropogenic impacts at various levels (Costanza et al., 2014). Thus, they are among the most threatened environments, where anthropogenic impacts are responsible for an increasing rate of loss of diversity and changes in species abundance, which may be accompanied by losses of important ecological functions (Hughes et al., 2003).

The coast of the Rio de Janeiro state encompasses several semi-enclosed coastal systems that despite their great ecological importance, have suffered several environmental impacts, mainly from human activities. The Sepetiba Bay has been subjected to an increase in anthropogenic activities in recent decades, deeply affecting the ichthyofauna (Araújo et al., 2016). In the last decades, intense anthropogenic activities have brought a great load of organic and industrial effluents into the bay, enhancing eutrophication and pollution (Magalhães et al., 2003; Molisani et al., 2006; Cunha et al., 2009) and habitat degradation (Molisani et al., 2004; Cunha et al., 2006). Overfishing is also a great concern (Freitas and Rodrigues, 2014). The most recent human interferences in the bay were the enlargement of the Sepetiba Port, including dredging of the access channel to 20 m depth, the construction of a large steel factory in 2010, and a terminal for building submarines in 2013 (Araújo et al., 2016). This bay plays an important ecological role as rearing grounds for several marine fish species that use the mangroves, sandy beaches and rocky shores during part or the entire life (Araújo et al., 2002; Azevedo et al., 2006). The ichthyofauna has been widely studied in recent decades using experimental beach seines and bottom trawls (e.g., Azevedo et al., 2006; 2007; Araújo et al., 2016). However, there are not yet data reporting the use of both the taxonomic and functional approaches to evaluate the fish biodiversity. In this study, an investigation on the taxonomic and functional diversity is proposed to compare the ichthyofauna over two periods (1993–1995 and 2012–2015) sampled using similar methods, with the objective of evaluating (1) how taxonomic diversity changed, and (2) whether functional diversity indices were modified over these 20 years of intense human activities. In the event of a decline in the abundance and richness of the ichthyofauna in response to anthropogenic activities, we tested

the hypothesis that: (1) the indices of functional diversity of the ichthyofauna will show reductions over time, unless (2) there was a high functional redundancy, whereas the vulnerable species were functionally similar to the persistent species. This would imply that, despite the reduction in the taxonomic diversity, the functional diversity would remain stable, and the functions performed by the lost species would still be maintained by the persistent species.

## 2. Materials and methods

### 2.1. Study area

Sepetiba Bay (22°54'–23°04'S; 43°34'–44°10'W) is a 450 km<sup>2</sup> sedimentary embayment system located in the south of the State of Rio de Janeiro, which was originated by extensive sand deposition, which formed a barrier beach as its southern boundary (Fig. 1) The bay has a narrow connection with the sea in the east, and a wide connection in the west. The mean water temperature ranges from 21.5 °C in the winter to 26.5 °C in the summer. Average salinity ranges from 29 in the inner bay to 33 in the outer bay. This microtidal system has tide range of approximately 1 m (Mahiques et al., 2010). Predominant northeasterly and southwesterly winds activate thermal currents between the bay and the ocean. The bay is rich in organic nutrients from continental drainage, and in recent decades, it has suffered degradation due to the increases of industrial and municipal effluents brought into the bay by rivers and drainage channels from the outskirts of the city of Rio de Janeiro (Fig. 1) (Araújo et al., 2002, 2016; Pessanha and Araújo, 2003).

This coastal system can be divided into three zones (inner, middle and outer) (Fig. 1), according to environmental conditions and human influences (Araújo et al., 2002; Azevedo et al., 2006). These zones are geographically continuous and reflect hydrology and sedimentology. The inner zone is influenced by discharges from small perennial rivers, which contribute to increases in the turbidity and temperature, and decreases in the salinity; the depth is mostly <5 m and the substratum is dominated by mud (Leal Neto et al., 2006). This zone is the most altered because the influence of nearby industrial plants in the shoreline. The outer zone, located near to the limit with the ocean, has a comparatively lesser influence of anthropogenic activities, a maximum depth of ca. 30 m and exhibits more stable environmental conditions (Araújo et al., 2016). The substrate is predominantly sandy, the water temperature is comparatively lower, with comparatively higher salinity and transparency. The middle zone exhibits intermediate environmental conditions between the inner and outer zones.

In the last decades, agricultural activities have been replaced by industrial development and urbanization has increased in the bay shoreline (Fig. 1). During the 1970s, chemical and metallurgical factories (Barcellos and Lacerda, 1994) start to operate in the area. The bay has been subjected to overfishing, building construction and habitat degradation since the eighties (Barcellos and Lacerda, 1994; Molisani et al., 2004, 2006; Cunha et al., 2006). Recent enhancement of the Itaguaí Port included dredging of the access channel to 20 m depth, which enables it to receive ships up to 150,000 t (Azevedo et al., 2007). The construction of a major steel factory in the mid-2000s, and a terminal for building submarines in the late 2010s are the last sources of impact in the bay shoreline. All these recent changes have contributed to massive coastal habitat destruction and the increases in discharges of pollutant loads into the bay. It is, therefore, reasonable to suppose that such alterations are reflected in changes in the fish communities' structure and functions in the last decades.

### 2.2. Fishing procedure

Fishes were collected by bottom trawl with a 12 m long net with 25-mm mesh at the wings and 12-mm mesh at the cod end. The length of the ground rope was 8 m and the head rope was 7 m. The distance travelled was obtained using the coordinates registered at the beginning and at



**Fig. 1.** Study area, Sepetiba Bay indicating the three sampling zones (inner, IZ; middle, MZ; and outer, OZ). The increase of land use by anthropogenic activities and urbanization (gray areas) can be depicted in these two maps that represent the 1990 and 2015 years. Source: Google Earth (2019).

the end of each trawl with a global positioning system (GPS, Garmin III). For each sample, the swept area ( $A$ ) was estimated:  $A = D \times h \times X2$ , where  $D$  is the length of the path,  $h$  is the length of the head rope and  $X2$  is that fraction of the head rope which encompasses the width of the path swept by the trawl, i.e. the net spread (Sparre and Venema, 1995). The samples were taken at speeds between 2 and 2.5 knots during 20 min covering an extension of ca. 1.5 km, and it was assumed that  $X2 = 0.6$ , with the swept area corresponding to approximately 6000 m<sup>2</sup>. Each trawl followed a given depth contour to minimize the impact of any depth change during a trawl. The actual position of the sampling sites inside the zones was chosen to encompass most of the bay area. Two sampling periods were analyzed using the same collection techniques and methodologies, a first period between June 1993 to April 1995, and another period about twenty years later, between August 2012 and August 2015. In 1993 (24 samples), the 3 zones were sampled with 4 replicates in 2 seasons (winter and spring); in 1994 (36 samples), the zones were sampled with 3 replicates in the 4 seasons, and in 1995 (18 samples) with 3 replicates in 2 seasons (summer and autumn). In 2012 (24 samples), the 3 zones were sampled with 4 replicates in 2 seasons (winter and spring), and with 3 replicates in 2013 (18 samples) in summer and autumn, in 2014 (18 samples) in spring and summer, and in 2015 (18 samples) in autumn and winter. In total 78 samplings were performed in both periods, with 26 samplings in each bay zone (inner, middle and outer). The fish were fixed in 10% formalin, and after 48 h, transferred to 70% ethanol. All fish were identified to species and

counted. Voucher specimens were deposited at the reference collection of the Laboratory of Fish Ecology of the University Federal Rural of Rio de Janeiro.

### 2.3. Data analyses

The abundance of the fish community, expressed as the number of individuals, was compared among the zones and sampling periods. A Permutational Analysis of Variance (PERMANOVA) with a Type I (sequential) sum of squares to calculate p-values was used, where fish richness and fish abundance were the response variables and the periods (1993–1995; 2012–2015) and zones (nested in the periods) were the fixed factors. PERMANOVA pairwise comparisons were performed to assess differences between the periods and zones. Prior to analysis, the fish assemblage, fish richness and abundance were square root transformed. Bray-Curtis similarity matrices were calculated for the multivariate data while Euclidean similarity matrices were calculated for the univariate variables. A non-metric Multidimensional Scaling (nMDS) ordination was used to detect temporal (sampling periods) and/or spatial (zones) patterns of the fish community and a Similarity Percentage (SIMPER) analysis was used to determine the species that most contributed to within-group average similarity for zones and sampling periods.

A taxonomic distinctness index was calculated according to Clarke and Warwick (1998) and Warwick and Clarke (2001). Species were



placed within a taxonomic hierarchy, according to Nelson et al. (2016) classification into species, genus, family and order. The average taxonomic distinctness (AvTD) was calculated as the mean number of steps up the hierarchy that are taken to reach a taxonomic rank common to two species, and the computation across all possible pairs of species in an assemblage (see Clarke and Warwick, 1998; Warwick and Clarke, 2001). Thus, if two species are congeneric, one step (species-to-genus) is necessary to reach a common node in the taxonomic tree; if the two species belong to different genera but to the same family, two steps will be necessary (species-to-genus and genus-to-family), and so on, with the number of steps average across all species pairs. The various branch lengths in the ontogenetic phylogeny were determined by species richness in every taxonomic category. By default, equal step lengths are assumed, with branch lengths standardized so that the shortest path and the longest path in the tree are set from zero to 100. AvTD is not dependent on sampling effort (Clarke and Warwick, 2001). The taxonomic distinctness indices and 95% confidence funnel curve were performed by routine TAXDTEST. The funnel graph shows the average expected value of the taxonomic distinctness, which is represented by the dotted line in the center, and the confidence interval shown by the solid lines with the approximate shape of a funnel. When samples are plotted into the funnel, the taxonomic distribution is considered to be within the expected confidence limits for the number of species. PERMANOVA, SIMPER AND TAXDTEST analyses were performed using the statistics package PRIMER 6 version 6.1.13. & PERMANOVA version 1.0.3 (Anderson et al., 2008).

For functional diversity analysis, information on life history and ecological traits for each species was compiled using a range of sources (Supplementary Data, Tables S1 and S2), starting with the Fishbase (Froese and Pauly, 2020), then searches of the primary literature, regional guidebooks, and species catalogues. We converted all traits to a series of binary possibilities by scoring the trait as 1 if a species falls within a trait category and 0 if it does not. A Principal Coordinate Analysis on a Gower dissimilarity matrix was computed from the species traits to produce the functional space. This functional space was used to illustrate the species functional groups and to calculate the functional indices.

To assess functional changes in ichthyofauna, five indices of functional diversity were used: functional richness (FRic), functional divergence (FDiv), functional evenness (FEve), (Villéger et al., 2008), functional specialization (FSpe) (Bellwood et al., 2006) and functional originality (FOri) (Mouillot et al., 2008). The functional richness (FRic) represents the amount of functional space filled by the community, being estimated by the convex hull generated by positioning the distribution of species in a space of n-dimensional attributes. According to Villéger et al. (2008), the functional richness is influenced only by the identity of the species and, more particularly, by the most extreme species (in terms of functional characteristics) that delimit the convex hull. In contrast, Functional Evenness (FEve) measures the regularity with which each functional space is occupied by species, taking into account abundance (Villéger et al., 2008, 2010). This index varies from 0 to 1, reaching lower values when the main abundances belong to species that are functionally close, while higher values are found when the abundance is proportionally distributed between species or when the distance between species in space is regular (Villéger et al., 2008, 2010).

The functional divergence (FDiv) reflects how abundances are distributed within the volume of functional attributes occupied by species (Villéger et al., 2008). This index also varies from 0 to 1, reaching low values when the most abundant species have functional attributes that are close to the center, while when the most abundant species have extreme functional attributes, the functional divergence is high (Villéger et al., 2008; Mouillot et al., 2013). The functional specialization (FSpe) represents the distinction of the characteristics of the functional attributes within the assembly. 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 exclusivity of

functional attributes in the community (Mouillot et al., 2013). The functional space and the functional diversity indices were calculated using functions contained in the FD package version 1.0–12 (Laliberte and Legendre, 2010) in the R environment (version 3.5.3; R Core Team, 2019). To build graphs of the referred indices, we follow the R script MultidimFD routine provided by Sébastien Villéger (available at <http://villegier.sebastien.free.fr>).

A Permutational Analysis of Variance (PERMANOVA) based on the Euclidean distance and permutation of residuals under a reduced model was performed to compare the indices among the zones and periods. Significant differences among the factors were followed by PERMANOVA pairwise comparison tests. PERMANOVA was performed with the software PRIMER version 6.1.11 with PERMANOVA+ 1.0.1 (Anderson et al., 2008).

### 3. Results

A total of 110 species in 11 orders were recorded in the Sepetiba Bay during the two sampled periods (Supplementary data, Tables S1 and S2). In the period between 1993 and 1995, a total of 88 species and 1380 individuals were recorded, whereas in 2012–2015, 76 species and 946 individuals were recorded. The species richness/sample was significantly higher in the period from 1993 to 1995 (Pseudo-F = 63.04;  $P = 0.001$ ) and significant differences between the zones nested in the periods were detected (Pseudo-F = 25.63;  $P = 0.001$ ). The outer zone showed lower species richness, being significantly different from the inner ( $P = 0.001$ ) and middle ( $P = 0.001$ ) zones. Despite having lower species richness, the outer zone presented the highest number of recorded species, totaling 84 species. Of these, 40 occurred in both periods, while 26 occurred exclusively in the 1993–1995 period and 18 occurred exclusively in the 2012–2015 period. The middle zone presented a total of 72 species. Of these, 24 occurred only in the first period, whereas 15 occurred only in the most recent period. The inner zone presented 79 species, of which 25 were exclusive for the period between 1993 and 1995, and 17 were found only in the period of 2012/2015. Our results showed a significant decrease in abundance over time (Pseudo-F = 4.52;  $P = 0.029$ ) and between zones (Pseudo-F = 2.62;  $P = 0.028$ ), with marked differences mainly in the inner zone, with a significant decrease in the abundance of species in the last studied period.

Temporal changes in the structure of the fish community were found according to the nMDS (Fig. 2) and the PERMANOVA analysis confirmed significant differences for zones and periods (Pseudo-F = 6.24;  $P = 0.001$ ). The structure of the fish community based on the numerical

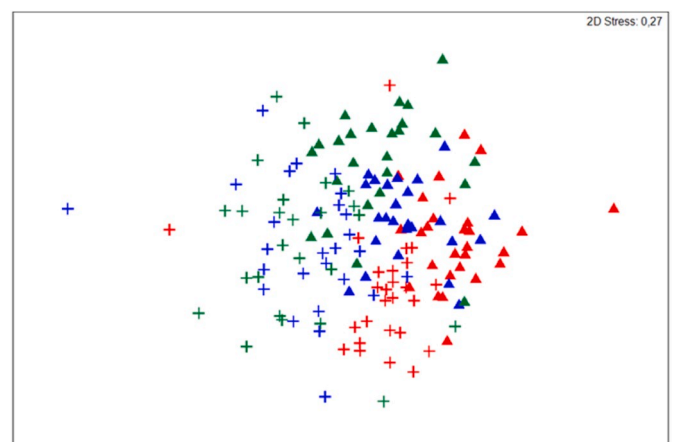


Fig. 2. Ordination diagram of non-Metric Multidimensional Scale on fish assemblage abundance with samples coded by zones and time periods. Codes: Triangle: 1993–1995; Cross: 2012–2015; red: inner zone; blue: middle zone; green: outer zone. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

abundance of fish species were strongly modified for each zone between the two periods. The highest mean dissimilarity was found in the outer Zone (88.28). The inner and middle zones presented similar values (85.34 and 85.94, respectively).

The species that most contributed to the average similarity within the inner zone that occurred in the two periods were the sciaenid *Micropogonias furnieri* (Desmarest, 1823), the ariid *Genidens genidens* (Cuvier, 1829) and the carangid *Chloroscombrus chrysurus* (Linnaeus, 1766). In addition, in the period 1993–1995, the gerreid *Diapterus rhombeus* (Cuvier, 1829) had a high contribution, while the ariid *Aspistor luniscutis* (Valenciennes, 1840) presented a high contribution in the period of 2012–2015, appearing in fourth place in the rank (Table 1). In the middle zone, in both periods, the gerreid *Eucinostomus argenteus* Baird & Girard, 1855, the ariid *G. genidens* and the triglid *Prionotus punctatus* (Bloch, 1793) appeared in the first places in the similarity ranking. In addition, the gerreid *Eucinostomus gula* (Quoy and Gaimard, 1824) had a high contribution of similarity in the first period, whereas the Sciaenidae *M. furnieri* was at the top of the ranking in 2012–2015 (Table 1). The outer zone, on the other hand, showed a lower average similarity in both periods (19.22 and 13.53, respectively). The species that appeared in both periods at the top of the ranking were the triglid *P. punctatus*, the haemulid *Orthopristes ruber* (Cuvier, 1830) and the serranid *Diplectrum radiale* (Quoy and Gaimard, 1824).

The taxonomic distinctness showed a pattern similar to the species richness, with higher values in the 1993–1995 (Pseudo-F = 17.78;  $P = 0.001$ ) but, we did not find any significant differences between the zones in both periods. Through the funnel graph, it was possible to verify that most samples fell within the 95% confidence interval, with the exception of only two cases (Fig. 3). In 1993–1995, the species richness was comparatively higher, with the majority of samples showing a higher taxonomic distinctness. On the other hand, in 2012–2015 comparatively lower species richness was detected and a wider variation in the taxonomic distinctness was recorded, with the majority of samples with lower taxonomic distinctness compared to 1993–1995.

Principal coordinate analysis (PCoA) on the functional traits (Fig. 4) revealed that 34.5% of the total variation was explained by the first axis (PCoA1) and 21.8% by the second axis (PCoA 2). This analysis enables to distinguish different functional groups. The first functional group (Group I) identified showed a direct relation to the second axis, being composed by the catfishes of the Ariidae family, with a dorsoventrally flattened body, indicating close associated to the substrate (benthic species) and with opportunist feeding habits (Fig. 4). The second group (Group II) presented an inverse relationship with the axis 1, with species of low mobility, flattened laterally asymmetrical body that characterize the flatfishes *Achirus lineatus* (Linnaeus, 1758), *Citharichthys spilopterus* Günther, 1862 and *Symphurus tessellatus* (Quoy and Gaimard, 1824). Two groups were positively correlated to axis 1, one formed by pelagic species, with planktivorous feeding habit and elongated body shape, represented by the anchovies *Anchoa tricolor* (Spix & Agassiz, 1829) and *Cetengraulis edentulus* (Cuvier, 1829) (Fig. 4; Group III). The other group (Group IV) was formed by pelagic species, with a flattened symmetrical body represented by the perciforms *Selene setapinnis* (Mitchill, 1815), *Chaetodipterus faber* (Broussonet, 1782) and *Chloroscombrus chrysurus* (Linnaeus, 1766) (Fig. 4). The Group V was composed of the other remaining species, presenting a great diversity of mobile species, with benthic habits and other different functional attributes that were not distinguished in the first two main components. (Fig. 4). In addition, *Dactylopterus volitans* (Linnaeus, 1758), a dactylopterid that has low mobility and with dorsoventrally flattened body was present in more than 20% of the samples and represented the sixth functional guild (Group VI) in second period only. Seventeen of these species were present in the functional groups of both periods (1993–1995 and 2012–2015), whereas 12 occurred only in 1993–1995, and only two species appeared in the functional groups of 2012–2015 (Fig. 4).

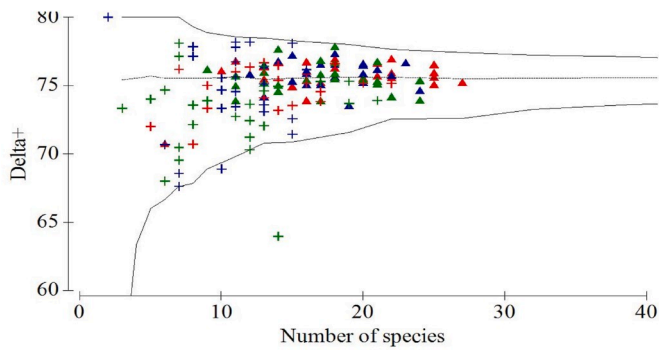
No significant changes were identified in the functional structure of the community within each zone ( $P > 0.05$ ) (Table 2), indicating that,

**Table 1**

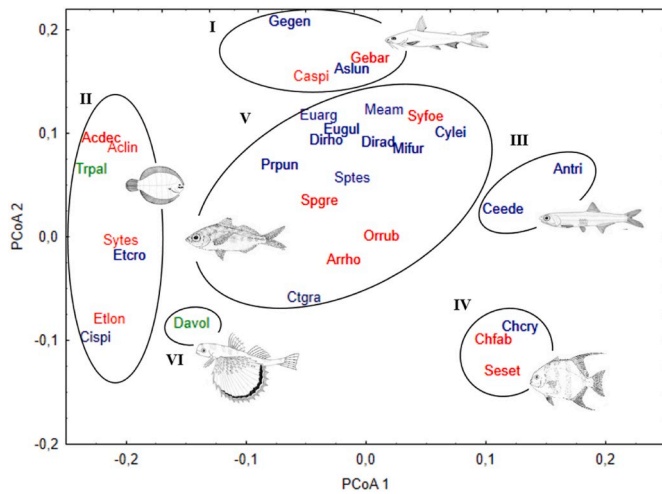
Species that most contributed to average similarity for each zone and periods in the Sepetiba Bay, according to SIMPER analyses. The average within-similarity of each species and their contribution (%) to each of zone/period in brackets. IZ, inner zone; MZ, middle zone; OZ, outer zone.

Species	1993–1995			2012–2015		
	I Z	M Z	O Z	I Z	M Z	O Z
Av. similarity (%)	17.85	22.65	19.22	20.63	16.86	13.5
<i>Eucinostomus argenteus</i>	1 (5.4)	6.4 (28.3)	1.8 (9.2)	0.9 (4.9)	1.6 (9.2)	0.4 (2.9)
<i>Micropogonias furnieri</i>	2.2 (12.4)	0.9 (3.8)	0.4 (1.9)	5.4 (25.9)	5.2 (31)	2.4 (18.1)
<i>Prionotus punctatus</i>	0.5 (2.6)	1.3 (5.9)	2.1 (10.5)	0.9 (4.4)	2.5 (14.7)	1.7 (12.4)
<i>Etropus crossotus</i>	0.4 (2.4)	0.5 (2.2)	1.6 (8)	–	0.4 (2.1)	0.5 (4)
<i>Eucinostomus gula</i>	0.5 (2.6)	1.1 (4.7)	1 (4.9)	0.3 (1.7)	–	–
<i>Genidens genidens</i>	2.7 (15.1)	5 (22.2)	–	4.8 (23.3)	1.6 (9.4)	0.7 (5.4)
<i>Diapterus rhombeus</i>	2.5 (13.7)	1 (4.6)	–	0.3 (1.6)	0.4 (2.6)	–
<i>Chloroscombrus chrysurus</i>	1.4 (7.6)	0.5 (2.4)	–	2.3 (11.1)	–	–
<i>Aspistor luniscutis</i>	0.3 (1.6)	0.7 (3.1)	–	2.3 (11)	–	–
<i>Cathorops spixii</i>	1 (5.8)	0.5 (2.1)	–	–	–	–
<i>Symphurus tessellatus</i>	–	0.8 (3.4)	1 (5.3)	0.8 (4)	0.8 (5)	–
<i>Orthopristes ruber</i>	–	1 (4.5)	1.9 (9.4)	–	–	1.6 (11.5)
<i>Menticirrhus americanos</i>	–	0.4 (1.7)	–	–	0.4 (2.2)	0.3 (2)
<i>Ctenosciaena gracilicirrus</i>	–	–	1 (5)	–	1.3 (7.7)	3.6 (26.9)
<i>Diplectrum radiale</i>	–	–	3.1 (15.9)	–	–	0.8 (5.6)
<i>Achirus declivis</i>	1.5 (8.4)	–	–	–	–	–
<i>Achirus rhomboidalis</i>	1.1 (6.3)	–	–	–	–	–
<i>Achirus lineatus</i>	0.7 (3.9)	–	–	–	–	–
<i>Cetengraulis edentulus</i>	0.5 (2.8)	–	–	–	–	–
<i>Selene setapinnis</i>	–	0.4 (1.8)	–	–	–	–
<i>Anchoa tricolor</i>	–	–	0.4 (2.3)	–	–	–
<i>Haemulon steindachneri</i>	–	–	1.7 (8.4)	–	–	–
<i>Sphoeroides greeleyi</i>	–	–	0.5 (2.5)	–	–	–
<i>Cynoscion leiarchus</i>	–	–	0.8 (3.9)	–	–	–
<i>Etropus longimanus</i>	–	–	0.6 (3.3)	–	–	–
<i>Citharichthys spilopterus</i>	–	–	–	0.5 (2.3)	–	–
<i>Trinectes paulistanus</i>	–	–	–	0.3 (1.5)	–	–
<i>Dactylopterus volitans</i>	–	–	–	–	1.4 (8)	–
<i>Stephanolepis hispidus</i>	–	–	–	–	–	0.3 (2.2)

despite the high turnover of species, the functional structure of the ichthyofauna remained stable. On the other hand, functional originality showed significant differences between the two periods (Pseudo-F = 5.54;  $P = 0.022$ ), with comparatively lower values in 2012–2015 (Fig. 5, Table 2). In the graphic representation of functional originality, species were plotted in the functional space according to their respective features, while the diameter of the blue circles is proportional to the abundance of species. In addition, the black lines represent the



**Fig. 3.** Average taxonomic distinctness of fish communities from Sepetiba Bay. The ‘expected’ taxonomic distinctness ( $\Delta+$ ) in random subsamples of the fish species included in the study (dashed line) and the 95% probability limits (continuous lines). Codes: Triangle: 1993–1995; Cross: 2012–2015; red: inner zone; blue: middle zone; green: outer zone. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)



**Fig. 4.** Functional groups depicted by the principal component analysis on the functional traits. Species are those found in at least 20% of the trawls in each zone and period. Species in blue color occurred in both periods; in red occurred only in 1993–1995; and in green occurred only in 2012–2015. Species code indicated in Table S3 in the Supporting information. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

**Table 2**

Functional diversity indices for each zone, between the two studied periods. FRic, functional richness; FDiv, functional divergence; FEve, functional evenness; FDis, functional dispersion; FSpe, functional specialization; FOri, functional originality. Significant differences (superscripts) indicated in bold.

Zone	Periods	FRic	FDiv	FEve	FSpe	FOri
Inner Zone	1993–1995	0.56 ± 0.09	0.7 ± 0.06	0.3 ± 0.05	0.5 ± 0.05	0.4 ± 0.1 <sup>a</sup>
	2012–2015	0.47 ± 0.09	0.8 ± 0.08	0.4 ± 0.1	0.5 ± 0.07	0.4 ± 0.1 <sup>a</sup>
Middle Zone	1993–1995	0.44 ± 0.1	0.8 ± 0.05	0.5 ± 0.03	0.6 ± 0.06	0.5 ± 0.07 <sup>a</sup>
	2012–2015	0.27 ± 0.21	0.7 ± 0.1	0.4 ± 0.09	0.4 ± 0.1	0.2 ± 0.2 <sup>b</sup>
Outer Zone	1993–1995	0.42 ± 0.17	0.6 ± 0.1	0.4 ± 0.06	0.3 ± 0.05	0.2 ± 0.06 <sup>b</sup>
	2012–2015	0.35 ± 0.05	0.7 ± 0.2	0.4 ± 0.1	0.4 ± 0.1	0.09 ± 0.06 <sup>c</sup>
<b>Pseudo-F;</b>		2.5;	1.81;	1.5;	2.23;	4.07;
<b>P</b>		0.08	0.17	0.273	0.096	<b>0.021</b>

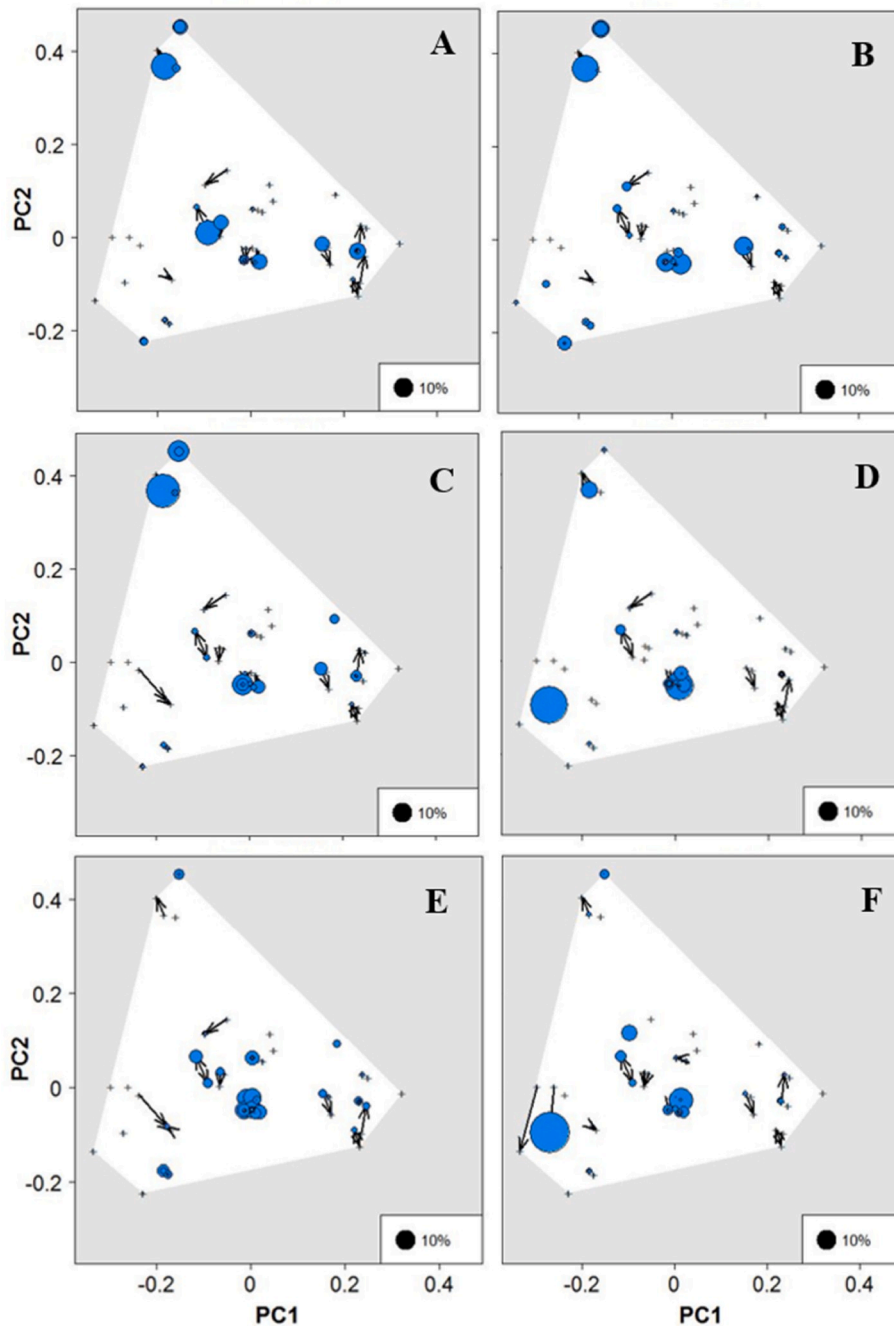
minimum functional distances between pairs of species. Thus, we observed more visible changes in the outer and middle zones (Fig. 5). The inner zone has maintained a certain stability in its originality over time (Table 2). The other graphics representation of the functional descriptors that did not differ between the two periods can be seen in the Supplementary Data (Fig. S1, S2 and S3 in the Supplementary Material).

**4. Discussion**

Our results indicate that, despite the taxonomic difference over time, the ichthyofauna functional features have remained relatively stable, which means that many species perform redundant functions, thus ensuring that, despite the losses in the species richness and abundance, main functions are being maintained by the fish community in this coastal system. The redundancy of functional features contributed to increase biological stability and its associated ecological processes, and can act as biological insurance against environmental disturbances (Rosenfeld, 2002). In addition, the highly diverse functional traits of estuarine species with rare species playing a key functional role in these ecosystems have been reported elsewhere (Teichert et al., 2017). The relationship between functional redundancy and resilience is based on the principle that, in an environment where several species perform similar functions, there are several ways to maintain the main ecosystem processes, so that, if there is a disturbance in the abundance of a given species, other ways can compensate for the change in function served by it, whose abundance has been disturbed (Rice et al., 2013). The probability of a whole functional group becoming extinct from a given community increases with the number of recognized functional groups (functional richness) but decreases with species richness and functional evenness (the distribution of species across functional groups) (Fonseca and Granade, 2001). The variability in the response to certain impacts between functionally redundant species can guarantee the representation of these groups, despite possible declines in species richness. However, if persistent species losses continue to occur over time, it is expected that there will be a reduction in the number of species that play similar roles, which may lead to the disappearance of certain functional groups essential to the overall functioning of the ecosystem. In addition, evidence for interactions among functional groups was relatively weak; however, density dependence at the functional group level suggested within-group compensation as an important stabilizing mechanism (Bell et al., 2014).

Few significant changes were observed in terms of functional characterization, assuming that the functional structure remained relatively stable. Functional originality was the only index that showed significant differences between the two studied periods, with lower values for the outer zone, suggesting that there is a redundancy of functional traits, with losses of functionally unique species. Brandl et al. (2016) evaluated the functional change of a reef fish community after a tropical cyclone and observed that, of all calculated functional indices, only the functional originality was correlated with habitat degradation, increasing after the disturbance. In this case, there was a loss of functionally similar species in susceptible groups, with simultaneous addition of functionally unique species in groups that benefited from the disturbance. The loss of the functional originality is worrying, since, with the local disappearance of species with original features, the functions performed by them are also lost, not being performed by other species in that community (Mouillot et al., 2013). Thus, the influence of species losses on the functioning of the ecosystem tends to be greater in less redundant assemblies, especially if unique traits are presented in the vulnerable species (Mouillot et al., 2008). There is a tendency for estuarine environments to present high functional richness associated to the environmental heterogeneity and physiological restrictions, which contribute for several species distributed along the estuaries to share similar functional attributes (Teichert et al., 2017).

The decrease in functional originality may be the result of environmental filtration, so that only species that have certain attributes that



**Fig. 5.** Graphical representation of the functional originality indices calculated for each zone in each period. A and B, inner zone in the periods of 1993–1995 and 2012–2015, respectively; C and D, middle zone in 1993–1995 and 2012–2015, respectively; E and F, in the outer zone in 1993–1995 and 2012–2015, respectively. Blue circles indicate the presence and the diameter is proportional to the abundance of the species. Black arrows are minimum functional distances between pairs of species. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

give them the ability to withstand limiting environmental conditions are able to survive in a given location (Bellwood et al., 2006). The reduction of the taxonomic distinctness, also observed in the present work, may be corroborating with the indication that there seems to be an environmental filtering process, since species that no longer occurred belonged to single families or genera, being replaced by species belonging to existing taxonomic groups, with similar functional features. In general, species with greater phylogenetic proximity tend to have a similar life history, reflecting in more similar niches, resulting in greater competition and greater probability of competitive exclusion (Wotom, 1992; Violle et al., 2012).

The composition of the ichthyofauna was profoundly modified between the two studied periods, as confirmed by the nMDS analysis and the high values of Bray-Curtis dissimilarity in the three studied zones. The outer zone showed the greatest dissimilarity between the two

periods, pointing out to be the most variable zone. This may be due to its greater connection to the sea, to which fish can enter and leave more easily, depending on the conditions of the bay. Although the inner zone presents greater proximity to the coastal area, suffering the greatest impact due to the anthropogenic influence, it presented greater stability in the composition of the ichthyofauna, although it has shown a great reduction in its abundance. This result may be related to the fact that altered environments hinder the dispersion of species, favoring the isolation of populations, creating a certain stability in the composition of communities (Moreno and Halfter, 2001).

The reduction in the species richness and abundance may be directly related to anthropic factors that occurred mainly in the 1980s, with the start of the activities in the Sepetiba Port, namely dredging of the access channel and population increase in the areas at the bay shoreline (Araújo et al., 2016), resulting in a large inputs of industrial and



domestic waste into the bay. Barletta and Lima (2019) and Ribeiro et al. (2008) reported changes in the structure of the marine fish community and associated it with pollution, industrial development of urbanization, discharges of biological effluents and channel dredging. These activities have direct and indirect impacts on the local fish communities, associated with changes in water quality and degradation in the physical habitat structure. Such alterations, result in the removal and redeposition of marine sediments, directly affecting the communities of benthic invertebrates and, indirectly, other higher trophic levels, mainly species that it uses the benthic fauna for feeding. These facts may be related to the marked decrease in the frequency of occurrence and abundance of several resident species that prey on benthic invertebrates, such as the demersal fish of the family Achiridae *Achirus declivis* Chabanaud, 1940 and *A. lineatus*, that use these estuarine or low salinity areas for reproduction, breeding and survival (Miller et al., 1991; Allen and Baltz, 1997).

Despite being closer to the coastal area and, consequently, to anthropogenic activities, the inner zone maintained stability of the most abundant species, with *G. genidens*, *M. furnieri* and *Chloroschombrus chrysurus* (Linnaeus, 1766) contributing 35.11% of the average similarity in the period 1993–1995, while these same species represented 60.3% of the average similarity in 2012–2015. The Sepetiba Bay has favorable environmental conditions for marine catfish (Ariidae family), which occur in tropical and subtropical coastal areas and are generally abundant in coastal waters with muddy and shallow bottoms (Araújo, 1988). The main representative species of this family in Sepetiba Bay are *G. genidens* and *Aspitor luniscutis* (Valenciennes, 1840), occurring mainly in the inner zone, and to a lesser extent in the middle zone. In the outer zone, few individuals representing this family were found in the period 1993–1995, with higher values, but decreasing in 2012–2015, namely *G. genidens*, which was identified in the top ranking of species that contributed to the average similarity of the group. Silva-Junior et al. (2013) advocate the use of *G. genidens* as a potential sentinel species, which could be used in monitoring, as it is in harmony with the dynamic characteristics of ecological processes that regulate tropical estuarine environments, being abundant and easy to capture.

In the last decades, the Sepetiba Bay has been evaluated in its taxonomic characterization, environmental conditions and trophic ecology (e.g., Pessanha and Araújo, 2014; Araújo et al., 2016). However, the present work is the first that deals with functional information. Araújo et al. (2016) evaluated the temporal changes of the ichthyofauna in Sepetiba Bay over three decades and concluded that most of the changes in the ichthyofauna occurred in the inner zone of the bay, due to its proximity to the source of impact, in comparison with areas close to the sea connection that was comparatively more stable. However, evaluating two specific times (1993–1995 and 2012–2015), we found that, differently from what was previously established, the outer zone seems to be the least stable in relation to species composition, with greater taxonomic dissimilarity between the two evaluated periods, being also the area with the greatest loss of functional originality. On the other hand, due to being closer to the area of anthropogenic influence and frequently suffering the impacts, the ichthyofauna of the resistant inner zone has had time to establish itself, with few specific variations, when compared to the outer zone, where there seems to be greater species replacement.

Our results indicate that the ichthyofauna was profoundly modified in its taxonomic features, whereas the functional diversity has hardly changed, corroborating the hypothesis that the vulnerable species appear to be functionally similar to the persistent species. Thus, despite the decline in fish species richness and abundance that occurred in Sepetiba Bay, functional diversity remained relatively stable. This means that the functions performed by lost species are still being maintained by other persistent species. Species richness alone is not sufficient to influence the functional vulnerability, especially in estuarine systems that have high functional redundancy associated to high environmental heterogeneity and physiological restrictions (Teichert

et al., 2017). This indicates that purely taxonomic indices provide an incomplete signal of the local biological variability, since the taxonomic characterization may not coincide with the functional characteristics. Thus, the use of functional indices seems to be essential to complement taxonomic assessments and to detect the real dimension of biodiversity loss in environments influenced by anthropogenic activities (Villeger et al., 2010; Rincón-Díaz et al., 2018). To understand how fish communities, respond to impacts, establishing long-term monitoring programs is essential and, because changes in species richness does not always correspond in a similar loss of functional structure, it becomes essential that there is also a monitoring of the functional characterization of biological communities. Marine population declines are ubiquitous, yet the consequences for the functioning of marine ecosystems are understudied (Luybaert et al., 2020). Conserving and restoring natural habitats is essential for promoting ecological stability. However, in scenarios when natural habitats are not viable, engineered landscapes designed to mimic the complexity of natural habitats may provide similar ecological functions (Scyphers et al., 2015). Sustainable coastal management concepts must take into account the compatibility between coastal systems function and ecosystem characteristics. Incompatibility causes either dysfunction and ecosystem degradation or the need of corrective management interventions which may have high costs to be implemented.

#### 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).

#### Author Statement

I declare, as corresponding author, that I am responsible for ensuring that the descriptions are accurate and agreed by all authors.

#### Compliance with ethical standards

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

#### 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.

#### CRediT authorship contribution statement

**Rafaela de Sousa Gomes-Gonçalves:** Formal analysis, Writing - original draft. **Fernanda Silva de Aguiar:** Formal analysis. **Marcia Cristina Costa de Azevedo:** Writing - review & editing. **Francisco Gerson Araújo:** Formal analysis, Writing - original draft.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.marenvres.2020.105016>.

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