

# Thermal impact of a nuclear power plant in a coastal area in Southeastern Brazil: effects of heating and physical structure on benthic cover and fish communities

Tatiana Pires Teixeira · Leonardo Mitrano Neves · Francisco Gerson Araújo

Received: 22 September 2010 / Revised: 27 August 2011 / Accepted: 23 December 2011  
© Springer Science+Business Media B.V. 2012

**Abstract** The influence of a nuclear power plant's cooling water and physical structure on benthic cover and fish communities were assessed in a coastal area in Southeastern Brazil. We hypothesised that thermal discharges decrease benthic cover and consequently, change the associated rocky reef fish assemblage structure and that physical structure is directly associated with fish richness and diversity. Twelve sites at different distances (close, near and far) from thermal discharge and types of physical structure (low and high) were sampled by visual census. The average surface temperature at the most impacted sites (close) ranged from 30.5 to 31°C, while at far sites it ranged from 25.5 to 28.5°C. Although thermal influences have decreased benthic cover, and consequently, decreased fish richness and diversity, we found that in near and far sites that had complex habitat structures (physical and benthic cover) fish communities were unaffected. The greatest abundances of *Eucinostomus argenteus*, *Mugil curema* and *Sphaeroides greeleyi* were associated with the highest temperatures at the most impacted sites. In contrast, *Abudefduf saxatilis*, *Chaetodon striatus*, *Stegastes fuscus*, *Diplodus*

*argenteus* and *Malacoctenus delalandii* were more abundant at high structured sites far from thermal discharges. Our data support the hypothesis that thermal discharge decreases benthic cover, fish richness and diversity but physical structure, when coupled with high diversity and abundant benthic cover, minimised thermal effects on fish communities.

**Keywords** Thermal pollution · Structural complexity · Habitat · Rocky shore fishes

## Introduction

Temperature is a very important ecological parameter that affects almost every aspect of aquatic life. Heat from the cooling water of nuclear power plants changes the biological and ecological components of coastal area systems in an unpredictable manner. These effects depend on the quantity of heated discharge, climate and the biological features of the environment (Schubel et al., 1978; Lardicci et al., 1999; Chou et al., 2004). Depending on the design and the operating units of the power plants, water temperature in effluent sites can increase by as much as 8°C (Laws, 1993). Consequently in tropical oceans, seawater temperatures can rise to 30°C or higher during the summer. Such high temperatures may approximate or even exceed what resident organisms can tolerate (Jokiel & Coles, 1974; Suresh et al., 1993; Wright et al., 2000). Thus, condenser effluents have the

Handling editor: I. A. Nagelkerken

T. P. Teixeira · L. M. Neves · F. G. Araújo (✉)  
Laboratório de Ecologia de Peixes, Universidade Federal Rural do Rio de Janeiro, BR 465, Km 7, Seropédica, RJ 23.890-000, Brazil  
e-mail: gerson@ufrj.br

potential to trigger thermal and chemical stress (e.g. dissolved oxygen depletion, water chlorination) and therefore, may pose environmental problems to the receiving water body (Krishnakumar et al., 1991; Chou et al., 2004). Tropical marine organisms are particularly susceptible to thermal effects because the water temperature in which they live in is generally close to their vital limits (Langford, 1990).

Several studies have assessed thermal influence on fish, mainly in temperate areas. Some of these studies refer to the biological features of single species that have been measured or observed in the field, such as early development and distribution (Shuter et al., 1985; Madenjian et al., 1986), reproduction (Luksiene & Sandström, 1994), and life history (Dembski et al., 2006). Others are restricted to laboratory experiments on thermal tolerance (Bennett & Judd, 1992; Mora & Ospina, 2001; Ospina & Mora, 2004). The effects of warmer temperatures on freshwater species can be both direct and indirect, and furthermore, the magnitude of these effects can range from minor to fully lethal (Verones et al., 2010). Direct effects include increased activity with faster digestion, which leads to greater food demand and disturbed reproduction (Sandstrom et al., 1997; Luksiene & Sandström, 1994). Indirect effects are related to changes in food availability, community structure, pathogens, chemical processes (modified oxygen content, increased effects of some pollutants) and competition with other, better adapted species (Mariuzzi et al., 1992; Penaz et al. 1999; Beitinger et al., 2000; Contador, 2005; Dembski et al., 2006; Encina et al., 2008; De Vries et al., 2008). However, in tropical coastal systems, the information available on how thermal influence impacts fish communities is limited (Teixeira et al., 2009).

Many fish food sources and shelters (e.g. corals, sponges, macroalgae) are sessile, and will be adversely affected because benthic species are particularly susceptible to thermal discharge effects (Bamber & Spencer, 1984; Bruno et al., 2007). Accordingly, fish may not be able to avoid the indirect effects of thermal pollution, such as decreased benthic cover or even the total lack of benthic cover, because it is used as a feeding resource or for protection. Fish are motile, and most can emigrate to safe areas if temperatures rise above tolerable levels, which would ultimately change their community structure. Moreover, apart from simply leaving an area, rock-dwelling fish may respond

more subtly by changing species distribution (Rong-Quen et al., 2001). Furthermore, a decrease in habitat complexity due to thermal pollution is also expected to decrease fish richness and benthic cover and increase opportunistic and ephemeral species, thereby changing the population dynamics (Devinny, 1980; Mahadevan, 1980; Verlaque et al., 1981; Bamber & Spencer, 1984; Suresh et al., 1993; Qian et al., 1993; Chou et al., 2004). Besides benthic cover, physical structure is known to influence fish richness because it forms a complex framework that supports a variety of microhabitats, thus increasing richness when increasing complexity (Roberts & Ormond, 1987; Chabanet et al., 1997; Öhman & Rajasuriya, 1998). Attributes such as number of holes and crevices, and rock size have been used as physical descriptors (McCormick, 1994). Gladfelter & Gladfelter (1978), working with coral reef communities in the western Atlantic, suggested that fish abundance increases with physical structure. The spatial distribution of topographical characteristics like rock size, holes and bottom types provide organisms with food, shelter and reproductive grounds (Aburto-Opereza & Balart, 2001). Therefore, it is widely accepted that sites with high physical structure and diverse and abundant benthic cover have increased fish richness and diversity.

In the tropics, there are few nuclear power stations, scattered throughout Mexico, India, South Africa, Argentina and Brazil, and studies on thermal influence on fish communities are lacking. In Brazil, there are two nuclear power stations that pump approximately  $120 \text{ m}^3 \text{ s}^{-1}$  of cooling water to generate 1,900 MW of power. These power plants use seawater from Ilha Grande Bay, an embayment area in Southeastern Brazil, for cooling. Given that its operation is likely to modify some physical and chemical characteristics of the local rocky shore environment, and consequently its biological features, it provides an interesting opportunity to evaluate thermal effects on fish communities. We tested the hypothesis that species richness, abundance and diversity in fish communities are influenced by thermal discharges, and that the physical structure of the habitat can minimise such effects because it increases diversity and richness. Our aims were to interpret the overall variation in the structure of a rocky reef fish assemblage across a thermal gradient, and to study the relationship between physical structure and fish richness and abundance. We also investigated the indirect effects of water

warming on fish as a result of the benthic cover modifications.

## Materials and methods

### Study area

This study was conducted along a rocky shore in Ilha Grande Bay, close to the water discharge of the Brazilian Nuclear Power Plant (BNPP), which is comprised of two power plant units. Thermal discharge in this area is a local anomaly that can reach some square kilometres from the outfall (Lucca et al., 2005). This thermal effluent is known to produce an increase of up to 8°C in the area of discharge compared to the adjacent area all year round (Bandeira et al., 2003).

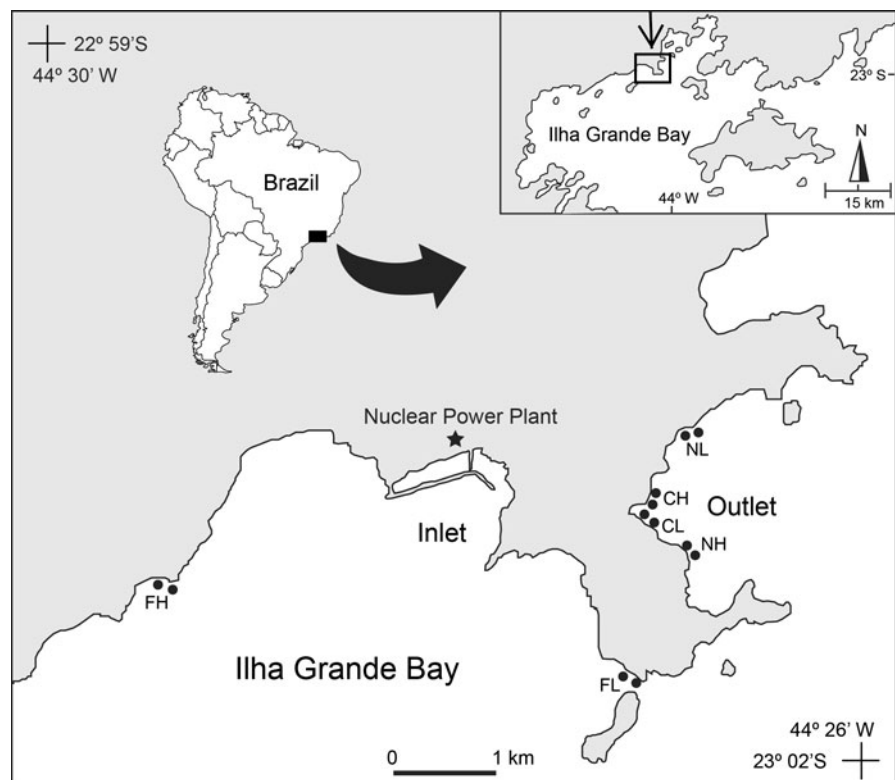
Twelve sites were selected and classified according to distance from the thermal discharge: four close (<200 m), four near (200–1,500 m) and four far (>1,500 m) from the impact source (Fig. 1). The close sites were directly affected by the thermal outfall, the near sites were still affected by outfall

because they were located in a small (2 km<sup>2</sup>) embayment, and the far sites were located outside of the influenced area, ~3.8 km and 9.0 km from the discharge outfall. We chose rocky shore sites in calm waters that were not near sandy beaches or estuarine areas.

### Physical structure

Sites also varied in physical structure. Each distance level had two low structured and two high structured sites. The physical structure of sites was assigned a priori as either high or low based on previous observations of topographical complexity and substratum diversity (Teixeira personal observation). This qualitative assignment was subsequently confirmed with a more detailed quantitative analysis based on a physical structure index. A housed digital camera was mounted onto a 1-m<sup>2</sup> polyvinyl chloride (PVC) photo quadrat framer. Thirty photographs taken in each sampling plot of 90 m<sup>2</sup> were used to quantify the variables describing physical structure following Chapman et al. (1995). Photographs were downloaded to a personal computer, renamed with a unique site

**Fig. 1** Map of the study area with indication of the sampling sites coded by distance from the outfall (close, near, far) and physical structure (low, high): *CL* close low, *CH* close high, *NL* near low, *NH* near high, *FL* far low, *FH* far high



code, and then cropped and colour autocorrected with image-editing software. The number of each descriptor was counted in each photograph (number per square metre).

Each site was analysed based on photographs to quantify the following physical descriptors: rugosity, refuge size categories (holes and crevices), substratum, height and hard substratum. The substratum profile across the 90 m<sup>2</sup> transect was examined to assign a rugosity score (in this study substratum refers to sand, boulders, bare rocks and any artificial structures). Flat sandy areas were given a score of 1 regardless of whether any *Sargassum*, filamentous algae or sessile invertebrates were present. If the substratum was generally flat with few bumps its score was 3, while a very complex substratum profile scored 5. A score of 3 was assigned to substrata of intermediate complexity. Two refuge-type categories (holes and crevices) and three size ranges (<30 cm; 30 cm–1 m; >1 m) were used following the method proposed by Aburto-Opereza & Balart (2001). Holes in rocky shores were easily measured and counted, and crevices (gaps between structures that could provide a path for a fish to escape a predator) were estimated visually. Substratum complexity was also assessed by quantifying rock diversity in different size categories (<30 cm; 30 cm–1 m; >1 m). Habitats that had rocks in a single size category scored 1 while substratum with all size categories present scored 5. The aim of this score was to assess the diversity of structural attributes across habitats, which have the potential to

provide settlement areas for benthic species and resources for different fish species. Habitat architecture height was a subjective visual estimate to distinguish between taller (>100 cm) and shorter (<50 cm) rocks in relation to the lowest point in the site. The height of the habitat architecture was used as a surrogate for the surface available for benthic colonisation. Hard substratum referred to the percentage of substratum that was not sand, rubble or shell–sand patches. Physical structure was assessed using an index score adapted from Gratwicke & Speight (2005) (Table 1). A total score was calculated by adding the scores of each of the five physical descriptors to give an estimate of the overall degree of complexity of the sites physical structure (Table 1). Sites that had a high physical structure had a total score equal or greater than 15, and sites with a low physical structure had a total score below 15. The majority of physical structure classifications made a priori were in agreement with the physical structure index results.

### Sampling

Three samples were undertaken at each site during each sampling occasion, between 2006 and 2008. Site sampling was accomplished on three sampling occasions in each season, totalling 90 samples in the dry/winter season and 84 in the wet/summer season, yielding a total of 174 samples. High structured sites close (30 samples), near (30) and far (30) from the outfall yielded 90 samples, and low structured sites

**Table 1** Physical structure score to discriminate between high and low structured sites

	Physical structure score		
	1	3	5
Rugosity (visual topographic estimate of the substratum in each site)	Flat areas—regardless of any branched algae or soft corals growing on it	Generally flat with few bumps	Very complex substratum profile
Number of refuge size categories: refuge (2): holes and crevices; Size classes (3): <30 cm; 30 cm–1 m; >1 m)	0–1	2–4	4–6
Substratum complexity (number of categories): rock size classes (3): <30 cm; 30 cm–1 m; >1 m.	0–1	2	3
Height (visual estimate of height of physical structure—cm)	0–50	51–100	>100
Hard substratum (%)	0–20	20–40	>40

High structured sites were those that have a total score  $\geq 15$ , while low structured sites have a total score  $< 15$

close (36), near (18) and far (30) yielded 84 samples. Some replicates were not obtained as a result of inclement weather. The factors “Season” and “Year” were not included in the design, because according to our observations, rocky fish in this area did not change structure seasonally (Linear mixed models with randomised sites for the Number of individuals,  $F_{1,173} = 0.08$ ,  $P = 0.77$ ; Species richness,  $F_{1,173} = 0.008$ ,  $P = 0.93$ ) and there were not enough samples to perform comparisons by year.

Underwater visual censuses were performed by SCUBA diving and snorkelling during a census parallel to the coastline along transects 30 m long and 3 m wide (90 m<sup>2</sup>). To obtain fish information, swimming along transects was performed twice. The first time, the observer recorded the conspicuous species, and the second time the observer focused the search beneath rocks and in all crevices to observe the more cryptic species following the procedure of Aburto-Opereza & Balart (2001). The sampling unit, number of fish per 90 m<sup>2</sup>, was defined as the pooled number of conspicuous and cryptic species. Samples were performed in good weather and stable oceanographic conditions, between 9:00 and 14:00 h, during neap tide, near quarter moon. Additionally, we measured the percent of benthic cover from the 30 photographs taken for each site. Each photo was analysed for percentage cover using *Count Point Coral with Excel Extensions*—CPCe 3.4 (Kohler & Gill, 2006), a software programme capable of random point analysis on digital photography. Twenty random points were generated for the photographic analysis.

Benthic cover, similar to the physical structure analysis, was assessed based on photographs to quantify the percentage of each benthic class. Benthic cover descriptors comprised three algae divisions (Chlorophyta, Phaeophyta and Rhodophyta) and sessile invertebrates, mainly represented by the phyla Porifera, Mollusca (genus *Petalconchus* vermitid), Echinodermata (Crinoid) and the class Anthozoa. The class Ascidiacea from the phylum Urochordata was also present. Benthic organisms, expressed as percentage of benthic cover, were grouped in the following classes (adapted from Gratwicke & Speight, 2005): (1) Growth forms applied very generally to living organisms such as coral, algae and other invertebrates that contribute to habitat complexity and support a variety of organism forms, such as branched, cylindrical, tube and pinnate among others.

With this class we assessed the diversity of structural attributes in habitats that could potentially provide resources for different fish species; (2) Encrusting cover referred to individuals that were encrusted in the substratum but did not form a complex habitat; (3) *Sargassum* spp., common and abundant phaeophyta in tropical rocky shores. (4) *Palythoa caribaeorum* encrusting coral; (5) *Jania* sp. rhodophyta algae; and (6) *Petalconchus* sp. vermitid mollusc.

During each sampling occasion, sub-surface (30 cm) and bottom (near to the bottom) water temperature were measured in triplicate.

#### Data analyses

Two-way analysis of variance was used to compare water temperature among the factor distances from the outfall and surface versus bottom, followed by a Tukey post hoc test. A principal component analysis (PCA) was performed on habitat structure descriptors (physical structure + benthic cover) to detect site environmental patterns and to assess thermal influence on benthic cover. Habitat structure descriptors included in the PCA were the scores obtained from physical structure index (rugosity, refuge size categories, substratum complexity, height and hard substratum) and benthic cover classes (growth forms, encrusting cover, *Sargassum* spp., *Palythoa caribaeorum*, *Jania* sp. and *Petalconchus* sp.). Benthic cover descriptors expressed as percentages were arcsine-square-root transformed, and then a  $\log_{10}(x + 1)$  transformation was applied to the whole dataset. Component loads greater than 0.5 were used to identify latent patterns.

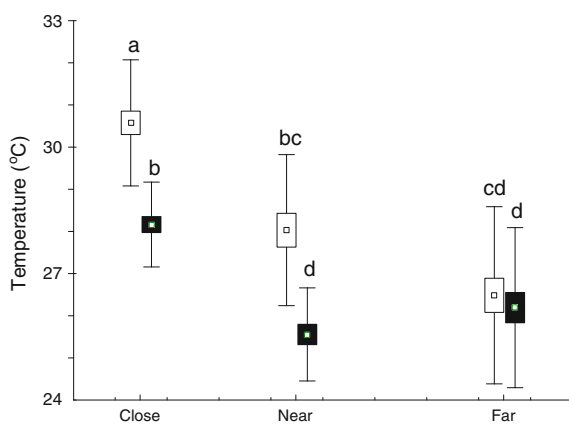
Two factors were tested using linear mixed models: “Distance from the thermal discharge” (close vs. near vs. far), 3 levels, and “Physical structure” (high vs. low), 2 levels. We compared fish species richness, number of individuals, Shannon diversity index (Magurran, 1988), densities of the 10 most abundant species and benthic cover classes among the distances from the outfall and physical structure (fixed factors), with the sites included as a random factor. Linear mixed models take into account the nested structure of data, considering physical structure nested within the distance from the outfall, which are nested into sites. In addition, a Tukey post hoc test was performed when  $H_0$  was rejected. Prior to analysis, the data were transformed to stabilise variances and to minimise the

effect of abundant species (Underwood, 1997). Fish data were  $\log_{10}(x + 1)$  transformed (Sokal & Rohlf, 1998). To explore the relationship between benthic cover (independent variables) and fish abundance (dependent variables), a stepwise multiple regression analysis was performed.

Two-way analysis of similarity (ANOSIM) was used to test for significant differences in fish community structure among distance from the discharges and physical structure. ANOSIM provides an  $R$  statistic that reflects the amount of dissimilarity associated with each factor. An  $R$  value close to one indicates very different composition, whereas values near zero show little difference (Clarke & Warwick, 1994). To reveal the percentage contribution of each taxon to average similarity within each factor, we ran the similarity percentages (SIMPER). Analyses were based on the Bray-Curtis measure and performed with the computer programme PRIMER, version 5 (Clarke & Warwick, 1994).

## Results

Temperature decreased significantly from the close to the far sites ( $F = 32.7$ ,  $P < 0.001$ ), and surface temperature was significantly different from bottom temperature ( $F = 24.3$ ,  $P < 0.001$ ) at the close and near sites but not at the far sites (Fig. 2). Significant distance versus water column interactions ( $F = 3.2$ ,



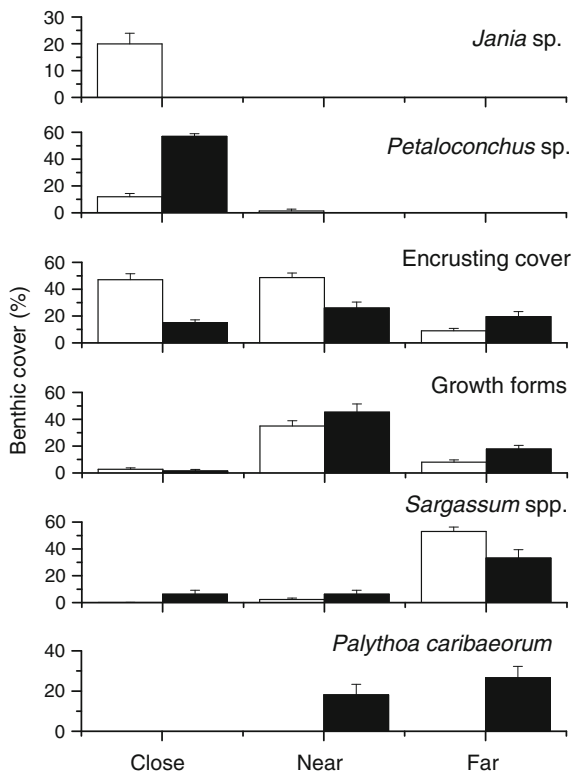
**Fig. 2** Box and whisker plot of variation in surface (white boxes) and bottom (black boxes) water temperatures among distance from the outfall categories. Letters indicate significant difference levels from ANOVA at  $P < 0.05$

$P = 0.04$ ) were found although these were comparatively lower than the main effects of distance from outfall and depth (bottom versus surface). The highest temperature was  $36.2^{\circ}\text{C}$ , recorded at the surface of the close sites, and the lowest temperature was  $23.9^{\circ}\text{C}$ , recorded at the bottom of near and far sites.

All benthic cover classes changed significantly with distance to the thermal discharge. Branched calcareous algae from the *Jania* genus ( $F = 21.8$ ,  $P < 0.01$ ) and *Petalochonchus* sp. vermitid ( $F = 175.4$ ,  $P < 0.01$ ) were found almost exclusively at the sites close to the outfall. *Jania* species were most abundant at low structured sites ( $F = 20.6$ ,  $P < 0.01$ ) and the *Petalochonchus* sp. vermitid was most abundant at high structured sites ( $F = 73.7$ ,  $P < 0.01$ ). Growth forms were most abundant at sites near the outfall ( $F = 29.6$ ,  $P < 0.01$ ), with decreasing abundance at further sites. *Sargassum* spp. were more abundant at the far sites compared to near and close sites ( $F = 81.5$ ,  $P < 0.01$ ). Growth forms were significantly more abundant at high structured sites ( $F = 6.1$ ,  $P < 0.05$ ), while *Sargassum* spp. did not differ in abundance between high and low structured sites. Encrusting cover was widely distributed at all sites irrespective of distance from the outfall but was more abundant at low structured sites ( $F = 27.9$ ,  $P < 0.01$ ) close and near to the outfall ( $F = 23.4$ ,  $P < 0.01$ ), while *Palythoa caribaeorum* was found exclusively at the high structured sites near and far from the outfall (Fig. 3).

The PCA showed that the high structured sites near and far from the outfall have more complex habitats compared to the low structured sites, and to a lesser extent, to the close high structured sites. The first two PCA axes accounted for 43.8% (Axis 1, 26.9%; Axis 2, 16.8%) of the variance among sites. Axis 1 was positively correlated with rugosity and height and negatively correlated with encrusting cover and *Jania* sp. (Fig. 4). Axis 2 was positively correlated with *Petalochonchus* sp. and negatively correlated with growth forms and hard substratum. Sites highly structured close to the thermal outfall corresponded to high percentages of *Petalochonchus* sp., while low structured sites close to the thermal outfall corresponded to high percentage of *Jania* sp. (Fig. 4). On the other hand, high structured sites near and far from the thermal outfall corresponded to high rugosity, hard substrate and height. The low structured sites near and far from the thermal effluent had high percentages of encrusting cover and *Sargassum* spp.





**Fig. 3** Mean percent of benthic cover ( $\pm$ SE) among the thermal gradient (close, near, far) and physical structure (white boxes low, black boxes high)

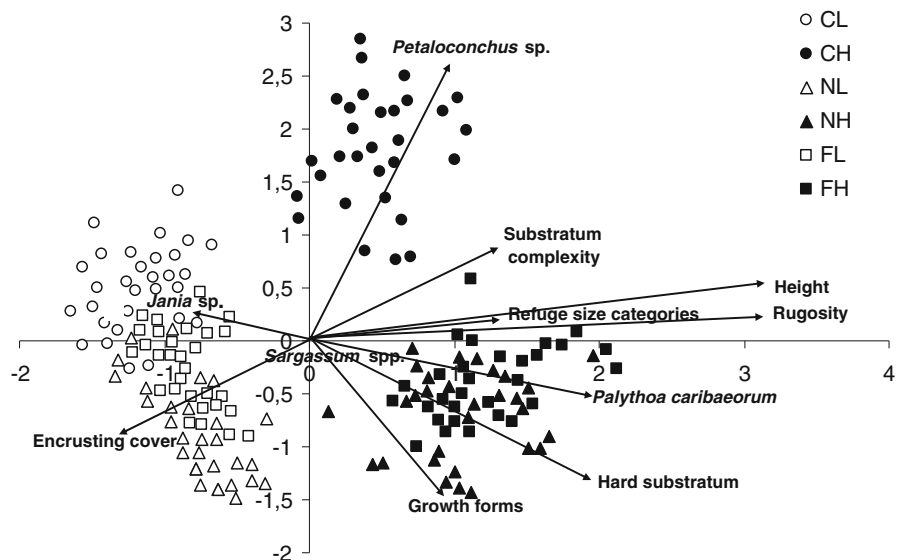
A total of 57 fish species comprising 32 families was observed. A member of the Haemulidae family (*Haemulon steindachneri*), a common group of coastal

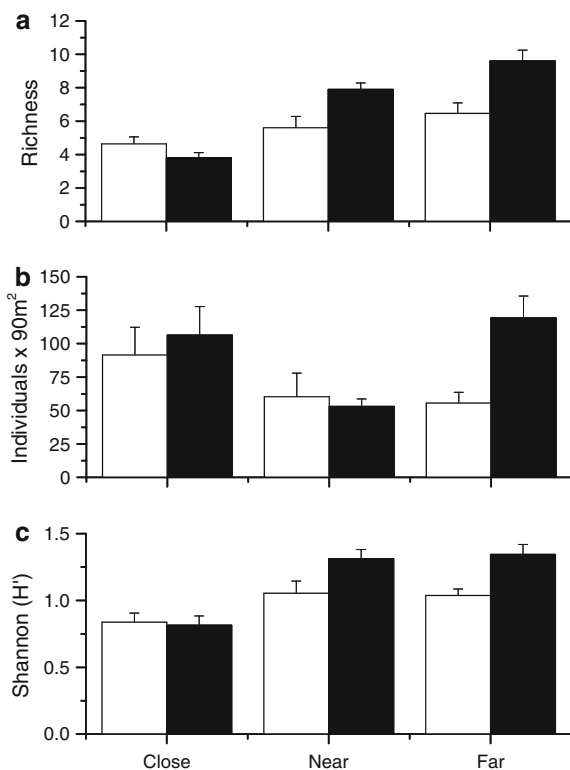
fish, was the most abundant species ( $>9$  individuals  $\times 90$  m<sup>-2</sup>) and was widely distributed across all sites. The 20 most abundant species accounted for 97% of the total number of individuals and had an occurrence higher than 1% throughout all sites.

The five most abundant species at close and near sites accounted for 89% of the total number of individuals (each with a frequency of occurrence  $>4\%$ ) and were, in decreasing order: the gerreid *Eucinostomus argenteus*, the haemulid *H. steindachneri*, the pomacentrid *A. saxatilis*, the haemulid *Haemulon aurolineatum*, and the mugilid *M. curema*. Conversely, the five most abundant species at sites far from the outfall, which contributed to 81% of the total number of fish (each with a frequency of occurrence  $>4\%$ ) were, in decreasing order: *A. saxatilis*, *H. steindachneri*, *H. aurolineatum*, the sparid *Diplodus argenteus* and the territorialist pomacentrid *Stegastes fuscus*.

Species richness increased significantly with distance from the outfall ( $F = 15.8$ ,  $P < 0.01$ ) with higher values at near and far sites compared with the close sites. Moreover, the high structured sites had higher species richness ( $F = 9.8$ ,  $P < 0.01$ ) compared with the low structured sites (Fig. 5a). The mean number of individuals (density) did not differ according to both the distance from the outfall and physical structure (Fig. 5b). Similarly to species richness, the Shannon diversity index differed among distance ( $F = 9.1$ ,  $P < 0.01$ ) and physical structure ( $F = 5.1$ ,  $P < 0.01$ ) (Fig. 5c).

**Fig. 4** Distribution of sampling points on the first two axes of the PCA based on habitat structure descriptors (physical + benthic cover) according to distance from outfall (open circle close, open triangle near, open square far) and physical structure (white low, black high)





**Fig. 5** Mean ( $\pm$ SE) of fish species richness, number of individuals and Shannon diversity index among the thermal gradient (close, near, far) and physical structure (white low, black high)

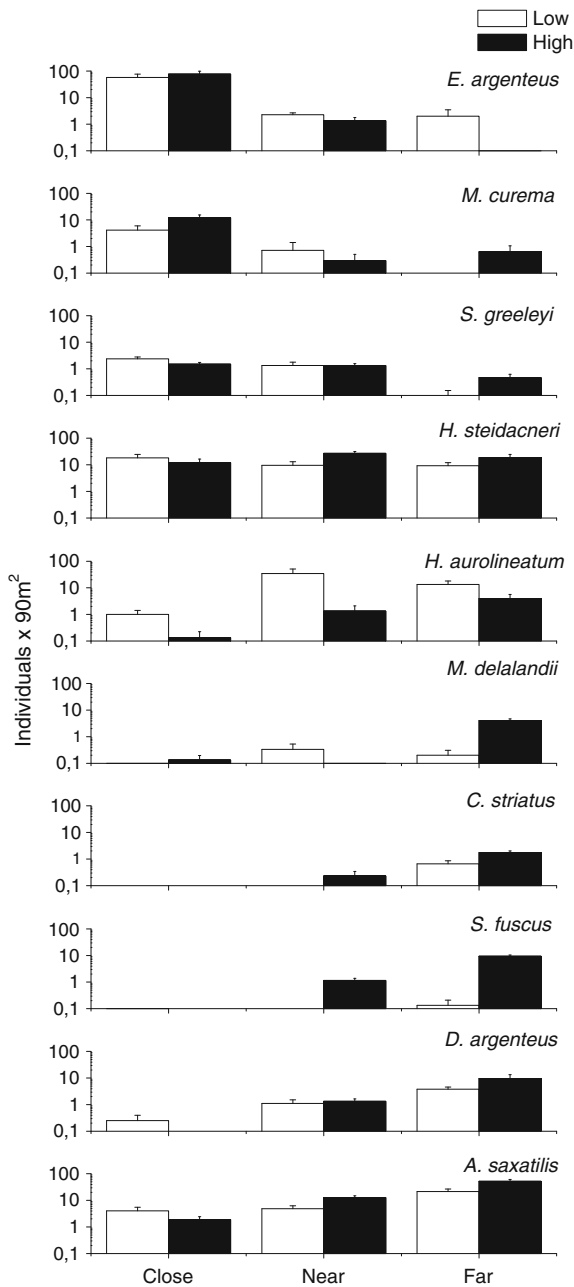
Nine (*E. argenteus*, *M. curema*, *S. greeleyi*, *A. saxatilis*, *D. argenteus*, *S. fuscus*, *H. aurolineatum*, *C. striatus* and *Malacoctenus delalandii*) of the most abundant species differed significantly among the distances from the outfall and six (*M. curema*, *A. saxatilis*, *S. fuscus*, *H. aurolineatum*, *C. striatus* and *M. delalandii*) differed between physical structure. *E. argenteus* ( $F = 5.1$ ,  $P < 0.01$ ) and *M. curema* ( $F = 7.1$ ;  $P < 0.05$ ) had higher abundances at the close sites compared with near and far sites. The pufferfish *Sphoeroides greeleyi* had higher abundance at the sites close to and near the outfall ( $F = 17.7$ ,  $P < 0.01$ ) compared with the far sites. In contrast, *A. saxatilis* ( $F = 33.3$ ,  $P < 0.01$ ), *D. argenteus*, ( $F = 9.8$ ,  $P < 0.01$ ), *S. fuscus* ( $F = 66.0$ ,  $P < 0.01$ ), the butterflyfish *Chaetodon striatus* ( $F = 36.4$ ,  $P < 0.01$ ) and the cryptic scaled blenny *Malacoctenus delalandii* ( $F = 29.1$ ,  $P < 0.01$ ) were significantly more abundant at sites far from the outfall. *H. aurolineatum* was the only species to have significantly higher abundance at the near sites ( $F = 6.2$ ,  $P = 0.01$ ).

*Abudefduf saxatilis* ( $F = 10.3$ ,  $P < 0.05$ ), *S. fuscus* ( $F = 82.1$ ,  $P < 0.01$ ), *C. striatus* ( $F = 10.9$ ,  $P < 0.01$ ) and *M. delalandii* ( $F = 23.2$ ,  $P < 0.01$ ) were significantly more abundant at the high structured sites, while *H. aurolineatum* ( $F = 13.6$ ,  $P < 0.01$ ) was more abundant at the low structured sites (Fig. 6).

When exploring the relationship between habitat structure (physical structure and benthic cover), species richness, number of individuals, Shannon index and abundance of the 10 most numerous species (Table 2), we found that with the exception of number of individuals and abundance of *H. steindachneri*, all fish variables showed significant dependency on habitat structure, save for one species (*H. aurolineatum*) in which the proportion of variance explained was below 10%. The multiple regression analysis of species richness and diversity index,  $H'$ , accounted for 26 and 19% of the variation of their estimated values, respectively. Twenty-six percent of total variation in species richness was explained by the variation in *Petalocochnus* sp., *Jania* sp. (negative relationship) and *Palythoa caribaeorum* and growth forms (positive relationship) (Table 2). Species abundance was generally higher in those sites with more *P. caribaeorum* and *Sargassum* spp. cover (e.g. *A. saxatilis*, *C. striatus*, *M. delalandii* and *S. fuscus*). In the case of *E. argenteus* and *M. curema* (positive relationship) and *A. saxatilis* (negative relationship), *Petalocochnus* sp. explained most of the variance.

Significant differences in the fish community among the distances from the thermal outfall were detected by ANOSIM ( $R$  global = 0.427;  $P < 0.01$ ), indicating shifts in the structure of the community. As expected, the largest difference was found between the close and far sites ( $R = 0.648$ ), but significant differences, although of smaller magnitude, were also found between the close and near ( $R = 0.366$ ;  $P < 0.01$ ) and between the near and far ( $R = 0.209$ ;  $P < 0.01$ ) sites. Furthermore, a low significant difference ( $R$  global = 0.101;  $P < 0.01$ ) was also found between the high and low structured sites. An increased number of typical species was detected at the far sites from the thermal outfall (8 species), whereas the sites close to the outfall were characterised by four species only (Table 3). The low structured sites had a lower number of species that contributed to mean within-group similarity than the high structured sites.





**Fig. 6** Mean abundance for the ten most numerous species ( $\pm$ SE) along the thermal gradient (close, near, far) and physical structure (white low, black high)

## Discussion

Thermal discharges from the Brazilian cooling water nuclear power plants alter fish composition and habitat structure, and this factor was more influential on fish assemblages than the physical structure of surrounding

habitats. Three species were directly associated with the most thermally impacted sites (close to and near the outfall) namely, *E. argenteus*, *M. curema* and *S. greeleyi*, but only the latter is a true rocky shore species. *E. argenteus* was the species that tolerates the most thermal stress, because it was highly abundant at sites that were close to the outfall. Our finding that species were less abundant at sites under thermal pollution coincides with the pattern reported by Rong-Quen et al. (2001), who conducted a 21-year study (1979–1999) on reef fish communities near a nuclear power station in southern Taiwan. Mora & Ospina (2001) determined the critical thermal maximum of reef fish from the tropical eastern Pacific and found that some species are able to colonise high temperature habitats, such as *Haemulon steindachneri* (36°C), *Mugil curema* (40.8°C) and *Eucinostomus gracilis* (38°C). In this study, a similar tolerance pattern was found for *Haemulon steindachneri*, *M. curema* and a *Eucinostomus* species (not *E. gracilis*). These species had a high occurrence at impacted sites and were abundant mainly at those sites, suggesting a preference for areas with high temperatures. The abundant *E. argenteus* and *M. curema* have been reported mostly in sandy banks and muddy flat areas, respectively (Benetti & Neto, 1991; Alvarez-Lajonchere et al., 1992; Chaves & Otto, 1999; Gaelzer & Zalmon, 2003). These species are probably already adapted to living in high temperature areas and as a consequence, they can use the resources available in these areas, whereas other species cannot subsist in high temperatures at all.

As a result of living in areas impacted by thermal influence, fish can alter their behaviour. For instance, some studies on thermal influence have focused on aspects of reproductive biology. Increased temperature can have positive or negative effects on reproductive capacity depending on whether the species is near its optimal thermal point (Munday et al., 2008). Ruttenberg et al. (2005) found that egg production in *Stegastes beebei* is optimal at 25°C and decreases to a very low production rate at either 20 or 27°C. This suggests that with only a few degrees of temperature difference the reproductive success of a given species can significantly decrease. One possible strategy for circumventing this problem is to change the timing of reproduction.

Some information on the influence of thermal discharge on freshwater fish from temperate areas is

**Table 2** Summary of the results of multiple linear regression analyses for fish assemblage structure and abundance of the 10 most numerous species, indicating the benthic cover independent variables included in each model and the sign of their relationship

Variable	Adj. $R^2$	$P$	Variables included in the model
Species richness	0.26	***	– <i>Petalocochus</i> sp.** – <i>Jania</i> sp.* + <i>Palythoa caribaeorum</i> ** + Growth forms**
Number of individuals	0.03	*	– <i>Jania</i> sp.*
Shannon diversity index	0.19	***	+ <i>Palythoa caribaeorum</i> ** + Growth forms*
<i>Eucinostomus argenteus</i>	0.37	***	+ <i>Petalocochus</i> sp.** + Encrusting cover*
<i>Mugil curema</i>	0.18	**	+ <i>Petalocochus</i> sp.**
<i>Sphoeroides greeleyi</i>	0.19	**	– <i>Sargassum</i> spp.** – <i>Palythoa caribaeorum</i> **
<i>Abudefduf saxatilis</i>	0.30	***	– <i>Petalocochus</i> sp.* + <i>Palythoa caribaeorum</i> ** + <i>Sargassum</i> spp.** + Growth forms**
<i>Haemulon steindachneri</i>	0.04	*	+ <i>Palythoa caribaeorum</i> ** + Growth forms**
<i>Diplodus argenteus</i>	0.20	**	+ <i>Sargassum</i> spp.** – <i>Petalocochus</i> sp.**
<i>Stegastes fuscus</i>	0.38	**	+ <i>Palythoa caribaeorum</i> ** + <i>Sargassum</i> spp.** + Growth forms** + Encrusting cover**
<i>Haemulon aurolineatum</i>	0.07	**	– <i>Petalocochus</i> sp.* + <i>Sargassum</i> spp.*
<i>Chaetodon striatus</i>	0.17	***	+ <i>Sargassum</i> spp.** + <i>Palythoa caribaeorum</i> ** + Growth forms* + Encrusting cover*
<i>Malacoctenus delalandii</i>	0.18	**	+ <i>Palythoa caribaeorum</i> ** + <i>Sargassum</i> spp.**

ns not significant; \*  $P < 0.05$ , \*\*  $P < 0.01$ , \*\*\*  $P < 0.001$

available. Luksiene & Sandström (1994) found that *Rutilus rutilus* fail to recruit in areas that are exposed to thermal discharge from a nuclear power station in Sweden. Madenjian et al. (1986) found a decreased abundance of *Alosa pseudoharengus* and *Perca flavescens* at a nuclear power plant on Lake Michigan when compared to a reference condition. Negative effects of thermal influence on the growth and the spawning period of *Micropterus dolomieu* were found for the Doré Bay in Lake Huron (Shuter et al. 1985).

We also found that physical structure plays an important role in fish species composition and distribution along the thermal gradient that seems to have more relevance at sites far from the thermal outfall. For example, higher occurrences and abundances of *S. fuscus*, *A. saxatilis*, *C. striatus* and *M. delalandii* were found only at high structured sites far from the outfall. This can be attributed to the high physical structure and benthic cover found in these areas. These species are cryptic or territorial and habitat complexity is an

**Table 3** Contribution made by each species to the overall measure of similarity within the factor (distance from the outfall and physical structure)

Similarity average (%)	Distance from the outfall			Physical structure	
	Close 37.49	Near 40.10	Far 34.31	Low 27.39	High 30.93
<i>Eucinostomus argenteus</i>	56.94	11.64		29.45	12.49
<i>Sphoeroides greeleyi</i>	17.82	9.25		10.14	8.31
<i>Haemulon steindachneri</i>	10.15	32.96	18.03	22.65	24.02
<i>Mugil curema</i>	6.85				3.27
<i>Abudefduf saxatilis</i>		25.53	40.98	18.41	31.35
<i>Diplodus argenteus</i>		5.50	8.92		3.00
<i>Haemulon aurolineatum</i>		4.74	6.20	9.64	
<i>Serranus flaviventris</i>		2.83			
<i>Stegastes fuscus</i>			7.76		7.84
<i>Chaetodon striatus</i>			4.17		
<i>Malacoctenus delalandii</i>			3.56		
<i>Halichoeres poeyi</i>			2.11		

Only species contributing to more than 2% similarity according to SIMPER are shown

important factor providing shelters, feeding and reproductive grounds (Ferreira et al., 2001). In contrast, warm-water species such as *E. argenteus*, *M. curema* and *S. greeleyi*, and species that occurred mainly at far sites, such as *D. argenteus*, seemed to not be influenced by physical structure.

Physical structure is closely related to fish species richness (Letourneur, 1996; Friedlander & Parrish, 1998; Grober-Dunsmore et al., 2008) because it provides shelter opportunities and places for fish to reproduce. Several studies (Öhman & Rajasuriya, 1998; Aburto-Opereza & Balart, 2001; García-Charton et al., 2004; Tittensor et al., 2007) have reported the effects of habitat structure on fish communities, but most of them fail to discriminate the particular influence of either physical structure or benthic cover. In this study, sites classified as highly structured had more structural complexity and showed greater rugosity and height than the other sites. They were comprised of more physical structures, such as rocks of different sizes, holes and crevices. These features were closely related to higher fish richness and abundance as compared to the low structured sites.

Structural complexity per se, aside from the benefits that structure can provide such as shelter or increased surface area for accumulation of food, may not be greatly attractive to juvenile fish (Laegdsgaard & Johnson, 2001). However, these benefits can be

important for habitat selection for certain species. Cabaitan et al. (2008) found that adding live corals to consolidated dead corals increased juvenile fish colonisation, with these patches acting as true artificial reefs to concentrate organisms. Friedlander & Parrish (1998) reported that physical structure and species abundance are weakly correlated, while Gladfelter et al. (1980) suggested that the abundance of coral fish increases with structural complexity. Therefore, physical structure, when coupled with diverse benthic cover, provided the best configuration for supporting a rich and diverse fish community, as it was observed at the high structured sites far from the outfall.

High fish richness was found at high structured sites near to the outfall. This suggests that the thermal plume was restricted to the more superficial layers. A small difference (about 2°C) between surface and bottom water temperatures seemed not to affect benthic cover, and consequently, fish species richness, which was favoured by high habitat complexity. Thermal discharge affected habitat structure more intensely at the sites close to the outfall, which had low benthic cover. In these sites, a few organisms were found, such as vermetid molluscs of the *Petalocochus* genus and calcareous algae of the *Jania* genus. Therefore, low richness at the most impacted sites was associated with thermal influence, and consequently, low benthic cover, which limited feeding resource

availability. Furthermore, fish abundance was high at the sites close to the outfall, indicating that tolerant species can take advantage of such unoccupied niches and increase in number. Overall, to support high fish richness and abundance, high benthic cover and large rocky areas are required; a notion corroborated at the high structured sites near and far from the outfall.

Lardicci et al. (1999), studying thermal discharges on the spatial distribution of meiobenthic and macrobenthic communities in Follonica Gulf (Western Mediterranean), found that thermal effluent did not influence benthic community distribution. On the other hand, Vilanova et al. (2004), working in the same area as this study, found differences in richness and abundance of sponge communities between an area under thermal influence from the BNPP and a control area. Overall, sessile benthic organisms have been recorded as being susceptible to thermal effluent, and an increase by a few degrees can jeopardise their survival (Laws, 1993; Logue et al., 1995), as was observed at the sites close to the outfall. This suggests that thermal pollution is the main limiting factor for the composition and abundance of benthic communities. In this study, most of the algae species were incrustant and filamentous with limited capacity to form banks and complex patches. Even the growth forms species had a small size (height average = 15 cm, T. P. Teixeira personal observation) with low contribution to structural complexity. Vegetated areas are attractive for fish because they are used as feeding resources and refuge against predators. Conversely, unvegetated areas usually have low fish abundance and diversity (Robertson, 1984; Bell et al., 1987; Ornellas & Coutinho, 1998). Several plants are associated with high diversity of invertebrates, and they are an important part of the diet of juvenile fish (Lubbers et al., 1990; Schneider & Mann, 1991). Zalmon et al. (2002), working with artificial reefs, detected that increased species richness of the reef fish community may be related to the gradual development of a fouling community on the rocky structure.

In this study, we confirmed the influence of thermal pollution on the biota. However, little information is available on the effects of the damage caused by high temperatures on the distribution of fish in tropical areas, and the present findings can provide relevant information for mitigating the potential impacts caused by this kind of anthropogenic influence that is becoming very common in developing countries.

Other natural environmental stressors not accounted for in this study (e.g. predation, limitation in food availability) and human disturbance (e.g. fisheries, pollution, increased flow near to the outfall, water chlorine discharges), may also play an important role in structuring fish communities. For instance, chlorine is added to the power station cooling water for antibiotic purposes (Jenner et al., 1997), and chlorination by-products (CBPs) may act as biocides once the water is introduced into the ocean.

Overall, the sites close to the thermal discharge were more heavily impacted by thermal pollution, reflected by low benthic cover, and consequently, low fish richness, where opportunist and tolerant species dominated. On the other hand, areas that were less impacted were colonised by typical rocky shore species. The more structured sites minimised the thermal influence, as was observed at some sites near the thermal discharge that had benthic cover and a fish community that was relatively unaffected.

We have provided basic information for environmental managers to minimise the influence of thermal pollution and to contribute to the knowledge of changes in habitat structure and fish communities in thermally impacted coastal tropical areas. Environmental policies should consider that the effects of cooling water from nuclear power plants can be minimised by increasing the complexity of physical structures at some hundreds of metres away from the discharges, where, as we report in the study, benthic cover can develop. In such cases, the thermal plume may be restricted to the surface layers making the thermal effects less harmful for fish communities.

**Acknowledgments** We thank Hamilton Hissa Pereira and Rafael Jardim Albiéri for their help in the field work, and to technical staff of the Laboratory of Fish Ecology, University Federal Rural of Rio de Janeiro for useful help in the laboratory. This study was partially financed by CNPq—Brazilian National Council for Research Development (Proc. 302555/2008-0).

## References

- Aburto-Opereza, O. & E. F. Balart, 2001. Community structure of reef fish in several habitats of a rocky reef in the Gulf of California. *Marine Ecology* 22: 283–305.
- Alvarez-Lajonchere, L., E. Trewavas & G. J. Howes, 1992. *Mugil curema* and *Mugil liza* Valenciennes in Cuvier and Valenciennes, 1836 (Osteichthyes, Perciformes): proposed conservation of the specific names. *Bulletin of Zoological Nomenclature* 49: 271–275.

- Bamber, R. N. & J. F. Spencer, 1984. The benthos of a coastal power station thermal discharge canal. *Journal of the Marine Biological Association of the United Kingdom* 64: 603–623.
- Bandeira, J. V., A. A. Barreto, V. L. Bomtempo, R. M. Moreira & L. H. Salim, 2003. The performance of a surface jet on a closed bay: the reality compared with the theory. Proceedings of 6th international conference on coastal and port engineering in developing countries, Colombo, 15 September 2003, CD-ROM, COPEDEC VI, Colombo, Sri Lanka
- Beitingger, T. L., W. A. Bennett & R. W. McCauley, 2000. Temperature tolerances of North American freshwater fishes exposed to dynamic changes in temperature. *Environmental Biology of Fishes* 58: 237–275.
- Bell, J. D., M. Westoby & A. S. Steffe, 1987. Fish larvae settling in seagrass. Do they discriminate between bed of different leaf density? *Journal of Experimental Marine Biology and Ecology* 111: 133–134.
- Benetti, D. D. & E. B. F. Neto, 1991. Preliminary results on growth of mullets (*Mugil liza* and *Mugil curema*) fed artificial diets. *World Aquaculture* 22: 55–57.
- Bennett, W. A. & F. W. Judd, 1992. Comparison of methods for determining low temperature tolerance: experiments with pinfish, *Lagodon rhomboids*. *Copeia* 1992: 1059–1065.
- Bruno, J. F., E. R. Selig, K. S. Casey, C. A. Page, B. L. Willis, C. D. Harvell, H. Sweatman & A. M. Melendy, 2007. Thermal stress and coral cover as drivers of coral disease outbreaks. *Public Library of Science (PLoS) Biology* 5(6): e124.
- Cabaitan, P. C., E. D. Gomez & P. M. Aliño, 2008. Effects of coral transplantation and giant clam restocking on the structure of fish communities on degraded patch reefs. *Journal of Experimental Marine Biology and Ecology* 357: 85–98.
- Chabanet, P., H. Ralambondrainy, M. Amanieu, G. Faure & R. Galzin, 1997. Relationships between coral reef substrata and fish. *Coral Reefs* 16: 93–102.
- Chapman, M. G., A. J. Underwood & G. A. Skilleter, 1995. Variability at different spatial scales between a subtidal assemblage exposed to the discharge of sewage and two control assemblages. *Journal of Experimental Marine Biology and Ecology* 189: 103–122.
- Chaves, P. T. C. & G. Otto, 1999. The mangrove as a temporary habitat for fish: the *Eucinostomus* species at Guaratuba Bay, Brazil (25°52'S; 48°39'W). *Brazilian Archives of Biology and Technology* 42: 61–68.
- Chou, Y., T. Y. Lin, C. T. A. Chen & L. L. Liu, 2004. Effect of nuclear power plant thermal effluent on marine sessile invertebrate communities in Southern Taiwan. *Journal of Marine Science and Technology* 12: 448–452.
- Clarke, K. R. & R. M. Warwick, 1994. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation. Natural Environment Research Council, Plymouth.
- Contador, J. F. L., 2005. Adaptive management, monitoring, and the ecological sustainability of a thermal-polluted water ecosystem: a case in SW Spain. *Environmental Monitoring and Assessment* 104: 19–35.
- De Vries, P., J. E. Tamis, A. J. Murk & M. G. D. Smit, 2008. Development and application of a species sensitivity distribution for temperature-induced mortality in the aquatic environment. *Environmental Toxicology and Chemistry* 27: 2591–2598.
- Dembski, S., G. Masson, D. Monnier, P. Wagner & J. C. Pihan, 2006. Consequences of elevated temperatures of life-history traits of an introduced fish, pumpkinseed *Lepomis gibbosus*. *Journal of Fish Biology* 69: 331–346.
- Deviny, J. S., 1980. Effects of thermal effluents on communities of benthic marine macro-algae. *Journal of Environmental Management* 11: 225–242.
- Encina, L., A. Rodríguez-Ruiz & C. Granado-Lorencio, 2008. Distribution of common carpina Spanish reservoir in relation to thermal loading from a nuclear power plant. *Journal of Thermal Biology* 33: 444–450.
- Ferreira, C. E. L., J. E. A. Gonçalves & R. Coutinho, 2001. Community structure of fishes and habitat complexity on a tropical rocky shore. *Environmental Biology of Fishes* 61: 353–369.
- Friedlander, A. M. & J. D. Parrish, 1998. Habitat characteristics affecting fish assemblages on a Hawaiian coral reef. *Journal of Experimental Marine Biology and Ecology* 26: 65–84.
- Gaelzer, L. R. & I. R. Zalmon, 2003. The influence of wave gradient on the ichthyofauna of Southeastern Brazil: focusing the community structure in surf-zone. *Journal of Coastal Research* 35: 456–462.
- García-Charton, J. A., A. Pérez-Ruzafa, P. Sánchez-Jerez, J. T. Bayle-Sempere, O. Reñones & D. Moreno, 2004. Multi-scale spatial heterogeneity, habitat structure, and the effect of marine reserves on Western Mediterranean rocky reef fish assemblages. *Marine Biology* 144: 161–182.
- Gladfelter, W. B. & E. H. Gladfelter, 1978. Fish community structure as a function of habitat structure on West Indian patch reefs. *Revista de Biología Tropical* 26: 65–84.
- Gladfelter, W. B., J. C. Ogden & E. H. Gladfelter, 1980. Similarity and diversity among coral reef communities: a comparison between tropical western Atlantic (Virgin Islands) and tropical central Pacific (Marshall Islands) patch reefs. *Ecology* 61: 1156–1168.
- Gratwicke, B. & M. R. Speight, 2005. The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. *Journal of Fish Biology* 66: 650–667.
- Grober-Dunsmore, R., T. K. Frazer, J. P. Beets, W. J. Lindberg, P. Zwick & N. A. Funicelli, 2008. Influence of landscape structure on reef fish assemblages. *Landscape Ecology* 23: 37–53.
- Jenner, H. A., C. J. L. Taylor, M. Van Donk & M. Khalanski, 1997. Chlorination by products in chlorinated cooling water of some European coastal power station. *Marine Environmental Research* 43: 279–293.
- Jokiel, P. L. & S. L. Coles, 1974. Effects of heated effluent on hermatypic corals at Kahe Point, Oahu. *Pacific Science* 28: 1–18.
- Kohler, K. E. & S. M. Gill, 2006. Coral Point Count with Excel extensions (CPCe): A Visual Basic program for the determination of coral and substrate coverage using random point count methodology. *Computers and Geosciences* 32: 1259–1269.
- Krishnakumar, V., J. S. Sastry & G. N. Swamy, 1991. Implication of thermal discharges into the sea—a review. *Indian Journal of Environmental Protection* 11: 525–527.

- Laegdsgaard, P. & C. Johnson, 2001. Why do juvenile fish utilize mangrove habitats? *Journal of Experimental Marine Biology and Ecology* 257: 229–253.
- Langford, T. E. L., 1990. *Ecological Effects of Thermal Discharges*. Elsevier, London.
- Lardicci, C., F. Rossi & F. Maltagliati, 1999. Detection of thermal pollution: variability of benthic communities at two different spatial scales in an area influenced by a coastal power station. *Marine Pollution Bulletin* 38: 296–303.
- Laws, E. A., 1993. *Aquatic Pollution—An Introductory Text*. Wiley, New York.
- Letourneur, Y., 1996. Dynamics of fish communities on Reunion fringing reefs, Indian Ocean: 1. Patterns of spatial distribution. *Journal of Experimental Marine Biology and Ecology* 195: 1–30.
- Logue, J., P. Tiku & A. R. Cossins, 1995. Heat injury and resistance adaptation in fish. *Journal of Thermal Biology* 20: 191–197.
- Lubbers, L., W. R. Boynton & W. M. Kemp, 1990. Variations in structure of estuarine fish communities in relation to abundance of submersed vascular plants. *Marine Ecology Progressive Series* 65: 1–14.
- Lucca, E. V. D., J. V. Bandeira, J. A. Lorenzetti, R. C. Moreira, R. M. Castro, L. H. Salim, O. D. Z. Júnior & E. S. C. Esposito, 2005. Uso de sensor hiperespectral aerotransportado no monitoramento da pluma termal oceânica decorrente da descarga de refrigeração da central nuclear de Angra dos Reis. *Revista Brasileira de Cartografia* 57: 48–55.
- Luksiene, D. & O. Sandström, 1994. Reproductive disturbance in a roach (*Rutilus rutilus*) population affected by cooling water discharge. *Journal of Fish Biology* 45: 13–625.
- Madenjian, C. P., D. J. Jude & F. J. Tesar, 1986. Intervention analysis of power-plant impact on fish populations. *Canadian Journal of Fisheries and Aquatic Sciences* 43: 819–829.
- Magurran, A. E., 1988. *Ecological Diversity and Its Measurement*. Croom Helm, London.
- Mahadevan, S., 1980. A study on the effects of power plant thermal discharges on benthic infaunal communities at Big Bend, Tampa Bay (Florida). *Florida Science* 43: 7–8.
- Mariazzi, A. A., J. L. Donadelli, P. Arenas, M. A. Di Siervi & C. Bonetto, 1992. Impact of a nuclear power plant on water quality of Embalse del Rio Tercero Reservoir, (Cordoba, Argentina). *Hydrobiologia* 246: 129–140.
- McCormick, M. I., 1994. Comparison of field methods for measuring surface topography and their associations with a tropical reef fish assemblage. *Marine Ecology Progress Series* 112: 87–96.
- Mora, C. & A. F. Ospina, 2001. Tolerance to high temperatures and potential impact of sea warming on reef fishes of Gorgona Island (tropical eastern Pacific). *Marine Biology* 139: 765–769.
- Munday, P. L., G. P. Jones, M. S. Pratchett & A. J. Williams, 2008. Climate change and the future for coral reef fishes. *Fish and Fisheries* 9: 261–285.
- Öhman, M. C. & A. Rajasuriya, 1998. Relationships between habitat structure and fish communities on coral and sandstone reefs. *Environmental Biology of Fishes* 49: 45–61.
- Ornellas, A. B. & R. Coutinho, 1998. Spatial and temporal patterns of distribution and abundance of a tropical fish assemblage in a seasonal *Sargassum* bed, Cabo Frio Island, Brazil. *Journal of Fish Biology* 53(Suppl A): 198–208.
- Ospina, A. F. & C. Mora, 2004. Effect of body size on reef fish tolerance to extreme low and high temperatures. *Environmental Biology of Fish* 70: 339–343.
- Penaz, M., V. Barus & M. Prokes, 1999. Changes in the structure of fish assemblages in a river used for energy production. *Regulated Rivers-Research & Management* 15: 169–180.
- Qian, S., H. Chen, X. Zhao & Q. Zhang, 1993. A study of the effect of thermal pollution on seaweeds. *Journal Ocean University of Qingdao* 23: 22–34.
- Roberts, C. M. & R. F. G. Ormond, 1987. Habitat complexity and coral reef fish diversity and abundance on Red Sea fringing reefs. *Marine Ecology Progress Series* 41: 1–8.
- Robertson, A. I., 1984. Trophic interactions between the fish fauna and macrobenthos of an eelgrass community in Western Port, Victoria. *Aquatic Botany* 18: 135–153.
- Rong-Quen, J., C. Jeng-Ping, L. Chun-Yu & S. Kwang-Tsao, 2001. Long-term monitoring of the coral reef fish communities around a nuclear power plant. *Aquatic Ecology* 35: 233–243.
- Ruttenberg, B. I., A. J. Haupt, A. I. Chiriboga & R. R. Warner, 2005. Patterns, causes and consequences of regional variation in the ecology and life history of a reef fish. *Oecologia* 145: 394–403.
- Sandstrom, O., I. Abrahamsson, J. Andersson & M. Vetemaa, 1997. Temperature effects on spawning and egg development in Eurasian perch. *Journal Fish Biology* 51: 1015–1024.
- Schneider, F. I. & K. H. Mann, 1991. Species specific relationships of invertebrates to vegetation in a seagrass bed. I. Correlational studies. *Journal of Experimental Marine Biology and Ecology* 145: 101–117.
- Schubel, J. R., C. C. Coutant & P. M. J. Woodhead, 1978. Thermal effects of entrainment. In Schubel, J. R. & B. C. Marcy (eds), *Power Plant Entrainment—A Biological Assessment*. Academic Press Inc., New York: 19–93.
- Shuter, B. J., D. A. Wismer, H. A. Regier & J. E. Matuszek, 1985. An application of ecological modeling: impact of thermal effluent on a smallmouth bass population. *Transactions of the American Fisheries Society* 114: 63–651.
- Sokal, R. R. & J. Rohlf, 1998. *Biometry: The Principles and Practice of Statistics in Biological Research*. W.H. Freeman, New York.
- Suresh, K., M. S. Ahamed, G. Durairaj & K. V. K. Nair, 1993. Impact of power plant heated effluent on the abundance of sedentary organism, off Kalpakkam, east coast of India. *Hydrobiologia* 268: 109–114.
- Teixeira, T. P., L. M. Neves & F. G. Araújo, 2009. Effects of a nuclear power plant thermal discharge on habitat complexity and fish community structure in Ilha Grande Bay, Brazil. *Marine Environmental Research* 68: 188–195.
- Tittensor, D. P., F. Micheli, M. Nyström & B. Worm, 2007. Human impacts on the species–area relationship in reef fish assemblages. *Ecological Letters* 10: 760–772.
- Underwood, A. J., 1997. *Experiments in Ecology: Their Logical Design and Interpretation Using Analysis of Variance*. Cambridge University Press, Cambridge.



- Verlaque, M., G. Giraud & C. F. Boudouresque, 1981. Effects of a thermal power plant on the Mediterranean marine phythobenthos: the area of high frequency temperature changes. *Botanica Marina* 24: 69–87.
- Verones, F., M. M. Hanafiah, S. Pfister, M. A. J. Huijbregts, G. J. Pelletier & A. Koehler, 2010. Characterization factors for thermal pollution in freshwater aquatic environments. *Environmental Science & Technology* 44: 9364–9369.
- Vilanova, E., M. M. Pinto, M. P. Curbelo-Fernandez & S. H. G. Silva, 2004. The impact of a nuclear power plant discharge on the sponge community of a tropical Bay (SE Brazil). *Bollettino dei Musei e degli Istituti Biologici dello Università di Genova* 68: 647–654.
- Wright, J. F., J. M. R. Winder, J. M. Gunn, J. H. Blackburn, K. L. Symes & R. T. Clarke, 2000. Minor local effects of a river Thames power station on the macroinvertebrate fauna. *Regulated Rivers: Research & Management* 16: 159–174.
- Zalmon, I. R., R. Novelli, M. P. Gomes & V. V. Faria, 2002. Experimental results of an artificial reef programme on the Brazilian coast north of Rio de Janeiro. *ICES Journal of Marine Science* 59: 83–87.