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# The effect of an artificial reef system on the transient fish assemblages – south-eastern coast of Brazil

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*Artificial reefs (ARs) are often used to improve fishing and, consequently, the economy of a region. However, the way in which the species use the reefs may vary between fish assemblages. An assessment was made of the influence of an AR complex on the transient fish population off the northern coast of Rio de Janeiro state and, therefore, two control areas were sampled. Gillnets were used to capture individual fish in six sampling surveys. Cumulative abundance and biomass curves (ABC) were used to assess the possible effects of the reefs on the community's functional structure. In the dry season, during which the influence of the Paraíba do Sul River is smaller, a larger richness of* r*-strategy species and juveniles of* K*-strategy species was observed in the reef area compared with the control areas, suggesting that the AR acts as a protective environment for these species. During the lower river discharge period the results indicated a potential disturbance in the functional structure of the AR fish community and, therefore, a less stable environment relative to the control areas. This 'instability' warrants a positive connotation, as it indicates that the artificial reefs are harbouring species that are particularly sensitive to predation, making the reef a powerful tool for maintaining these populations on the northern coast of Rio de Janeiro.*

**Keywords:** artificial reef, transient fish, gillnet, *r-k* strategy, life history, southeastern Brazil

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## INTRODUCTION

Artificial reefs (ARs) are often used as tools for fishing management under the assumption that they will increase the populations of exploited species in the areas influenced by the reef structures (Campbell *et al*., 2011). However, the way in which such species use the reef is still unclear (Szedlmayer & Shipp, 1994; Workman *et al*., 2002; Campbell *et al*., 2011; Hackradt *et al*., 2011). One of the main questions refers to whether an artificial reef influences an exploited species as a result of an increase in its production rate due to increased resource availability, or as a consequence of a mortality reduction resulting from the reefs' inhibition of fishing activities (Grossman *et al*., 1997; Pickering & Whitmarsh, 1997; Zalmon *et al*., 2002; Fowler & Booth, 2012).

Another question relates to the functional role of the reef structure for the species targeted by local fishing. Are the reefs a source of food or shelter for the initial life stages or for populations that sustain the fish community at lower trophic levels? Do the reefs serve as reproductive areas for some of these species? In this context, the artificial reefs have more than just local value, as the species that use them during their juvenile stages may migrate to outlying areas after reaching the adult phase. Most studies of fish communities in artificial reefs adopt a qualitative approach with the use of ecological indices, such as richness, diversity, dominance indices and distance statistics. However, these measures provide only an instantaneous snapshot of the resident or transient fish community in the reef area. As a consequence, the role of artificial reefs in maintaining fish communities and fishing stocks may be underestimated (Ponti *et al*., 2002; Sherman *et al*., 2002; Campbell *et al*., 2011; Hackradt *et al*., 2011).

The life history of species that potentially use the artificial reef must also be considered, since the life strategy of each species defines its position in the trophic net. If the reef harbours species of lower trophic levels (which tend to be *r*-strategists), it is possible that the reef complex maintains populations of these species, which may serve as resources for higher trophic level species (usually *K*-strategists). Thus it is possible that artificial reefs harbour juveniles of high commercial value species, making them both an important management tool and an area useful for habitat conservation (King & McFarlane, 2003; Santos *et al*., 2011; Fowler & Booth, 2012).

Along with life-history strategies, researchers showed that abundance and biomass data could also be used to analyse impact from different sources on natural communities as fishing or pollution (Warwick, 1986; Blanchard *et al*., 2004). The Abundance and Biomass Comparison (ABC curves) method was originally proposed by Warwick (1986) to understand disturbance effects in a benthic invertebrate community. More recently, it has been applied to investigate possible fishing effects, while allowing a comparison of fish assemblages

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on a spatial and temporal scale (Blanchard *et al*., 2004; Yemane *et al*., 2005). This analysis assumes that stable environments harbour more species tending towards the *K*-strategy, whereas stressful environments (e.g. overfishing sites) have several species tending towards the *r* strategy (Clarke & Warwick, 2001). Also, length-weight relationship (LWR) can be used to compare different populations in time and space with different purposes, including the estimation of weight of a specimen based on its length, the study of allometric growth or the calculation of indexes (Froese, 2006; Teixeira de Mello *et al*., 2006). Among these, the Fulton's condition factor (*K*) has been widely used in fish biology studies. This factor can be influenced by the LWR parameters, and is based on the assumption that heavier fish of a given length are in better condition. Furthermore, by comparing *K* values of different areas, this approach can also indicate that fishes of a particular site have a higher fitness as a result of distinct biological parameters and local advantages such as food availability (Froese, 2006; Mir *et al*., 2012).

Since 1996, investigations of invertebrate and fish colonization on artificial reefs along the south-east coast of Brazil have sought to assess the role of artificial reefs in the management and conservation of local fishery resources (Zalmon *et al*., 2002; Brotto *et al*., 2006a; Krohling *et al*., 2006; Santos *et al*., 2011). More than 40 fish species have been recorded in association with artificial reefs of different materials and complexity, though most reefs are built of concrete modules, the most effective in attracting and harbouring fishes (Zalmon *et al*., 2002). Local artisanal fisheries focus mainly on such demersal and soft-bottom living fishes as members of the family Carangidae, Haemulidae and Sciaenidae caught with gillnets, which is the main gear used by the fishermen. Therefore, the deployment of artificial structures on the homogeneous and plain bottom of the northern coast of Rio de Janeiro is regarded as a promising alternative to mitigate local losses of fishery resources and for habitat conservation of soft-bottom fishes by the inhibition of trawling (Krohling *et al*., 2006).

The present study attempted to evaluate the spatio-temporal influence that the artificial reef complex in northern Rio de Janeiro State has on the transient fish assemblages in the area. If fish populations use this environment for shelter and/or recruitment, the nearby resident populations may be mainly composed of juveniles, and/or by smaller-sized species (characteristic of most *r*-strategy species) more so than populations in adjacent areas. The artificial reefs would represent a positive net effect at the level of the local fish community, reflecting the harbouring of juveniles of important fishing species and/or lower trophic levels species, which are essential for the maintenance of the regional fish community.

## MATERIALS AND METHODS

### Study area

The study area is located on the continental shelf north of Rio de Janeiro State (south-eastern Brazil), adjacent to the mouth of the Paraíba do Sul River (PSR) (Figure 1). The north coast of Rio de Janeiro is naturally depleted of rock substratum or other hard substrates, and it is covered by extensive sandy beaches with variable amounts of mud and calcareous nodules (i.e. rhodolites; Zalmon *et al*., 2002). Pluviometric precipitation in the Paraíba do Sul River drainage basin is the primary factor controlling the flow rate and exhibits two distinct periods: dry from May to September and rainy from October to April (Carvalho *et al*., 2002). Data on the average monthly flow of the Paraiba do Sul River in the region were obtained from the National Water Agency (www.ana.gov.br).

### Experimental design

Artificial reefs were deployed in March 1996 on a flat and homogeneous sandy bottom, 9m deep and 5 km offshore of the Guaxindiba Beach (21°29′S 41°00′W) on the northern Rio de Janeiro coast (Figure 1). The reef complex was initially comprised of modules of concrete pipes (12), tyre bundles (12) and brick piles (4), covering approximately 1500 $m^2$ of sea bottom (Godoy *et al*., 2002; Zalmon *et al*., 2002). Subsequently, the reef complex was increased by adding tyre bundles (N = 12) and cement prefabricated blocks (7) in February 1997 and 36 Prefabricated Reef Balls® (approximately 1.0 $m^3$) in January 2002 (Brotto *et al*., 2006a, b; Krohling *et al*., 2006).

In addition to the AR, the fish communities of two sandy bottom control areas (CTs), located approximately 1000 m south (SC) and north (NC) of the reef were sampled. These distances were based on Santos *et al*. (2010), who observed that this artificial reef has an influence radius of approximately 100 m.

At the three sites (AR and 2 CTs), 18 gillnets (25 × 7 m; 20, 30, 40 and 50 mm mesh) were used to capture individuals of different size classes, remaining underwater for approximately 24 h. Three surveys were performed in the dry season when the discharge of the Paraíba do Sul River (PSR) is lowest, and three at the rainy season, when the PSR discharge is greatest. Gillnets were chosen because the water turbidity is very high (Sechi disk ~1 m), also these gear has been traditionally used along the northern Rio de Janeiro coast (Zalmon *et al*., 2002) and have the same technical features as nets used by local fishermen. The nets were deployed above the reef modules and their position was chosen according to the current direction. A global positioning system (GPS) was used to set replicates of the gillnets within the reef complex and control areas.

### Data analyses

All of the sampled individuals were taken to the laboratory, where they were identified to the species level, and biometric length and biomass data were obtained. The main species were defined according to Dajoz's (1978) constancy (*D*), in which species found in less than 25% of the samples are considered accidental, species found in 25–50% are considered accessory and those found in more than 50% of the samples are considered constant. Only constant and accessory species were used in the analyses.

The fish number and biomass of each species per sampling period were determined at each site, and comparisons between AR and CTs for both periods (higher and lower PSR discharge) were made with a non-parametric analysis (Kruskal–Wallis test) followed by *a posteriori* test (Tukey HSD). The temporal variations of the constant and accessory species were analysed with the Pearson's Chi-squared test (Zar, 1999).

The Importance Percentage index (IP) was used to record the most important species in each area and sampling season.

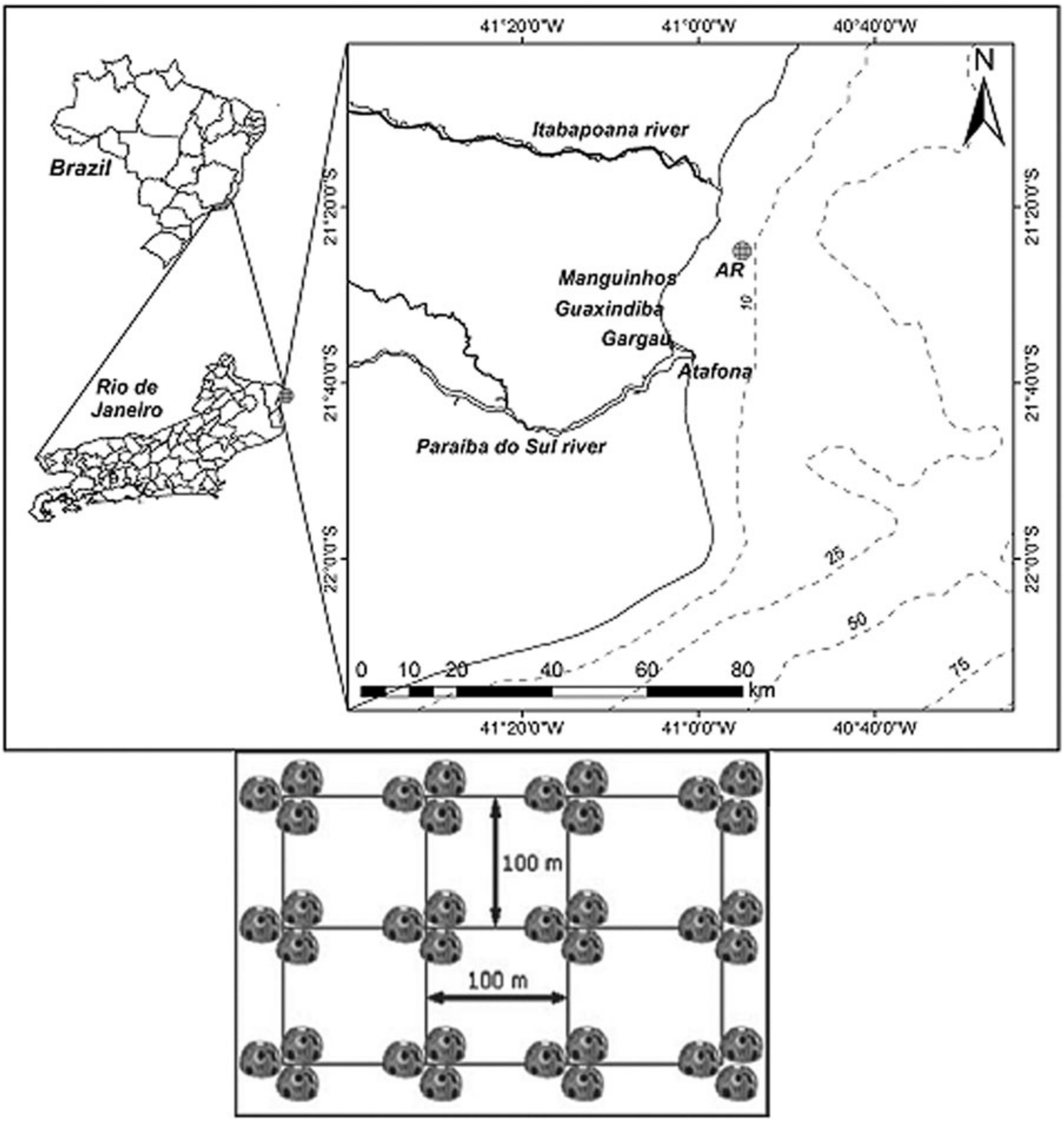

**Fig. 1.** Geographic location of the north coast of Rio de Janeiro (south-eastern Brazil), where the artificial reef complex (AR) was deployed. The spatial arrangement of the reef ball replicates and sets is also shown.

This is a weighted index that considers the per cent number of individuals (%$N$), biomass (%$B$) and relative frequency (%$F$) of each species (Zar, 1999), calculated as:

$$\mathrm{IP} = (\%N + \%B + \%F)/3.$$

The species association patterns in the reef complex and in the control areas were assessed by a cluster analysis (UPGMA) through a matrix of the transformed abundance data using the Bray–Curtis similarity index in the spatial (AR × NC × SC) and temporal (periods of higher and lower PSR discharge) dimensions (Clarke & Warwick, 2001).

The AR influence on the functional structure of the fish community, that is, on the prevalent life-history strategy of the main species was assessed by ABC curves. The degree of overlap in dominance between $r$-$K$ strategy species is measured with the $W$ statistic. Positive values indicate good environmental quality, which means that the curve patterns are within the expected variability for stable communities. Negative values indicate environmental disturbance, such that smaller values of $W$ indicate greater environmental stress (Clarke & Warwick, 2001).

In order to define the life strategy trend ($r$–$K$) of the main species several parameters were obtained from the literature (Appendix 1): (1) total average length at first sexual maturity, which corresponds to the length at which approximately 50% of the individuals in a population are able to reproduce ($L_m$); (2) maximum length recorded for the species ($L_{max}$); (3) average first maturity age ($T_m$), i.e. the age at which at least 50% of the individuals in a population are able to reproduce; (4) gestation time ($T_g$); (5) life expectancy; (6) growth coefficient ($K$), representing the mean annual growth rate of a species; (7) annual food consumption, corresponding to the quantity of food ingested divided by the biomass of the adult population; and (8) feeding habits and trophic level (TL), i.e. the hierarchical position occupied by a species in a trophic chain (Froese & Pauly, 2014).

The length-weight relationship (LWR) of the five species with highest IP contribution was calculated for each site (AR, NC and SC) using the logarithmic transformation of the equation presented by Froese (2006) as follows:

$$W = aL^b,$$

where $W$ is the fish weight (g), $L$ is the fish length (mm), $a$ is a constant and $b$ represents the exponential expressing relationship between length and weight. A Student's $t$-test was used to verify if the $b$ values were significantly different from 3 (isometric growth).

The condition factor ($K$) compared individuals from the same species in each site. The use of this coefficient is based on the assumption that heavier specimens of a given length are in better condition and, thus, have a higher $K$ (Froese, 2006). The Fulton's condition factor was calculated following Ricker (1975):

$$K = 100 \times (W/L^3),$$

where $W$ is the fish weight (g), $L$ is the fish length (cm) and 100 is a factor that approximate $K$ values near unity. Mean values of $K$ were compared with a Kruskal–Wallis non-parametric test. Since $K$ values may be influenced by allometric growth ($b \neq 3$), and can only be compared when related to similar length fishes (Froese, 2006), a relative condition factor ($K_{\text{rel}}$) was calculated according to Le Cren (1951):

$$K_{\text{rel}} = W/aL^b,$$

where $W$ (weight – g) and $L$ (length – cm) are the mean values observed for each species considering all individuals sampled, and $a$ (constant) and $b$ (linear coefficient) are the parameters of the LWR for each site.

Multivariate analysis and the ABC method were performed on Primer 6.0 statistical package. Non-parametric statistics were used when the normality distribution and homogeneity of variances were not observed (Fry, 1993). Statistical analysis was performed with BioEstat 5.0.1 and Statistica 8.0. A $P \leq 0.05$ was chosen to indicate statistical significance.

# RESULTS

The total sample consisted of 1014 individuals belonging to 51 species, with seven constant ($C > 50\%$), 10 accessory ($25\% < C \leq 50\%$) and 34 accidental species ($C > 25\%$). Of the 34 species identified in the AR, six were exclusive to the area. Five of the 32 species captured at the NC were exclusive, whereas nine of the 34 species captured at the SC were exclusive.

The largest number of individuals (413) was captured at the SC, followed by the AR (331) and the NC (270) (Figure 2A). Biomass results did not follow those of abundance, as the largest biomass value was recorded at the AR (42231.31g), followed by the SC (34824.79 g) and the NC (31173.22 g) (Figure 2B). The differences were significant between NC and the other sites (AR and SC) for both descriptors (Figure 2A, B). In the reef and the SC area, the number of individuals and the biomass captured during the lowest discharge period corresponded to 60–70% of the total number of fishes sampled. In the NC, the proportions of individuals captured in the different seasons were equivalent (Figure 2A, B).

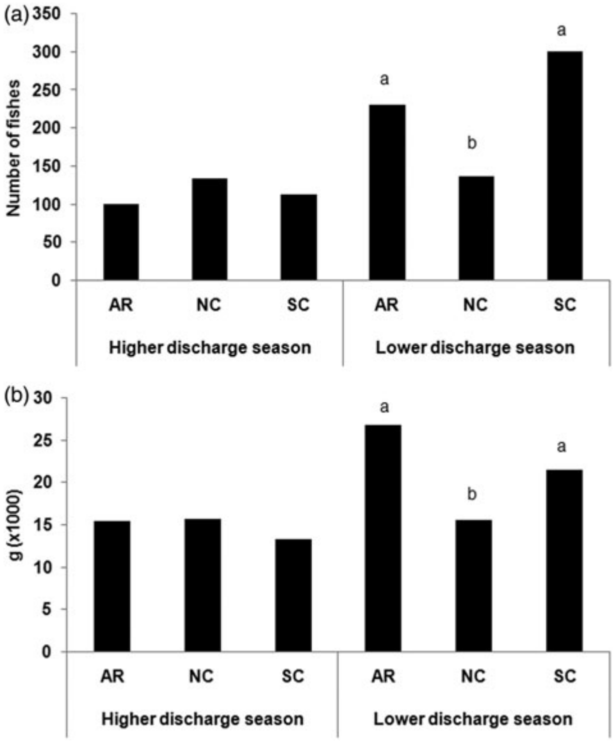

**Fig. 2.** Abundance (A) and biomass (B) of the fishes captured in the sampling areas (AR, artificial reef; NC, north control; SC, south control) during the higher and lower PSR discharge seasons. Different letters represent significant differences between areas ($P < 0.05$).

Species' temporal variation was characterized by differences in fish community composition during the different PSR discharge periods (Figure 3). Thirty-six species (15 exclusive) were captured during the largest PSR discharge period, whereas 39 species (10 exclusive) were identified during the lower discharge period. Considering the most frequent species, the Atlantic bumper (*Chloroscombrus chrysurus* Linnaeus, 1766), Guri catfish (*Aspistor luniscutis* Valenciennes, 1840), Guri sea catfish (*Genidens genidens* Cuvier, 1829), Caribbean sharpnose shark (*Rhizoprionodon porosus* Poey, 1861), Atlantic anchoveta (*Cetengraulis edentulous* Cuvier, 1829) and Bigtooth corvina (*Isopisthus parvipinnis* Cuvier, 1830) were predominant in the period of higher discharge, whereas Banded croaker (*Paralonchurus brasiliensis* Steindachner, 1875), Atlantic thread herring (*Opisthonema oglinum* Lesueur, 1818), American coastal pellona (*Pellona harroweri* Fowler, 1917), Guiana longfin herring (*Odontognathus mucronatus* Lacepède, 1800), King weakfish (*Macrodon ancylodon* Bloch & Schneider, 1801), Coco sea catfish (*Bagre bagre* Linnaeus, 1766), Shorthead drum (*Larimus breviceps* Cuvier, 1830), Spicule anchovy (*Anchoa spinifer* Valenciennes, 1848), Jamaica weakfish (*Cynoscion jamaicensis* Vaillant & Bocourt, 1883) and Rake stardrum (*Stellifer rastrifer* Jordan, 1889) were predominant during the lower discharge period. American harvestfish (*Peprilus paru* Linnaeus, 1758) did not show significant temporal variability (Figure 3).

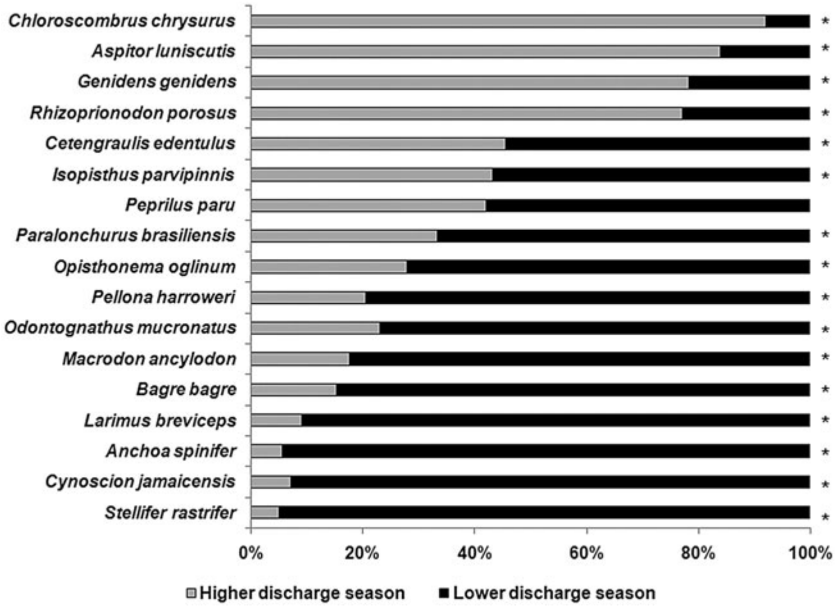

**Fig. 3.** Temporal distribution of the number of individuals (%) of constant and accessory species captured during the higher and lower discharge of the Paraíba do Sul River. $^*P < 0.05$: significant temporal variation.

The fish assemblages did not vary between sampling areas considering the IP of the main captured species (Table 1). Among the 15 most important species, 11 were common to the AR, SC and NC: *P. harroweri*, *A. luniscutis*, *O. oglinum*, *R. porosus*, *M. ancylodon*, *I. parvipinnis*, *L. breviceps*, *O. mucronatus*, *G. genidens*, *B. bagre* and *C. edentulus*.

Spatial variation of the most frequent species was characterized by the preferences of *P. brasiliensis* (~70%) for the SC, *C. chrysurus* for the NC (~75%) and *C. jamaicensis* and *P. paru* for the AR (~70 and ~72%, respectively). The other frequent species were common in the three areas, without significant differences (Figure 4).

The species association pattern revealed by the cluster analysis was mainly temporal with two groups, each including the three areas during the higher or lower PSR discharge period (Figure 5).

An analysis of the IP, the total average length at first sexual maturity ($L_m$) and the populations' estimated average length ($L_{med}$) during the higher PSR discharge period revealed that the AR contained two juvenile populations of species with *K*-strategy tendencies (*R. porosus* and *A. luniscutis*) and three species with *r*-strategy tendencies (*G. genidens*, *C. edentulus* and *O. oglinum*) (Table 2). In the NC, three important species were composed of juveniles of *K*-strategy species (*R. porosus*, *C. chrysurus* and *A. luniscutis*) and three species that are more likely to be *r*-strategists (*O. oglinum*, *O. mucronatus* and *G. genidens*). In the SC, the populations of *R. porosus* and *A. luniscutis*, both of which tend towards the *K*-strategy, were also composed mainly of juveniles (Table 2).

During the period of lower PSR discharge, the AR assemblage was dominated by juveniles of four species that tended towards *K*-strategy (*L. breviceps*, *M. ancylodon*, *A. luniscutis*

**Table 1.** Importance Percentage index (IP) of the constant and accessory species captured in each sampling area (AR, artificial reef; NC, north control; SC, south control).

| AR |  | NC |  | SC |  |
|---|---|---|---|---|---|
| Species | IP | Species | IP | Species | IP |
| *Pellona harroweri* | 40.91 | *Rhizoprionodon porosus* | 32.16 | *Pellona harroweri* | 38.27 |
| *Aspistor luniscutis* | 39.09 | *Pellona harroweri* | 31.99 | *Macrondon ancylodon* | 35.94 |
| *Ophistonema oglinum* | 32.74 | *Ophistonema oglinum* | 27.32 | *Rhizoprionodon porosus* | 28.26 |
| *Isopisthus parvipinnis* | 30.17 | *Macrodon ancylodon* | 26.37 | *Ophistonema oglinum* | 24.83 |
| *Rhizoprionodon porosus* | 28.38 | *Aspistor luniscutis* | 24.10 | *Paralonchurus brasiliensis* | 24.67 |
| *Macrodon ancylodon* | 25.39 | *Cynoscion jamaicensis* | 18.92 | *Odontognathus mucronatus* | 23.70 |
| *Larimus breviceps* | 24.75 | *Odontognathus mucronatus* | 18.22 | *Isopisthus parvipinnis* | 23.63 |
| *Peprilus paru* | 23.98 | *Genidens genidens* | 15.21 | *Aspistor luniscutis* | 20.12 |
| *Cynoscion jamaicensis* | 19.11 | *Chloroscombrus chrysurus* | 14.10 | *Cetengraulis edentulus* | 19.02 |
| *Odontognathus mucronatus* | 18.14 | *Larimus breviceps* | 13.47 | *Bagre bagre* | 18.62 |
| *Genidens genidens* | 18.12 | *Isopisthus parvipinnis* | 13.39 | *Stelifer rastrifer* | 18.03 |
| *Cetengraulis edentulus* | 17.98 | *Bagre bagre* | 12.48 | *Genidens genidens* | 15.60 |
| *Bagre bagre* | 13.15 | *Cetengraulis edentulus* | 12.25 | *Menticirrhus americanus* | 12.60 |
| *Trichiurus lepturus* | 9.07 | *Menticirrhus americanus* | 8.04 | *Larimus breviceps* | 12.27 |
| *Stelifer rastrifer* | 7.41 | *Anchovieta brevirostris* | 7.31 | *Conodon nobilis* | 11.80 |

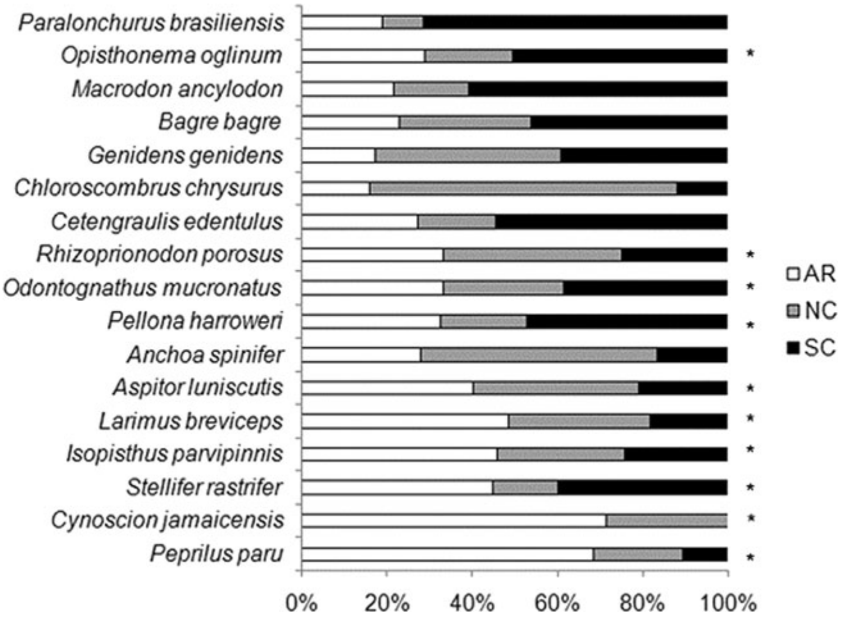

**Fig. 4.** Spatial distribution of the number of individuals (%) of constant and accessory species in the three sampling areas (AR, artificial reef; NC, north control; SC, south control). *$P < 0.05$: significant spatial variation.

and *P. paru*) and sexually mature individuals of three more species (Table 2) tending towards *r*-strategy. In the NC, two populations were considered juveniles and also comprised species that appear to be *K*-strategists, *A. luniscutis* and *B. bagre*, in addition to two populations that tend towards the *r*-strategy, *I. parvipinnis* and *C. jamaicensis*. In the SC, one population (*R. porosus*) was estimated to be essentially composed of young individuals with *K*-strategy characteristics, in addition to three species that tended towards the *r*-strategy, *P. brasiliensis*, *S. rastrifer* and *O. mucronatus* (Table 2).

The ABC curves for each area and sampling period show that during the higher PSR discharge period, there was no overlap between the abundance and biomass curves on the three areas, and the *W* value was positive (Figure 6A–C). During the lower PSR discharge period, there was a nearly complete overlap between the abundance and biomass curves in the AR, with a negative *W* value ($W = -0.167$) (Figure 6D). Conversely, both control areas exhibited a narrow overlap between the curves, with positive *W* values (Figure 6E, F).

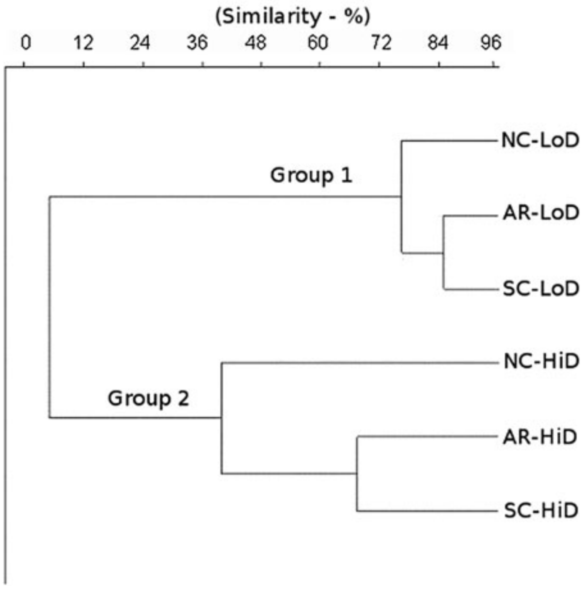

**Fig. 5.** Cluster analysis (UPGMA, Bray–Curtis similarity) through the species abundance matrix in the three sampling areas (AR, artificial reef; NC, north control; SC, south control) and during the higher (HiD) and lower (LoD) PSR discharge periods.

The length-weight relationship (LWR), condition factor (*K*) and relative condition factor ($K_{rel}$) results for the five species with highest IP contribution are summarized in Table 3. All length-weight regressions were significant ($P < 0.001$), with the coefficient of determination ranging from 0.636 for *A. luniscutis* (NC) to 0.978 for *O. oglinum* (NC). The parameter *b* was significantly different from 3 (*t*-test, $P < 0.01$), indicating an allometric growth for the five species on all the three sites. The mean condition factor (*K*) did not show significant differences (ANOVA, $P > 0.05$) between sites. The $K_{rel}$ values were higher in AR for four of the five species, but overall they were very similar to those found for the CTs (close to 1). The only exceptions were *M. ancylodon* (0.3) and *R. porosus* (0.4) in the NC and SC, respectively.

## DISCUSSION

This study shows the direct influence of artificial reefs on the functional structure of the transient ichthyofauna on the northern coast of Rio de Janeiro State. The fish assemblage of the reef complex area was distinct from those of control areas in terms of the age structure of the population and the life-history strategy used by the frequent species. This effect was most evident when the influence of the Paraíba do Sul River was minimal, during the period of lower discharge.

The presence of reef modules creates a more complex environment by offering a larger quantity of shelters (Brotto *et al.*, 2006a) in addition to concentrating a larger density of

**Table 2.** Importance Percentage index (IP), average length at first sexual maturity ($L_m$) (values from literature) and total estimated length ($L_{med}$) of the population of the seven most important species in each sampling area (AR, artificial reef; NC, north control; SC, south control) during the higher and lower outflow season of the Paraíba do Sul River-PSR.

| **Higher outflow season of the PSR** | | | | **Lower outflow season of the PSR** | | | |
|---|---|---|---|---|---|---|---|
| **Species** | **IP** | **$L_m$ (cm)** | **$L_{med}$ (cm)** | **Species** | **IP** | **$L_m$ (cm)** | **$L_{med}$ (cm)** |
| AR | | | | AR | | | |
| *R. porosus* | 47.24 | 70.0 | 40.3 | *P. harroweri* | 49.27 | 11.8 | 13.2 |
| *A. luniscutis* | 46.54 | 62.9 | 23.6 | *O. oglinum* | 39.51 | 15.5 | 18.6 |
| *P. harroweri* | 29.29 | 11.8 | 13.5 | *L. breviceps* | 36.77 | 19.0 | 16.5 |
| *I. parvipinnis* | 27.21 | 15.7 | 17.1 | *M. ancylodon* | 36.72 | 28.8 | 26.8 |
| *G. genidens* | 25.21 | 21.2 | 32.8 | *A. luniscutis* | 35.40 | 62.9 | 27.7 |
| *C. edentulus* | 25.11 | 10.6 | 21.3 | *P. paru* | 35.34 | 18.5 | 13.6 |
| *O. oglinum* | 24.57 | 15.5 | 19.0 | *C. jamaicensis* | 34.68 | 22.4 | 43.9 |
| NC | | | | NC | | | |
| *R. porosus* | 50.68 | 70.0 | 33.9 | *P. harroweri* | 39.63 | 11.8 | 13.4 |
| *O. oglinum* | 30.03 | 15.5 | 19.6 | *M. ancylodon* | 38.21 | 28.8 | 30.5 |
| *C. chrysurus* | 28.23 | 19.6 | 16.2 | *M. americanus* | 24.78 | 26.9 | 27.8 |
| *O. mucronatus* | 24.19 | 10.6 | 16.9 | *I. parvipinnis* | 24.38 | 15.7 | 19.8 |
| *A. luniscutis* | 22.90 | 62.9 | 23.7 | *C. jamaicensis* | 24.26 | 22.4 | 29.4 |
| *P. harroweri* | 17.68 | 11.8 | 13.4 | *A. luniscutis* | 23.93 | 62.9 | 29.3 |
| *G. genidens* | 14.72 | 21.2 | 24.5 | *B. bagre* | 23.56 | 36.1 | 30.8 |
| SC | | | | SC | | | |
| *R. porosus* | 39.59 | 70.0 | 40.5 | *P. harroweri* | 43.66 | 11.8 | 13.8 |
| *G. genidens* | 34.78 | 21.2 | 33.9 | *M. ancylodon* | 41.05 | 28.8 | 26.5 |
| *P. harroweri* | 31.88 | 11.8 | 13.6 | *O. oglinum* | 35.88 | 15.5 | 17.5 |
| *M. ancylodon* | 26.46 | 28.8 | 29.2 | *R. porosus* | 35.39 | 70.0 | 49.1 |
| *I. parvipinnis* | 24.88 | 15.7 | 20.5 | *P. brasiliensis* | 35.29 | 18.5 | 26.6 |
| *A. luniscutis* | 18.87 | 62.9 | 24.4 | *S. rastrifer* | 31.68 | 12.9 | 18.1 |
| *O. oglinum* | 16.33 | 15.5 | 20.4 | *O. mucronatus* | 24.94 | 10.6 | 17.0 |

potential prey both on the reef itself and in the surrounding area (Krohling *et al*., 2006). Artificial reefs tend to attract the adjacent substrate organisms that are important in the diet of piscivorous and/or invertivorous fish, suggesting that transient shoals of opportunist fish are directly affected by the biological productivity of the associated sediments (Lindquist *et al*., 1994; Relini *et al*., 2002; Zalmon *et al*., 2002; Leitão *et al*., 2007). Optimal foraging theory suggests that the less energy is expended during foraging, the smaller is the predation risk, as the organism remains exposed for less time (MacArthur & Pianka, 1966; Krebs *et al*., 1974). It is likely that transient, small-sized opportunistic fishes, such as *P. harroweri* and *O. oglinum*, or juveniles of species whose adults generally reach greater sizes, such as *C. jamaicensis*, *M. ancylodon* and *A. luniscutis*, feed on the prey that is closer to the reef.

Predation is suggested by Talbot *et al.* (1978) as one of the main factors that regulate ichthyic communities in isolated habitats associated with sandy substrates. This attraction effect regulated by feeding behaviour is also held to be responsible for attracting transient fishes in artificial reef habitats (Harding & Mann, 2001; Simonsen, 2008). The apparent rarity of strictly reef-associated species in our artificial reefs could be related to the sampling device, which is selective to transient species. One example that supports this explanation is the dominance of *Haemulon aurolineatum* (Cuvier, 1829) showed by Santos *et al.* (2010) in the same reef complex studied herein. This species, normally found in habitats such as reefs and coral patches, contributed to nearly 70% (408 individuals) of the total number of fishes observed through visual census, but not a single specimen was caught during our study. Still, it is important to emphasize that other strictly reef-associated species, such as snappers and groupers, were apparently rare in our artificial reefs even when a visual census was the sampling method (Brotto *et al*., 2006b, Santos *et al*., 2010).

Hackradt *et al.* (2011), also working in south-eastern Brazil, showed that the habitat complexity generated in artificial reefs by the shape and proximity of structures and the number of available cavities was directly correlated with the richness and abundance of certain species. The structure of the modules that constitute the reef complex in northern Rio de Janeiro also encompasses a variety of configurations, with different numbers of artificial structures and distances between modules and cavities used for shelter. This arrangement has generated a complex environment relative to the adjacent areas, which are covered by sandy and homogeneous substrate. The resulting complexity was tested in the same area by Brotto *et al.* (2006a), who verified that more complex reefs, with a larger number of cavities and a greater availability of encrusting prey (tested by the presence/absence of anti-encrusting paint on the modules), attracted a larger number of species and individuals, leading to increased diversity of transient fish in more complex environments.

The main difference between the AR and CTs in terms of community structure indicators was related to biomass, with superior values in the AR for both sampling periods (higher and lower PSR outflow season). These data show that, with the exception of biomass, community structure does not differ between the areas, and, therefore, these analyses by themselves would underestimate the influence of the reef complex on the transient opportunist ichthyofauna. When considering the functional structure of the fish assemblage, represented by the life strategies attributed to each species

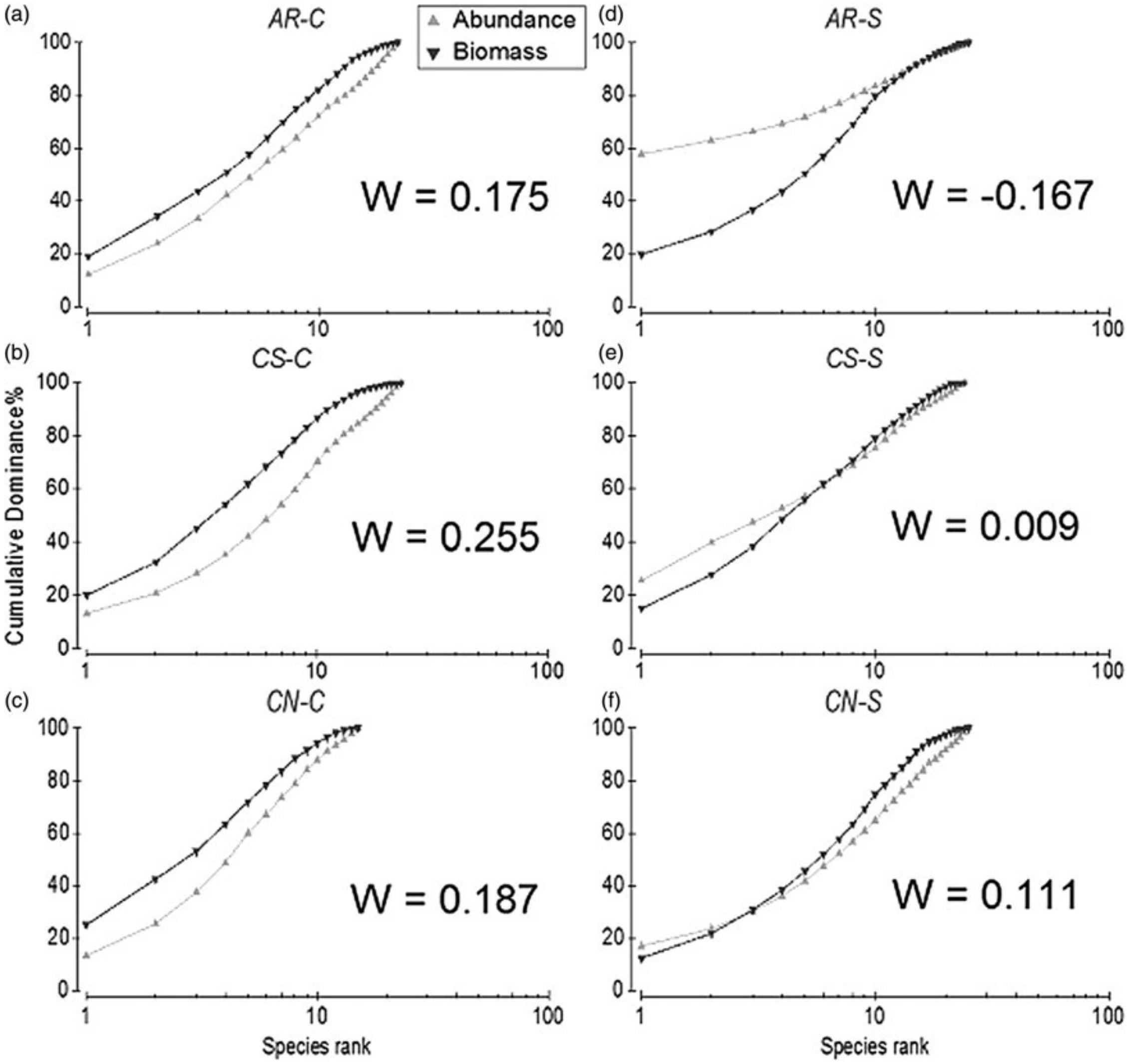

**Fig. 6.** Dominance curves related to the abundance and biomass of the species captured during the sampling periods: the higher PSR discharge period (A) in the artificial reef – AR; (B) south control – SC; and (C) north control – NC (C); and during the lower PSR discharge period (D) in the AR, (E) SC and (F) NC.

**Table 3.** Total length (mean ± SD, minimum and maximum values), length-weight relationship, Fulton's condition factor ($K \pm$ SD) and relative condition factor ($K_{rel}$) for the five species with the higher importance percentage index (IP) on the three studied areas (AR, artificial reef; NC, north control; SC, south control).

| **Species** | | **Total Length (cm)** | | | **Length-Weight Relationship (LWR)** | | | | **Condition Factor (*K*)** | |
|---|---|---|---|---|---|---|---|---|---|---|
| | ***N*** | **Mean ± SD** | **Min** | **Max** | ***a*** | ***b*** | **SE (*b*)** | $R^2$ | $K \pm$ **SD** | $K_{rel}$ |
| *Aspistor luniscutis* (AR) | 24 | 24.4 ± 4.9 | 14.7 | 33.7 | 0.016 | 2.828* | 0.169 | 0.924 | 0.94 ± 0.14 | 1.3 |
| *Aspistor luniscutis* (NC) | 24 | 24.4 ± 5.4 | 15.5 | 36.6 | 0.032 | 2.658* | 0.431 | 0.636 | 1.33 ± 1.55 | 1.1 |
| *Aspitor luniscutis* (SC) | 13 | 24.4 ± 5.9 | 17.0 | 40.7 | 0.023 | 2.747* | 0.196 | 0.942 | 1.04 ± 0.19 | 1.2 |
| *Macrodon ancylodon* (AR) | 13 | 28.4 ± 3.7 | 21.3 | 34.6 | 0.024 | 2.722* | 0.240 | 0.833 | 0.97 ± 0.16 | 1.1 |
| *Macrodon ancylodon* (NC) | 9 | 31.4 ± 2.7 | 26.8 | 36.6 | 0.004 | 3.277* | 0.421 | 0.879 | 1.02 ± 0.10 | 1.0 |
| *Macrodon ancylodon* (SC) | 21 | 27.1 ± 4.0 | 17.6 | 36.4 | 0.001 | 4.034* | 0.378 | 0.941 | 0.86 ± 0.18 | 0.3 |
| *Opisthonema oglinum* (AR) | 30 | 18.5 ± 3.0 | 14.4 | 25.8 | 0.006 | 3.094* | 0.032 | 0.969 | 0.81 ± 0.07 | 1.2 |
| *Opisthonema oglinum* (NC) | 24 | 19.8 ± 3.5 | 14.5 | 31.9 | 0.007 | 3.056* | 0.089 | 0.978 | 0.84 ± 0.06 | 1.1 |
| *Opisthonema oglinum* (SC) | 60 | 17.9 ± 2.8 | 13.6 | 26.7 | 0.007 | 3.024* | 0.066 | 0.882 | 0.84 ± 0.20 | 1.2 |
| *Pellona harroweri* (AR) | 104 | 13.2 ± 1.0 | 10.5 | 15.8 | 0.023 | 2.659* | 0.022 | 0.835 | 0.97 ± 0.09 | 1.0 |
| *Pellona harroweri* (NC) | 41 | 13.5 ± 0.7 | 12.0 | 15.2 | 0.047 | 2.382* | 0.047 | 0.672 | 0.94 ± 0.09 | 1.0 |
| *Pellona harroweri* (SC) | 141 | 13.1 ± 0.7 | 11.4 | 15.6 | 0.012 | 2.908* | 0.028 | 0.678 | 0.95 ± 0.10 | 1.0 |
| *Rhizoprionodon porosus* (AR) | 16 | 43.5 ± 5.5 | 36.5 | 58.0 | 0.001 | 3.339* | 0.209 | 0.967 | 0.41 ± 0.03 | 1.3 |
| *Rhizoprionodon porosus* (NC) | 20 | 42.8 ± 3.6 | 33.7 | 51.6 | 0.001 | 3.606* | 0.264 | 0.877 | 0.95 ± 0.10 | 0.4 |
| *Rhizoprionodon porosus* (SC) | 12 | 44.8 ± 7.9 | 25.4 | 56.0 | 0.060 | 2.314* | 0.448 | 0.790 | 0.95 ± 0.10 | 1.0 |

*Significantly different from 3 ($P < 0.001$).

or by the age structure of the most frequent populations (Appendix 1), proportionally greater numbers of *r*-strategist species and predominantly juvenile populations of *K*-strategist species were observed during the lower influence period of the Paraíba do Sul River on the reef area.

In addition, the ABC curve showed that *r*-strategist species used the reef complex more efficiently than the control areas, particularly during the lower PSR discharge period. These potentially opportunistic species are generally smaller than *K*-strategist species and fall at the base of the trophic chain as primary or secondary consumers (Pianka, 1970). Therefore, these species sustain the populations of larger species and form an essential link in the equilibrium of fish communities (King & McFarlane, 2003). In the AR, there was a nearly complete overlap between the abundance and biomass curves, with a negative $W$ value, indicating a potential disturbance in the functional structure of the fish community, especially during the lower PSR discharge period. In the CTs, the ABC curves revealed the patterns expected for functionally stable communities under the $r/K$ selection theory ($W \geq 0$) (Clarke & Warwick, 2001).

The LWR parameters observed for *A. luniscutis*, *M. ancylodon* and *P. harroweri* showed that the specimens caught in each area were mostly juveniles. These results suggest that there were no differences in the LWR parameters between sites, but also that the AR is not attracting adult individuals. The fact that all species in the three areas showed an allometric growth ($b \neq 3$) highlights the importance of the use of a relative condition factor as discussed in Froese (2006). The similar values of $K_{rel}$ for each species on the sites corroborates the LWR results and ABC curves, and shows no difference in the condition or well-being of the fishes theoretically promoted by the AR.

The results highlight the importance of evaluating the life-history traits of individual species in addition to community characteristics such as richness, abundance and descriptive analyses of the ichthyofauna to achieve a real assessment of the influence of artificial reefs on the associated fish. However, the influence of the Paraiba do Sul river seemed to have prevailed over the effects of the reef complex, most likely due to the biology of species at higher or intermediate trophic levels, such as *A. luniscutis*, which lives in more turbid waters and migrates to the river mouth for spawning during the lower PSR discharge period, and *R. porosus*. This latter species is more often found in coastal areas close to estuaries, where water mixing and the increase in nutrient concentration make this environment favourable to the shark's prey, e.g. various members of the family Sciaenidae (*I. parvipinnis*, *C. jamaicensis*, *M. ancylodon*, *L. breviceps* and *P. brasiliensis*). These prey live close to rivers and are predominant in the region during the entire year (Gomes *et al.*, 2003; Fulgêncio, 2004; Souza & Chaves, 2007; Militelli *et al.*, 2013), with young and adult individuals using estuarine and shallow-water areas for growth and feeding (Menezes & Figueiredo, 1980; Godefroid *et al.*, 2004). These species are found in beach environments in the periods of reproduction and recruitment, which occur from spring to autumn (Godefroid *et al.*, 2004). This migratory behaviour of the Sciaenid family justifies their predominance in the artificial reefs during the lower-discharge period of PSR.

This study emphasizes the reef complex usage by the local transient fish especially during the lower PSR discharge period. However, it is still unclear whether the fishes use the reef for food or shelter. Species like *A. luniscutis*, *M. ancylodon* and *L. breviceps* are important fishing resources considered to be at the top of the trophic web (Soares & Vazzoler, 2001; Carneiro & Castro, 2005). In the reef area, their populations were mainly composed of juveniles during the lower-discharge period. The attraction exerted by the artificial reefs on local transient fish reinforces the potential role these reefs have in harbouring juveniles of overexploited species, as those mentioned above. The reef's attraction of juveniles of *A. luniscutis*, *R. porosus* and *M. ancylodon*, together with the attraction of *r*-strategist species support the community and allow energy transfer along the trophic chain.

Gillnets were chosen because they are the main gear used by local fishermen and to allow a direct comparison to previous research in the same area (Santos *et al.*, 2010; Gatts *et al.*, 2014). But, like any fishing gear, gillnets have a bias associated with their selectivity (King, 2007). Local artisanal fishing activities often target larger individuals and fishing nets are consequently selective for larger fish; thus the effect of the reef complex is likely to be even greater than observed herein. However this possibility does not weakens the hypothesis that the populations associated with the studied reef complex are mainly composed of young individuals and/or smaller-sized species than the surrounding areas.

In summary, variation in community composition was observed along the temporal, but not on the spatial (AR × CTs) dimension. The association pattern of the transient species revealed a seasonal effect, illustrating the potential effect of the Paraíba do Sul River on the artificial reef and control areas, resulting in a temporal distribution pattern of the main species and masking the spatial differences. As observed by Brotto & Zalmon (2007), adverse environmental conditions (for example, strong bottom currents, turbid waters and the presence of a polyhaline plume) are most likely the key factors affecting the fish colonization patterns in the north coast of Rio de Janeiro State (Krohling & Zalmon, 2008; Santos *et al.*, 2010). Therefore, this factor should be considered in the implementation of artificial reefs in typically seasonal regions, such as those under strong influence of fluvial discharges, aiming for the management of the transient ichthyofauna.

However, the control areas were considered more stable than the reef area, as the reef community was primarily composed of *r*-strategist species and/or juveniles, especially during the lower Paraíba do Sul River discharge period, indicating a less stable environment. This 'instability' warrants a positive connotation, as it means that the artificial reefs are harbouring individuals that are more susceptible to predation and also that the reefs therefore represent an important tool for maintaining these populations at least on a local scale. In this way, artificial reefs have the potential to promote the more efficient management and conservation plans of the artisanal fishery on the northern coast of Rio de Janeiro. Finally, when evaluating the influence of an artificial reef, it is important that researchers expand their focus beyond a snapshot of the fish community obtained from richness, abundance, biomass, species composition and descriptive indices. Specific patterns must also be measured, including age and size classes of the main fish populations in the community, as these populations may be critical for the maintenance of the ecosystem; even so, their importance is often underestimated.

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# APPENDIX

**Appendix 1.** Life history parameters of the constant and accessory species ($C > 25\%$): species code, total length at first sexual maturity ($L_m$), maximum total length ($L_{max}$), age of first maturity ($T_m$), time generation ($T_g$), life expectancy, growth coefficient ($K$), annual consumption and trophic level (TL).

| Species | $L_m$ (cm) | $L_{max}$ (cm) | $T_m$ (years) | $T_g$ | Life expectancy | K (/ano) | Food intake (x/ano) | TL |
|---|---|---|---|---|---|---|---|---|
| *Macrodon ancylodon* | 70.0 | 110 | 2.9 | 3.8 | 9.3 | 0.30 | 26.4 | 4.1 |
| *Pellona harroweri* | 62.9 | 120 | 1.2 | 1.9 | 5.8 | 0.50 | 10.7 | – |
| *Aspitor luniscutis* | 31.6 | 55.0 | 3.7 | 4.7 | 15.9 | 0.18 | 6.5 | 4.0 |
| *Opisthonema oglinum* | 24.8 | 45.0 | 2.6 | 3.1 | 10.6 | 0.27 | 6.9 | 3.9 |
| *Rhizoprionodon porosus* | 22.4 | 50.0 | 1.8 | 2.0 | 7.1 | 0.40 | 7.3 | 4.2 |
| *Isopisthus parvipinnis* | 21.2 | 35.0 | 2.5 | 2.9 | 10.2 | 0.28 | 12.9 | – |
| *Odontognathus mucronatus* | 19.6 | 65.0 | 1.3 | 1.4 | 5.1 | 0.56 | 13.8 | 3.2 |
| *Cetengraulis edentulus* | 19.0 | 31.0 | 1.3 | 1.5 | 5.3 | 0.54 | 8.7 | 3.1 |
| *Larimus breviceps* | 18.5 | 30.0 | 1.0 | 1.1 | 3.8 | 0.75 | 5.5 | 4.0 |
| *Paralonchurus brasiliensis* | 18.5 | 30.0 | 1.3 | 1.4 | 5.2 | 0.55 | 9.3 | 3.1 |
| *Bagre bagre* | 15.7 | 36.0 | 1.5 | 1.7 | 6.1 | 0.47 | 8.5 | 4.2 |
| *Genidens genidens* | 15.5 | 38.0 | 1.1 | 1.2 | 4.4 | 0.65 | 11.5 | 3.1 |
| *Peprilus paru* | 15.2 | 24.0 | 1.2 | 1.2 | 4.5 | 0.63 | 10.8 | 4.1 |
| *Anchoa spinifer* | 12.9 | 20.0 | 1.3 | 1.0 | 3.5 | 0.80 | 11.4 | 3.5 |
| *Stellifer rastrifer* | 11.8 | 18.0 | 0.6 | 0.9 | 2.3 | 1.25 | 13.1 | 3.2 |
| *Chloroscombrus chrysurus* | 10.6 | 16.0 | 1.0 | 1.3 | 3.4 | 0.84 | 37.1 | – |
| *Cynoscion jamaicensis* | 10.6 | 17.1 | 0.8 | 1.1 | 2.7 | 1.05 | 40.9 | 2.1 |

Values extracted from Froese & Pauly (2014).