



Effects of a nuclear power plant thermal discharge on habitat complexity and fish community structure in Ilha Grande Bay, Brazil

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ABSTRACT

Fish communities and habitat structures were evaluated by underwater visual censuses a rocky location impacted by thermal discharge (I) and at two control locations, one in a *Sargassum* bed (C1) and the other in a rocky shore with higher structural complexity (C2). Habitat indicators and fish communities exhibited significant differences between the impacted and control locations, with the impacted one showing a significant decrease in fish species richness and diversity, as well as a decrease in benthic cover. At the I location, only 13 fish species were described, and the average water temperature was 32 ± 0.4 °C, compared with 44 species at C1 (25.9 ± 0.3 °C) and 33 species at C2 (24.6 ± 0.2 °C). Significant differences in fish communities among locations were found by ANOSIM with *Eucinostomus argenteus*, *Mugil* sp. and *Haemulon steindachneri* typical of location I, while *Abudefduf saxatilis*, *Stegastes fuscus* and *Malacoctenus delalandi* were typical of the control locations. Our study shows that thermal pollution alters benthic cover and influences fish assemblages by altering composition and decreasing richness.

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1. Introduction

Fish assemblages are widely used in applied ecology to detect human impact on marine environments (Warwick and Clarke, 1993; Contador, 2005). Some variations in species assemblages can allow early detection and monitoring of human impacts on natural ecosystems.

Temperature is a crucial environmental factor affecting marine organisms and ecosystems. It affects the distribution of populations on both small and large geographical scales (Wilson, 1981), and determines the structure of communities and ecosystems (Glynn, 1988) by affecting the physiological processes and behavior of fish species (Dembski et al., 2006). Reefs and marine ecosystems around the world are exposed to the effects of thermal phenomena such as global warming, El Niño and localized thermal pollutants (Forchhammer et al., 2000). Heated effluents introduced on the marine environment may induce dramatic and unpredictable effects, depending on the amount and temperature of discharged material, as well as the climatic, hydrological and biological features of the local environment (Lardicci et al., 1999). Fish are mobile, and most can migrate to safe areas when chronic low levels of heat pollution. However, many of their food sources (i.e. corals, sponges, macroalgae, etc.) are sessile, and may be adversely affected. Apart from simply leaving an area, reef fish may

be indirectly affected by the decreasing quality of food resources. Given the intensity and frequency of human disturbance and the ecological importance of coastal areas, it is critically important to understand the different aspects of the thermal effect on tropical rock shore organisms. Although there have been numerous studies performed on the impacts of heat pollution on coral reef environments (Roberts and Ormond, 1987; Chabanet et al., 1997; Öhman and Rajasuriya, 1998), little is known about the impact of thermal modification on tropical rocky shores.

Changes in water temperature caused by power station thermal discharge affect fish assemblages by decreasing species richness (Rong-Quen et al., 2001). Furthermore, rocky shores have a variety of microhabitats, which increase fish diversity (Luckhurst and Luckhurst, 1978). Thermally polluted rocky substrate may be unable to support sessile invertebrates or microalgae vegetation and will have a negative impact on fish using the habitat for shelter, food, nesting and juvenile settlement. A decrease in habitat complexity can also decrease species richness.

The southeastern coast of Brazil does not support any true coral reefs and rocky shores are the main habitat for reef fish and reef-associated biota (Floeter et al., 2007). The Ilha Grande Bay, a relatively well-preserved coastal system, is exposed to the thermal discharge of the Brazilian Nuclear Power Plant (BNPP), and provides a unique opportunity to evaluate thermal effects on the fish community. The current study will contribute to an early assessment of the global warming and some specific changes in the rocky habitat structure.

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In the present study, tropical rock reef fish were sampled in a location exposed to thermal discharge and at two controls areas of natural thermal conditions in order to evaluate the effect of thermal pollution on habitat structure of local fish assemblages. All sites have similar dimensions, type of substrate and depth. It was hypothesized that thermal pollution alters habitat structure changing fish assemblages. We focused on two specific research goals: to find out whether the cooling power station alters the habitat structure, and whether fish assemblages in this location shift in composition and richness, due to the thermal effect. In addition, we determined whether both control locations varied in structural complexity and, consequently, in fish assemblage composition.

2. Material and methods

2.1. Study area

The study was conducted at a rocky shore in Ilha Grande Bay, close to the discharge water of the Brazilian Nuclear Power Plant (BNPP), the only one in Brazil. The plant began commercial operation in 1985 and consists of two sections, producing 600 and 1300 MW, respectively, and the cooling water flow discharge into the sea is 40 and $80 \text{ m}^3 \text{ s}^{-1}$, respectively, from each of the sections. Three study locations were chosen for sampling near BNPP (Fig. 1).

The mean temperature of the seawater surface ranged from 29.5°C in winter to 36.5°C in summer at a distance of approximately 100 m from the outfall in the impacted location (I). The thermal effluent produced a mean increase of 8°C in the discharge area (Bandeira et al., 2003). Two control locations (C1 and C2) with similar depth (1 – 2.5 m) were chosen at approximately 10 km (C1) and 9 km (C2) from the impacted location (I), following the coast-

line. Surface temperature was measured to be 23°C in winter to 28°C in summer. Salinity was found to be $34.5 (\pm 1.2 \text{ SE})$ in all locations, consistent throughout the year.

2.2. Sampling design and methods

Our sampling methods allowed an in-depth comparison of the rock reef fish community features, and their relationship with local habitat structures, both at a location close to the BNPP effluent outlet, and at two undisturbed control locations. Monthly surveys were conducted between October 2005 and July 2006. The impacted location was chosen on the basis of high temperature (a mean value of 7°C higher than the control locations).

2.3. Habitat structure

The study area is characterized by rocky shores covered by granitic boulders and a sandy bottom. All three study locations (C1, C2 and I) are typically rocky shores. The first area is characterized by *Sargassum* beds (C1), the second exhibits a comparatively more complex habitat structure (C2), and the third is impacted by heated effluent discharge (I), with the presence of rock rubble in some places.

Quadrants of 2 m^2 were used to quantify the variables describing physical structure. The crevices in each quadrant were counted and pooled into three size classes (Ca $\leq 30 \text{ cm}$; Cb = 30 cm – 1 m ; Cc $\geq 1 \text{ m}$), as well as the rocks, which were also counted and pooled into four size classes (Ra $\leq 30 \text{ cm}$; Rb = 30 cm – 1 m ; Rc = 1 m – 3 m ; Rd $\geq 3 \text{ m}$). In each area, a visual census estimated the percent benthic cover of the *Sargassum* (Phaeophyta, Fucales), other Phaeophyta algae, *Palythoa* (a genus of colonial cnidarians, Order Zoanthidea), encrusting calcareous algae, branched calcareous algae, sponges, barnacles, vermetids (genus *Petalocochus*), bare rocks, sand and shell-sand patches.

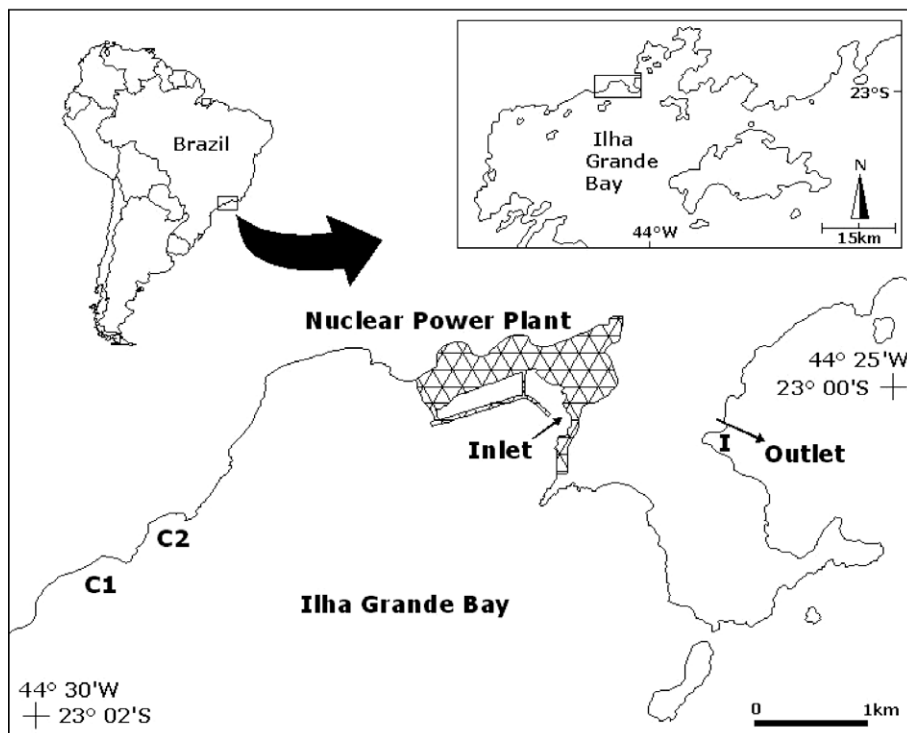


Fig. 1. Map of the study area in Ilha Grande Bay, Brazil: thermal impacted site (I) and controls (C1 and C2) locations.

2.4. Rocky fish

The composition and relative abundance of fish species were surveyed using the non-destructive visual census diving transect technique adapted from Sale (1997). The technique was selected as the most appropriate for surveying both smaller specimens and fast-swimming species (Harmelin-Vivien and Francour, 1992). Underwater visual censuses were performed by SCUBA diving and snorkeling during a census parallel to the coastline along transects 30 m long and 3 m wide. Transects were each observed twice. The first time, the observer recorded the conspicuous species, and the second time focused the search beneath rocks and in all crevices to observe the more cryptic species (Aburto-Opereza and Balart, 2001). Surveys were carried out in shallow waters (0.5–2.5 m) because benthic communities in these areas are generally primary producers with relevant secondary production, and they maintain the richest ichthyofauna (Bouchereau and Lam Hoai, 1997; Methven et al., 2001). Three repeated censuses were performed at each of the three locations during the ten field trips, yielding a total of 90 samples.

2.5. Data analysis

Patterns of fish data were assessed by non-metric multidimensional scaling (nMDS) using the Bray-Curtis measure of similarity. A fourth root transformation (Field et al., 1982) was performed on the abundance data prior to the analyses. Differences in fish communities between locations were performed using “Analysis of Similarities” (ANOSIM), and the “Similarity of Percentage” procedure (SIMPER) was employed to identify the contribution of each species to differences between sites (Clarke, 1993). The Shannon-Wiener (H') index described differences in the heterogeneity of each community. These analyses were performed with the software package PRIMER (Plymouth Routines Multivariate Ecological Research) (Clarke and Warwick, 1994).

The mean value and the standard error of each habitat descriptor were calculated for each location. Differences in habitat descriptors were compared among the three locations using one-way analysis of variance (ANOVA) and a *post-hoc* multiple comparisons Tukey HSD test. A principal component analysis (PCA; correlation matrix) on the log-transformed ($x + 1$) environmental variables matrix was performed to detect any spatial pattern. For

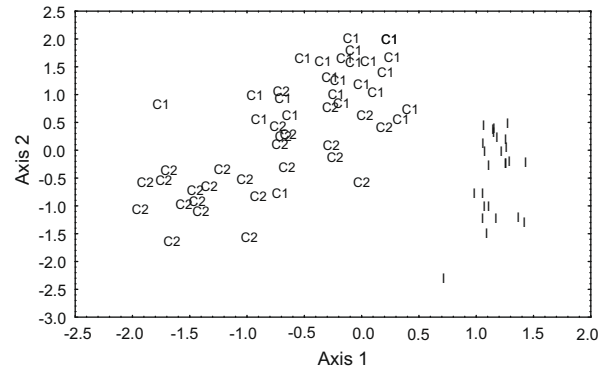


Fig. 2. Scores on the first two axes from PCA on habitat descriptors in Ilha Grande Bay. Samples code: I, Impacted location; C1 and C2, control locations.

this later analysis, only descriptors that show differences between locations were used. ANOVA and PCA were conducted using Statistica 7.1 software (StatSoft, Inc., 2005).

3. Results

3.1. Habitat complexity

The impacted location (I) consisted mainly of bare rock, rock rubble and branched calcareous algae. Rock rubble and branched calcareous algae were found exclusively in this habitat. The C1 location was characterized by coverage of *Sargassum*, encrusting calcareous algae and sponge colonies, while C2 consisted mainly of crevices and rocks, with coverage by *Palythoa* and several types of Phaeophyta algae (Table 1).

The results of the PCA analysis show the relation between geographical locations and habitat descriptors (Fig. 2). The PCA of habitat descriptors is shown in Table 2, with the two first axes accounting for 43% of the total variance. The first axis summarizes thermal influence on benthic cover between locations. In C2, the density of Phaeophyta and encrusting calcareous algae was highest, while opportunistic branched calcareous algae and bare rocks were rare. In location I, this situation was reversed (Fig. 2). Locations I and C2 were separated along the axis 1.

Table 1
Mean values (\pm standard error) and *F*-values from ANOVA comparisons of the habitat descriptors measured in 26 replicates of 2 m² quadrants among locations (I, impacted; C1 and C2, controls) at Ilha Grande Bay (–, absence of descriptor).

Habitat descriptors	C1	C2	I	<i>F</i>	Tukey <i>post-hoc</i>
<i>Physical structure (number)</i>					
Crevices < 30 cm	0.77 \pm 0.34	10.15 \pm 1.57	1.69 \pm 0.90	33.15 ^b	C2 > C1, I
Crevices 30–1 m	0.08 \pm 0.08	0.92 \pm 0.33	0.85 \pm 0.30	3.8 ^a	C2, I > C1
Crevices > 1 m	0.08 \pm 0.08	0.23 \pm 0.17	–	n.s.	
Rocks < 30 cm	10.08 \pm 5.11	9.23 \pm 2.72	59.46 \pm 20.84	n.s.	
Rocks 30 cm–1 m	4.38 \pm 1.08	5.77 \pm 1.35	1.92 \pm 0.61	4.02 ^a	C1, C2 > I
Rocks 1–3 m	0.46 \pm 0.18	0.69 \pm 0.31	2.00 \pm 0.80	n.s.	
Rocks > 3 m	0.08 \pm 0.08	0.23 \pm 0.12	–	n.s.	
<i>Benthic cover (%)</i>					
Bare rock	12.00 \pm 3.93	1.35 \pm 0.82	31.35 \pm 5.91	20.48 ^b	I > C1 > C2
Sand	8.27 \pm 3.69	18.27 \pm 7.34	3.27 \pm 1.64	n.s.	
<i>Sargassum</i>	53.27 \pm 8.68	11.65 \pm 5.15	–	31.55 ^b	C1 > C2
<i>Palythoa</i>	6.54 \pm 3.86	17.19 \pm 5.86	–	6.19 ^a	C2, C1 > I
Encrusting calcareous algae	8.08 \pm 2.52	9.23 \pm 2.80	–	9.49 ^b	C1, C2 > I
Phaeophyta algae	9.62 \pm 3.52	38.35 \pm 6.76	–	34.25 ^b	C2 > C1 > I
Branched calcareous algae	–	–	51.15 \pm 6.61	122.0 ^b	I > C1, C2
Sponge	1.65 \pm 0.59	1.08 \pm 0.77	–	3.63 ^a	C1, C2 > I
Barnacle	–	2.12 \pm 1.60	1.15 \pm 0.54	n.s.	
Vermetid	–	–	13.08 \pm 4.68	10.80 ^b	I > C1, C2

Significance level: ^a*p* < 0.05; ^b*p* < 0.01; n.s. – non significant.

Table 2

Results of Principal Components Analysis based on habitat descriptors at the studied sites in Ilha Grande Bay. Values higher than 0.6 are indicated in bold.

Variables	Axis 1	Axis 2
Crevices < 30 cm	-0.573	-0.594
Crevices 30 cm–1 m	0.029	-0.393
Rocks 30 cm–1 m	-0.424	-0.078
Bare rock	0.650	0.047
<i>Sargassum</i>	-0.121	0.805
<i>Palythoa</i>	-0.464	-0.352
Encrusting calcareous algae	- 0.605	0.081
Phaeophyta algae	- 0.745	-0.193
Branched calcareous algae	0.716	-0.383
Sponge	-0.403	0.170
Vermetid	0.427	-0.494
Eigen value	2.944	1.753
% Variance explained	26.7	15.9

The second PCA axis corresponded to the influence of the *Sargassum* bed on structural complexity (Fig. 2). The sample scores for the second axis were positively correlated with *Sargassum* (little physical structure), and negatively correlated with crevices, *Palythoa* and other types of Phaeophyta algae (considerable physical structure). Locations I and C2 were separated from C1 along this axis.

3.2. Fish community

A total of 44 species and 31 families were recorded throughout the study period (Table 3). Mugilids were not identified at species level due to the difficulty of specific determination by visual observation. Five families showed a high prevalence of species: Chaetodontidae, Pomacentridae and Haemulidae, Blenniidae and Labrisomidae. The twelve most abundant species accounted for 94% of all the fish, each one contributing more than 0.1% of relative abundance: *Eucinostomus argenteus*, *Abudefduf saxatilis*, *Haemulon steindachneri*, *Sardinella janeiro*, *Diplodus argenteus*, *Mugil* sp., *Stegastes fuscus*, *Orthopristis rubber*, *Malacocteus delalandi*, *Chaetodon striatus*, *Pempheris schomburgkii*, *Sphoeroides greeleyi*. The most abundant species in C1 was *S. janeiro*, in C2 was *A. saxatilis*, and in I was *E. argenteus*.

3.3. Comparison of fish between the studied locations

A significant decrease in the mean species richness was described at site I compared to the control locations ($F = 45.83^{**}$, $p < 0.05$; Fig. 3), although no difference was found in total fish abundance between the three locations. A greater number of species was recorded at the two control sites (C1 = 40; C2 = 33) compared to I (13). These differences were due to 18 fish that were only observed at the control sites. Eleven species were common to the three locations and no species were exclusive to site I.

The Shannon–Wiener diversity index showed significant differences between I and the controls ($F = 19.82^{**}$, $p < 0.05$). This trend was inversely related to temperature (Fig. 4).

ANOSIM showed significant differences ($R = 0.592$, $p < 0.01$) between locations (Table 4). *Post-hoc* pairwise comparisons exposed significant differences between I and the controls (C1 and C2), although the two controls did not significantly differ from one another. The MDS configuration of individual replicates (Fig. 5) shows that the fish community in site I is clearly distinct from those recorded at the two controls, with the latter grouping indistinguishably. SIMPER provided evidence that *E. argenteus* and *A. saxatilis* mostly contributed to the dissimilarity between each control and I (Table 4).

SIMPER analysis identified the species that made the greatest contribution to the fish composition similarities within the locations (Table 5). Within C1, Pomacentridae *S. fuscus* (12%), *A. saxatilis* (17.8%) and Labrisomidae *M. delalandi* (16.8%) were responsible for most of the similarity within this location. In C2, *A. saxatilis* (25%), *S. fuscus* (26%) made the greatest contribution to the average similarity. In total, these two Pomacentrids accounted for over 30% of the average similarity for C1, and 51% for C2. The most prominent species in I were *E. argenteus* and *Mugil* sp., which contributed 42.71% and 19.3%, respectively to the average similarity. These species were nearly absent in the control locations. Another important species in this analysis was *H. steindachneri*, which contributed to average similarity for all locations, particularly in site I.

4. Discussion

4.1. Habitat complexity

This study provides compelling evidence that heated effluents can affect habitat structure, and consequently, community structure and spatial distribution. The heat-impacted location showed low benthic cover and was characterized mainly by bare rock. In addition, vermetids were found only at I, indicating that these organisms are opportunistic and tolerant to high temperatures ($>30^{\circ}\text{C}$). Thermal discharge has been shown to reduce species richness of algal and zoobenthic communities, to increase the abundance of the so-called opportunistic or ephemeral species and to alter the population dynamics of the most abundant species (Deviny, 1980; Mahadevan, 1980; Verlaque et al., 1981; Bamber and Spencer, 1984; Suresh et al., 1993; Qian et al., 1993). However, Lardicci et al. (1999) studied the influence of thermal discharge on spatial variability and abundance of meio-benthic and macrobenthic communities in the Gulf of Follonica (Western Mediterranean), and found that heated effluent did not appear to influence community structure or spatial distribution of the study taxa. Vilanova et al. (2004) studied the effect of thermal discharge on sponge community in an area influenced by the BNPP, and found that species richness, diversity and community density were lower than in control locations. Overall, sessile benthic organisms have been reported as particularly susceptible to heated effluents, and an increase of a few degrees centigrade can threaten their survival (Laws, 1993; Logue et al., 1995). According to Mate (1997), lethal temperatures to reef corals exist within or below the $30\text{--}35^{\circ}\text{C}$ range, which is often reached in the impacted location by BNPP, suggesting that thermal pollution is the main factor in reducing composition and abundance of benthic assemblages.

Other factors introduced by BNPP, including chlorine pollutants and outflow currents, may also influence community density and/or richness in this area. Chlorine is added to the power station cooling water for antibiotic purposes (Jenner et al., 1997), and chlorination byproducts (CBPs) may act as biocides once the water is introduced into the ocean. The chemical reactivity and biocidal properties of these oxidants are well known, and, in principle, both micro- and macro-biota can be inhibited on any given site. Additionally, many benthic marine plants and animals release propagules that serve as the organism's dispersal mechanism. Although propagules are able to explore and select an appropriate settlement site, a wide range of flow regimes exist under which these organisms are unable to explore the surface (Abelson and Denny, 1997). Pawlik and Butman (1993) observed the "erosion" of larvae from the bed at shear velocities (measuring the magnitude and correlation of turbulent fluctuations in velocity near the substratum) higher than 1.03 cm s^{-1} . Under such flow conditions, propagule swimming is entirely ineffective.

Table 3
Total composition as determined by visual census at the studied locations. Density per 540 m² (mean ± SE), total number of specimens counted (*n*) and percent of total observed. The twelve most abundant species are indicated in bold. Locations: I, impacted; C1 and C2, controls.

Family/Species	C1			C2			I		
	Density	<i>n</i>	%	Density	<i>n</i>	%	Density	<i>n</i>	%
Acanthuridae									
<i>Acanthurus chirurgus</i>	1.06 ± 0.47	38	1.90	0.93 ± 0.36	28	0.86	–	–	–
Belontiidae									
<i>Strongylura timucu</i>	0.03 ± 0.03	1	0.05	–	–	–	–	–	–
Blenniidae									
<i>Parablennius</i> sp	0.08 ± 0.05	3	0.15	0.23 ± 0.10	7	0.21	–	–	–
Carangidae									
<i>Caranx latus</i>	–	–	–	0.03 ± 0.03	1	0.03	–	–	–
<i>Trachinotus carolinus</i>	–	–	–	0.07 ± 0.07	2	0.06	–	–	–
Centropomidae									
<i>Centropomus undecimalis</i>	0.11 ± 0.07	4	0.20	–	–	–	–	–	–
Chaetodontidae									
<i>Chaetodon striatus</i>	2.36 ± 0.36	85	4.25	1.73 ± 0.29	52	1.59	–	–	–
Clupeidae									
<i>Sardinella janeiro</i>	13.53 ± 8.07	487	19.59	6.00 ± 2.91	180	5.22	–	–	–
Dactylopteridae									
<i>Dactylopterus volitans</i>	0.64 ± 0.33	23	1.15	–	–	–	0.07 ± 0.07	2	0.06
Dasyatiidae									
<i>Dasyatis americana</i>	–	–	–	0.03 ± 0.03	1	0.03	–	–	–
Diodontidae									
<i>Chilomycterus spinosus</i>	0.14 ± 0.06	5	0.25	0.10 ± 0.06	3	0.09	–	–	–
Gerreidae									
<i>Diapterus rhombeus</i>	0.03 ± 0.03	1	0.05	–	–	–	1.57 ± 1.13	47	1.47
<i>Eucinostomus argenteus</i>	0.72 ± 0.69	26	1.30	0.03 ± 0.03	1	0.03	75.03 ± 22.35	2251	70.37
Haemulidae									
<i>Anisotremus surinamensis</i>	0.33 ± 0.11	12	0.60	0.67 ± 0.21	20	0.61	0.03 ± 0.03	1	0.03
<i>Anisotremus virginicus</i>	0.83 ± 0.23	30	1.50	0.57 ± 0.22	17	0.52	–	–	–
<i>Haemulon aurolineatum</i>	0.14 ± 0.14	5	0.25	0.23 ± 0.13	7	0.21	–	–	–
<i>Haemulon steindachneri</i>	7.33 ± 3.27	264	13.21	18.37 ± 6.26	551	16.87	12.60 ± 3.59	378	11.82
<i>Orthopristis rubber</i>	6.72 ± 2.54	242	12.11	5.40 ± 2.03	162	4.96	0.07 ± 0.07	2	0.06
Holocentridae									
<i>Holocentrus adscensionis</i>	0.22 ± 0.12	8	0.40	0.23 ± 0.08	7	0.21	0.03 ± 0.03	1	0.03
Kyphosidae									
<i>Kyphosus incisor</i>	0.03 ± 0.03	1	0.05	–	–	–	–	–	–
Labridae									
<i>Halichoeres poeyi</i>	0.72 ± 0.18	26	1.30	0.47 ± 0.13	14	0.43	–	–	–
Labrisomidae									
<i>Labrisomus nuchipinnis</i>	0.92 ± 0.18	33	1.65	0.63 ± 0.18	19	0.58	–	–	–
<i>Malacoctenus delalandii</i>	6.39 ± 0.96	230	11.51	3.73 ± 0.66	112	3.43	0.27 ± 0.11	8	0.25
Lutjanidae									
<i>Ocyurus chrysurus</i>	0.03 ± 0.03	1	0.05	–	–	–	–	–	–
Monacantidae									
<i>Monacanthus ciliatus</i>	0.33 ± 0.12	12	0.60	0.07 ± 0.05	2	0.06	–	–	–
<i>Stephanolepis hispidus</i>	0.19 ± 0.10	7	0.35	–	–	–	–	–	–
Mugilidae									
<i>Mugil</i> sp.	0.36 ± 0.33	13	0.65	0.63 ± 0.33	19	0.58	12.73 ± 3.35	382	11.94
Mullidae									
<i>Pseudupeneus maculatus</i>	0.78 ± 0.21	28	1.40	0.50 ± 0.18	15	0.46	–	–	–
Muraenidae									
<i>Gymnothorax funebris</i>	0.03 ± 0.03	1	0.05	0.03 ± 0.03	1	0.03	–	–	–
Ostraciidae									
<i>Acanthostracion polygonius</i>	0.03 ± 0.03	1	0.05	–	–	–	–	–	–
Pempheridae									
<i>Pempheris schomburgkii</i>	–	–	–	3.30 ± 1.75	99	3.03	–	–	–
Pomacentridae									
<i>Abudefduf saxatilis</i>	9.42 ± 1.74	339	16.96	50.23 ± 9.30	1507	46.13	2.40 ± 0.80	72	2.25
<i>Chromis multilineata</i>	0.06 ± 0.04	2	0.10	0.20 ± 0.09	6	0.18	–	–	–
<i>Stegastes fuscus</i>	3.64 ± 0.51	131	6.55	9.30 ± 0.88	279	8.54	–	–	–
<i>Diplodus argenteus</i>	9.17 ± 4.98	330	16.51	9.60 ± 4.67	288	8.82	–	–	–
Scaridae									
<i>Sparisoma</i> sp	0.06 ± 0.04	2	0.10	0.47 ± 0.18	14	0.43	–	–	–
Sciaenidae									
<i>Pareques acuminatus</i>	0.25 ± 0.12	9	0.45	0.27 ± 0.11	8	0.24	–	–	–
Scorpaenidae									
<i>Scorpaena plumieri</i>	0.31 ± 0.12	11	0.55	–	–	–	–	–	–
Serranidae									
<i>Mycteroperca acutirostris</i>	0.83 ± 0.17	30	1.50	0.17 ± 0.07	5	0.15	–	–	–
<i>Mycteroperca bonaci</i>	–	–	–	–	–	–	–	–	–
<i>Serranus flaviventris</i>	0.31 ± 0.10	11	0.55	0.17 ± 0.08	5	0.15	0.07 ± 0.07	2	0.06
Sphyraenidae									
<i>Sphyraena barracuda</i>	0.03 ± 0.03	1	0.05	–	–	–	–	–	–
Syngnathidae									
<i>Micrognathus crinitus</i>	0.03 ± 0.03	1	0.05	–	–	–	–	–	–

(continued on next page)

Table 3 (continued)

Family/Species	C1			C2			I		
	Density	n	%	Density	n	%	Density	n	%
Tetraodontidae									
<i>Sphoeroides greeleyi</i>	0.86 ± 0.17	31	1.55	0.47 ± 0.18	14	0.43	1.60 ± 0.35	48	1.50
<i>Sphoeroides testudineus</i>	0.03 ± 0.03	1	0.05	0.03 ± 0.03	1	0.03	0.17 ± 0.14	5	0.16
Total		2486			3447			3199	

Physical structure has been directly correlated with species richness. Characteristics including rock size, hollows and bottom types provide the organisms with shelter and reproductive habitat. Most studies on habitat complexity have examined the effects of both habitat variables (abiotic physical structure and biotic benthic cover) on fish communities, without discriminating the influence

of each variable separately. Physical structure such as rugosity (topographic complexity) is reliable predictor of fish diversity and, to a lesser extent to their abundance (Chabanet et al., 1997). Ferreira et al. (2001) found rugosity to be a poor descriptor of fish density in rocky shore habitats. In the present work the physical structure was assessed by counting and measuring crevices and rocks in order to have more detailed information on the habitat.

Sargassum bed was the main factor differentiating structural complexity between the two control locations, with lower structural complexity (physical structure) at C1, and higher complexity (physical and benthic cover) at C2. The impacted location showed physical structure comparable to the control location C2, and no significant differences were found in the number of crevices and size of rocks present. However, the locations clearly differed in benthic cover, which was significantly higher at C2. Benthic cover forms a complex framework and supports a variety of microhabitats, thus increasing the potential for fish diversity. Laegdsgaard and Johnson (2001) added artificial mangrove structures to field areas, and found that the increase in fish number was relatively small compared to unstructured areas. However, introduction of artificial structures with accumulated algae attracted approximately four times the number of juvenile fish than either bare stakes or no structure. Many forms of vegetation allow the growth and accumulation of small invertebrates (Stoner, 1982; Robertson, 1984; Lubbers et al., 1990; Schneider and Mann, 1991; Ornellas and Coutinho, 1998), which are an important food source for many juvenile fish species. In the present work, benthic cover, rather than the physical structure, was found to be the underlying factor correlating with fish species richness at the control sites, since a

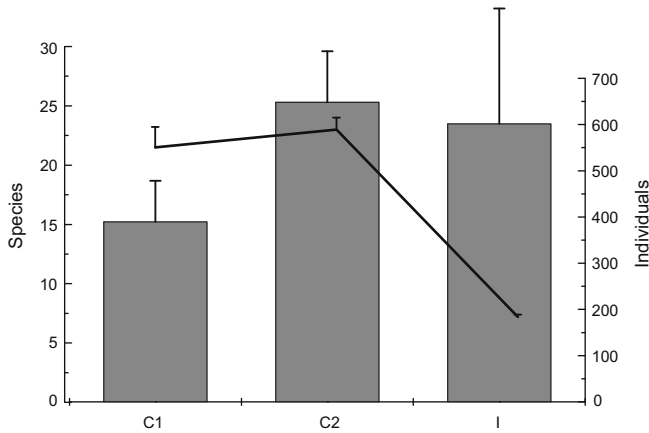


Fig. 3. Mean total number of individuals (column) and species (line) in 540 m². I, impacted location; C1 and C2, control locations.

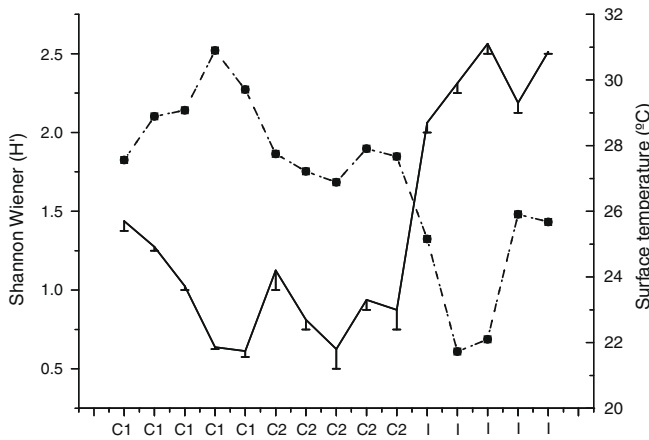


Fig. 4. Spatial patterns of the Shannon–Wiener diversity index (H') of rock fish (dotted line) and water surface temperature (solid line) in Ilha Grande Bay. I, impacted location; C1 and C2, control locations.

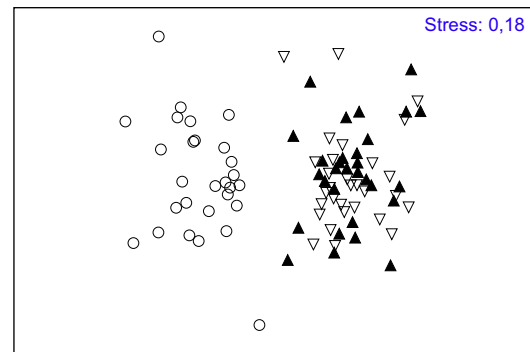


Fig. 5. MDS ordination plot of the relationship between fish community and locations: O, impacted; ▲, C1 and ▽, C2 are controls.

Table 4

Results of one-way ANOSIM (R -values and significance levels) and SIMPER analysis on the species abundance from three locations of the rocky shore in Ilha Grande Bay (I, impacted; C1 and C2, controls).

Groups	ANOSIM		SIMPER		
	R	p	Average dissimilarity (%)	Discriminating species	Contribution (%)
I, C1	0.800	<0.001	83.97	<i>E. argenteus</i>	13.21
I, C2	0.835	<0.001	84.76	<i>E. argenteus</i>	13.32
C1, C2	0.113	<0.001	58.42	<i>A. saxatilis</i>	8.28

Table 5

Discriminating species at each location using SIMPER analysis. Locations: I, impacted; C1 and C2, controls.

Average Similarity (%)	C1 (43.80)	C2 (45.64)	I (48.93)
<i>A. saxatilis</i>	17.71	24.92	
<i>C. striatus</i>	9.14	6.91	
<i>D. argenteus</i>		3.92	
<i>E. argenteus</i>			42.71
<i>H. steindachneri</i>	6.14	9.62	12.24
<i>L. nuchipinnis</i>	4.56		
<i>M. acutirostris</i>	5.62		
<i>M. delalandi</i>	16.77	9.45	
<i>Mugil sp.</i>			19.30
<i>O. rubber</i>	6.56	3.29	
<i>S. fuscus</i>	12.16	26.46	
<i>S. greeleyi</i>	4.74		18.43

low number of species was found at site I even though high physical structure was present at that location.

Correlation is less consistent between habitat structural complexity and fish abundance (Friedlander and Parrish, 1998), although it has been suggested that fish abundance on different reefs increases with structural complexity (Gladfelter et al., 1980). According to Laegdsgaard and Johnson (2001), structural complexity alone may not be greatly attractive to juvenile fish independently of the structure benefits, including shelter or increased surface area for food accumulation. Habitat descriptors showed a little influence on fish community structure in the controls, since most species, with few exceptions, were found to be present in both locations in similar abundance. The ANOSIM showed low *R*-values in comparing C1 and C2, and the MDS confirmed this trend by showing the control locations scattered mainly along the right side of the diagram (Fig. 5).

4.2. Fish communities

Heated effluent appears to influence the fish community structure and the eco-spatial distribution in BNPP area. A thermal discharge effect was detected with the impacted location (I) significantly different from the controls, which exhibited the highest number of fish species. Species richness did not differ between the controls, and was 2.5- to 3-fold higher at the controls than at I. Previous studies examining environmental impacts on fish communities have reported conflicting results. Rong-Quen et al. (2001) performed long-term monitoring of reef fish communities around a nuclear power station, found some inconsistency in the temporal variations of the local fish communities. Since water temperature variations are minor at underwater stations, such temporal variations may be due to natural environmental fluctuations.

Low species richness in the impacted location may be due to the low benthic cover availability, and the resulting limited food availability. With exception of *H. steindachneri*, which is typically associated with rocky shores (Ferreira et al., 2001; Floeter et al., 2007), the abundant species *E. argenteus* and *Mugil sp.* have been reported mostly in sandy banks and muddy flat areas, respectively (Benetti and Neto, 1991; Alvarez-Lajonchere et al., 1992; Chaves and Otto, 1999; Gaelzer and Zalmon, 2003). These species are likely specialized to exploit the available resources in high temperature areas where others species are not able to thrive.

Our description of species compositions provides evidence that opportunistic or tolerant species are present at the impacted location. *E. argenteus*, a genera considered tolerant of thermal stress, was the most abundant species near the power station discharge. In addition, *Mugil sp.* and *H. steindachneri* were observed frequently in high numbers at I. Mora and Ospina (2001) determined the critical thermal maximum (CTM) of reef fish from the tropical eastern

Pacific and found that the CTMs ranged from 36 °C for *H. steindachneri*, 38 °C for *Eucinoctomus gracilis* and 40.8 °C for *Mugil curema*. According to Urban (1994), reef fish are more tolerant on increased temperatures than invertebrates such as bivalves and corals, which almost entirely disappeared from the impacted location.

Effect of thermal discharge on freshwater fish has also been studied in temperate areas. Luksiene and Sandström (1994) documented recruitment failure in a roach (*Rutilus rutilus*) population exposed to cooling water from a Swedish nuclear power plant. Madenjian et al. (1986), studying abundance of alewife (*Alosa pseudoharengus*) and yellow perch (*Perca flavescens*) in a nuclear power plant area in southeastern Lake Michigan found a significant decrease in abundance at the plant-discharge transect relative to the reference transect when plant operation began. The impacts of thermal effluent from a nuclear power plant on the smallmouth bass *Micropterus dolomieu* population in Baie du Doré, Lake Huron was also detected by Shuter et al. (1985), who observed negative effects on growth and spawning timing.

5. Conclusions and future concerns

This study showed that there are different responses of fish assemblage structures, community attributes (species richness and total number of fish), and fish abundance to thermal impact. Heated effluent affected fish assemblage structure at a small spatial scale, and also affected benthic cover, which indirectly impacted the fish community. As observed in this study, areas with no impact by heated water did not show the benthic cover having significant effect on fish community structure. Conversely, in thermally impacted areas, shifts in the composition and structure of the communities are common and severe. As a rule, heated effluent effect on water temperature is a point-source disturbance that gradually decreases at higher distances from the outflow site. In this area, even though the outflow volumes are comparatively small (40 and 80 m³ s⁻¹), their effects were observed as far as 500–600 m from the impacted site.

Our study confirms that thermal pollution induces changes in the marine environment. This is the first result from a South American tropical bay. Our results provide compelling evidence that the benthic cover reduction caused by heated effluent affects fish communities and populations, and the potential consequences that need to be thoroughly investigated. This negative ecological impact can be controlled and mitigated by management and monitoring plans. In addition, our results may provide a critical assessment of potential impacts of global warming on coastal shallow waters, which has already been associated with negative changes in several marine populations.

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