The impact of coastal urbanization on the structure of phytobenthic communities in southern Brazil

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ABSTRACT

The anthropogenic pressures on coastal areas represent important factors affecting local, regional, and even global patterns of distribution and abundance of benthic organisms. This report undertakes a comparative analysis of the community structure of rocky shore intertidal phytobenthos in both pristine like environments (PLE) and urbanized environments (UBE) in southern Brazil, characterizing variations on different spatial scales. Multivariate analysis of variance indicated that the PLE is characterized by a larger number of taxa and an increased occurrence of Rhodophyta species in relation to UBE. In contrast, UBE were dominated by opportunistic algae, such as *Cladophora* and *Ulva* spp. Significance tests further indicated higher species richness and Shannon–Wiener diversity on the PLE in relation to UBE. Here we provide data showing the magnitude of seaweed biodiversity loss and discuss direct and indirect consequences of unplanned urbanization on these communities.

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1. Introduction

Although the patterns of distribution and abundance of marine biota are known and reasonably foreseeable at different scales (Lüning, 1990), the increasing human ecological footprint on coastal ecosystems may profoundly change the local, regional, and even global structure of these communities. Such pressures were highlighted recently with the decline of phytoplankton on a global scale, mainly in tropical and subtropical regions, which, among other reasons, were attributed to coastal processes such as the input of continental effluents (Boyce et al., 2010). At the end of the last century, coastal population was approximately 3.6 billion people living no more than 100 km from the coastline (Vitousek et al., 1997). This has resulted in a pattern of irregular and disordered occupation which exerts even greater stresses on the coastal ecosystem. In the developing and underdeveloped countries this reality is even more striking, with occupancy rates larger than those observed in developed countries. Since most of these impacted regions are located in the southern hemisphere, this scenario is particularly worrisome because survival of the local population

depends, either directly or indirectly, on the services and products provided by these ecosystems. This is aggravated by deforestation, combined with the release of domestic sewage *in natura*, which increases the degradation potential on coastal environments. Organic and inorganic pollutants, nutrients, turbidity, suspended solids in excess, and habitat changes, including hydrological regime, are examples of stressors that may impact the biota, singly or in combination, including synergistic effects (Adams, 2005).

Anthropogenic stressors promote a particular impact on benthic communities, which are often structured by sessile organisms such as seaweeds. As seen in other regions, such impacts can result in the reduction of species number and abundance of primary producers, with consequent simplification of community structure. This, in turn, leads to an eventual increase in the abundance of opportunistic taxa with high reproductive capacity and tolerance against pollution. By integrating the effects of prolonged exposure to adverse conditions, the rocky shore seaweeds can be considered good indicators of environmental changes caused by different types of disturbances (Gorostiaga and Díez, 1996). The potential application of these organisms as bioindicators is even greater because they are widely and easily found in intertidal regions, which frequently represent the entry point of continental effluents. Studies using seaweeds as a means of diagnosing the degradation of coastal environments have been conducted mostly in the northern

hemisphere, as in Asia (Reopanichkul et al., 2009), Mediterranean (Ballesteros et al., 2007; Orfanidis et al., 2001, 2007; Terlizzi et al., 2002), Ireland, Scotland and England (Wells et al., 2007; Wilkinson et al., 2007), Spain (Juanes et al., 2008), Red Sea (Bahartan et al., 2010) and Caribbean (Littler et al., 2010). On the other hand, few studies have been reported on the subtropical southern Atlantic Ocean. Furthermore, these studies have not presented an adequate sampling design or large enough scale, most treating only single bays or cities (Amado Filho et al., 2003; Oliveira and Qi, 2003).

Based on accumulated evidence, human impact is putting the composition and structure of seaweed communities at risk. Therefore, the current challenge is to understand the conditions and mechanisms that cause the loss of habitats, with particular emphasis on lesser known areas of the world, especially those located in the southern hemisphere, in order to prevent future declines in biodiversity (Connell et al., 2008). These conditions imply an urgent need to provide subsidies for the development of tools that contribute to the management of human activities that impact the marine environment, especially in regions still under-developed, providing indicators that measure the extent of the impact at different scales (Rogers and Greenaway, 2005). To address this need, the present study aimed to compare both structure and composition of rocky shore intertidal phytobenthic communities in pristine like and urbanized environments in southern Brazil, where urban areas have experienced a sharp growth process in the past few decades. In so doing, this study not only shows the magnitude of the qualitative and quantitative macroalgae biodiversity loss, but also discusses the direct and indirect consequences of rapid and unplanned urbanization on such seaweed communities.

2. Materials and methods

2.1. Study area and sampling design

The study area is located on the southern Brazilian continental shelf, with relatively homogeneous oceanographic characteristics (Pereira et al., 2009), in an area subjected to the action of the convergence of Brazil and Falklands currents. According to Sanderson et al. (2003), the human footprint of the southwestern Atlantic, based on the Global Human Influence Index (GHII) is considered moderate to high (Fig. 1).

We selected four municipalities (sites) along the southern Brazilian coast, between the latitudes 26°58′19″S and 29°21′63″S, and in each one we selected two beaches with features of urban and non-urban or pristine like environments (treatments). Three rocky shores were randomly selected for sampling within each beach. Definition of urbanized (UBE) and pristine like (PLE) environments was based on the characterization of the density patterns of human occupation measured from data gathered by the Brazilian national census (IBGE, 2010) and high-resolution Digital Globe images, taken from 10 km in altitude (Google Earth®), for analysis of land use (size of urban area and percent of urban and vegetation cover) and the distance from the nearest discharge of sewage.

To complement the characterization, in each site we analyzed the concentration of dissolved inorganic nutrients and suspended particulate matter. These descriptors were selected because they are possible indicators of the presence of effluents and because they directly influence the ecophysiology of seaweeds. Triplicates of one liter of water, in each sampled station were passed through GF/F Whatman filters 0.45 um. Filters for determination of suspended particulate material (SPM) were prepared according to Strickland and Parsons (1972). Dissolved inorganic nutrients (NO_3^{-2}, NH_4^+, PO_4^{3-}) were determined by a colorimetric method, using a spectrophotometer (Pro-analysis, UV-1100), equipped with
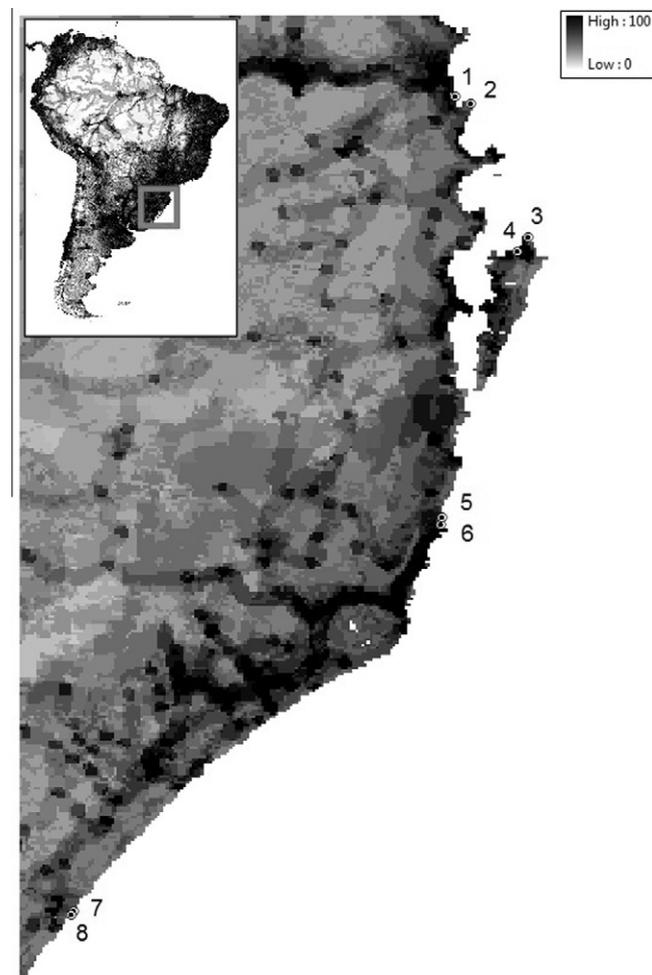

Fig. 1. Study area map based on human footprint showing sites sampled. The gray scales represent the relative density of human occupation. Modified from Sanderson et al. (2003).

buckets with 5 cm optical step. Phosphate and nitrate were determined according to Grasshoff et al. (1983). Concentration of ammonia was determined according to Tréguer and Le Corre (1976).

2.2. Sampling procedures

For qualitative and quantitative phytobenthos analysis, three rocky shores were selected in each PLE and UBE. Only rocky substrate with similar slope, orientation and wave exposure were selected for comparison, in order to minimize the effects of other sources of variability. Seaweeds were manually collected during one sampling campaign, between February 10 and March 31, 2010. Qualitative samples were made in all selected rocky shores for identification and documentation. For quantitative analysis each selected rocky shore was characterized by 15 photo-quadrates (25 × 25 cm), randomly positioned following a Cartesian plot (10 × 2 m), with the longer side placed parallel to the coast line and with the lower limit adjacent to the water level, during low spring tides. In a pilot study we determined the minimum number of samples (around 10) that included more than 95% of species richness by plotting the cumulative number of species in relation to the number of samples, according to Murray et al. (2006). As the number of samples depends mainly on the spatial distribution and density of benthic populations and this is in the sampling area, 15 photo-quadrates can be considered representative for our purposes.

Coverage was subsequently analyzed using the Coral Point Count software with Excel extensions CPCe (Kohler and Gill, 2006). All species were identified under 30 random points plotted by the software within each sample element.

2.3. Data analysis

To verify the similarity and dissimilarity between preserved and urbanized environments, as well as the percentage contribution of each species, SIMPER analysis was applied. The data were not transformed to perform such tests, which were made using the software PRIMER 6.0 (Clarke and Warwick, 2001). Multivariate analysis of variance was performed using the PERMANOVA software (Anderson, 2001) to test the differences in phytobenthos community structure between PLE and UBE. The analysis was based on Bray–Curtis dissimilarity matrix using all samples with not transformed data, and was made on unrestricted raw permutation data run 9999 times. Univariate analyses of urbanization effects were done on the synthetic descriptors of abundance (coverage of macroalgae), species richness, Shannon–Wiener diversity and Pielou evenness indexes using two-way ANOVAs on treatments (PLE and UBE) and sites (four sites). We used sites as random factor and treatments as fixed factor. Student–Newman–Keuls (SNK) tests were used to evaluate significant differences among means. The homogeneity of variances was previously checked using the Cochran test. ANOVA analysis were run with R 2.12.1 software (R development core team, 2011), using the GAD package (Sandrini-Neto and Camargo, 2010).

3. Results

Urbanized coastal environments were characterized by soil impermeability, deforestation, continental runoff and sewage discharge, including domestic sewage *in natura*. In the surveyed urbanized areas an average of 85% of the population is not served by sanitary sewage system (Cunha et al., 2008) and the beaches are labeled by local Sanitary Agency as unsatisfactory for swimming for a large proportion of the year (FATMA, 2010). On the other hand, the less impacted areas selected for comparison, here named as pristine like environments, corresponded to areas with little or no urbanization, surrounded by vegetation and distant from sewage outfalls (Table 1).

We identified 74 taxa, represented by 49 Rhodophyta, 13 Phaeophyceae (Ochrophyta) and 12 Chlorophyta (Table 2). Within Rhodophyta, Ceramiales contributed to the largest species number (18). Among the Phaeophyceae, the orders Ectocarpales (4 spp.)

and Fucales (3 spp.) were more represented, while among the Chlorophyta, Cladophorales (6 spp.) and Ulvales (3 spp.) were more represented. Stations classified as PLEs hold the largest total number of species (59), among which 39 Rhodophyta, while the UBEs hold 55 taxa with 34 Rhodophyta.

Within the PLEs, there was a similarity of 37.14% among the samples, while among the samples in the UBEs, similarity was 49.55%. The dissimilarity between the beaches of the two environments was 67.20%. The genus *Ulva* had the highest percentage of coverage in UBE (62.55%) and PLE (50.17%) stations. Genera corresponding to the largest percentage of coverage in PLEs were, respectively: *Ulva, Centroceras, Gelidium, Chaetomorpha, Hypnea, Sargassum* and *Gymnogongrus*. Data of impacted beaches indicated a greater occurrence of species of *Centroceras, Hypnea, Gymnogongrus, Chaetomorpha* and *Gelidium*, preceded by *Ulva*, as reported above (Table 3).

The community structure analysis performed through a permutational multivariate analysis of variance indicated significant differences between the pristine and urbanized environments irrespective of among site differences (Table 4).

All two-way ANOVAs showed no significant differences among the four sites, significant differences between treatments depending of the descriptor analyzed, and no interaction between sites and treatments (Fig. 2). The univariate tests indicated that species richness and Shannon–Wiener diversity were higher on the PLEs (4.36 ± 1.24 and 1.57 ± 0.43, respectively) compared to UBEs (3.06 ± 0.71 and 1.15 ± 0.30, respectively). There were no significant differences between treatments for abundance and evenness synthetic descriptors (99.57 ± 2.05 and 0.74 ± 0.10, respectively).

4. Discussion

Our results illustrate the impact of human activities along coastal areas of the subtropical southwestern Atlantic corroborating studies made elsewhere, which also showed a decrease in species richness in coastal regions through different descriptors (Bahartan et al., 2010; Cardoso et al., 2008; Johnston and Roberts, 2009; Reopanichkul et al., 2009; Scrosati and Heaven, 2007). Such impact undermines the maintenance of benthic communities by reducing their diversity, and consequently affecting their balance and resilience.

Knowledge of phytobenthos community structure, including its qualitative aspects, has been the basis for assessing the impact of pollution on many marine environments (Connell et al., 2008; Hardy et al., 1993; Liu et al., 2007). Some algae species have been used as biological indicators of pollution, either by their tolerance or

Table 1
Land use of pristine like (PLE) and urbanized (UBE) beaches in the coastal plain of southern Brazil.

Site		Coordinates	Urban area (km²)	Sewage outfall distance (km)	Surrounding urban cover (%)	Surrounding vegetation cover (%)	Nitrate (μM)	Ammonia (μM)	Phosphate (μM)	MPS mg l⁻¹
Torres	UBE	29°21′8.84″S, 49°43′48.60″W	15	0	90	2	0.96	1.07	0.35	22.5
	PLE	29°23′7.07″S, 49°45′25.05″W	0	2	1	95	1.20	2.13	0.35	21.7
Imbituba	UBE	28°13′50.24″S, 48°38′50.67″W	12	0	75	5	0.27	5.09	0.93	27.5
	PLE	28°12′7.89″S, 48°39′43.70″W	0	2	0	100	2.58	1.02	0.46	36.4
Florianópolis	UBE	27°23′15.71″S, 48°25′46.37″W	1	0	60	20	2.93	2.97	0.71	27.8
	PLE	27°25′35.82″S, 48°28′24.47″W	0.8	1	20	70	1.55	1.02	0.33	27.1
Balneário Camboriú	UBE	26°59′59.05″S, 48°36′9.03″W	50	0	90	10	0.44	6.15	0.33	50
	PLE	27°0′35.02″S, 48°34′12.91″W	0	3	0	100	1.60	2.73	0.31	31.5

Table 2
List of species sampled on the pristine like (PLE) and urbanized (URB) beaches.

	Torres		Imbituba		Florianópolis		Balneário Camboriú	
	PLE	UBE	PLE	UBE	PLE	UBE	PLE	UBE
Rhodophyta								
Acanthophora spicifera (M. Vahl) Børgesen					x			
Aglaothamnion fellipponei (M.A. Howe) Aponte, D.L. Ballantine and J.N. Norris			x		x	x		
Aglaothamnion uruguayense (W.R. Taylor) N.E. Aponte, D.L. Ballantine and J.N. Norris					x			
Amphiroa beauvoisii J.V. Lamouroux					x			
Arthrocardia flabellata (Kützing) Manza			x	x				
Bangia fuscopurpurea (Dillwyn) Lyngbye					x			
Bryocladia thyrsigera (J.Agardh) F. Schmitz			x	x			x	x
Bryothamnion seaforthii (Turner) Kützing							x	x
Centroceras clavulatum (C. Agardh) Montagne	x	x	x	x	x	x	x	x
Ceramium flaccidum (Harvey ex Kützing) Ardissone					x	x		
Ceramium gracillimum C. Agardh								x
Ceramium tenerrimum (G. Martens) Okamura					x	x		x
Chondracanthus acicularis (Roth) Fredericq in Hommersand, Guiry, Fredericq and Leister			x		x			
Chondracanthus teedei (Mertens ex Roth) Kützing	x			x	x			x
Colaconema codicola (Børgesen) H. Stegenga, J.J. Bolton, and R.J. Anderson					x			
Cryptopleura ramosa (Hudson) L. Newton			x					
Erythrotrichia carnea (Dillwyn) J. Agardh			x	x	x			
Gayliella flaccida (Harvey ex Kützing) T. Cho and L.J. McIvor					x			
Gelidiopsis variabilis (Greville ex J. Agardh) Schmitz								x
Gelidium crinale (Hare ex Turner) Gaillon	x	x						
Gelidium floridanum W.R. Taylor	x		x	x			x	x
Gelidium pusillum (Stackhouse) Le Jolis			x	x	x	x	x	x
Gracilaria cervicornis (Turner) J. Agardh								x
Gracilaria domingensis (Kützing) Sonder ex Dickie					x			
Gracilaria tepocensis (E.Y. Dawson) E.Y. Dawson	x				x			
Grateloupia cuneifolia J. Agardh	x		x					x
Grateloupia doryphora (Montagne) M.A. Howe					x			
Grateloupia filicina (J.V. Lamouroux) C. Agardh								x
Gymnogongrus griffithsiae (Turner) Martius	x	x	x	x	x			x
Herposiphonia secunda (C. Agardh) Ambronn					x			
Hypnea volubilis Searles								x
Hypnea musciformis (Wulfen) J.V. Lamouroux	x		x	x	x	x		x
Hypnea nigrescens Greville ex J. Agardh	x	x	x	x		x	x	x
Hypnea spinella (C. Agardh) Kützing			x		x			
Jania adhaerens J.V. Lamouroux					x	x		
Jania crassa J.V. Lamouroux	x	x	x	x	x	x	x	x
Jania ungulata (Yendo) Yendo				x				
Laurencia majuscula (Harvey) A.H.S. Lucas						x		
Lithophyllum stictaeforme (Areschoug) Hauck	x	x			x	x		
Neosiphonia howei (Hollenberg) Skelton and G.R. South						x		
Palisada perforata (Bory de Saint-Vincent) K.W. Nam							x	
Plocamium brasiliense (Greville) M.A. Howe and W.R. Taylor			x					
Polysiphonia scopulorum Harvey		x	x					
Pterocladiella capillacea (S.G. Gmelin) Santelices and Hommersand						x		
Pterosiphonia parasitica (Hudson) Falkenberg	x		x	x	x			
Pterosiphonia pennata (C. Agardh) Sauvageau								x
Rhodymenia pseudopalmata (J.V. Lamouroux) P.C. Silva					x			x
Sahlingia subintegra (Rosenvinge) Kornmann				x	x			
Stylonema alsidii (Zanardini) K.M. Drew					x			
Phaeophyceae								
Asteronema breviarticulatum (J. Agardh) Ouriques and Bouzon					x	x		
Bachelotia antillarum (Grunow) Gerloff			x			x		
Colpomenia sinuosa (Mertens ex Roth) Derbès and Solie						x		
Dictyopteris delicatula J.V. Lamouroux					x		x	x
Feldmannia irregularis (Kützing) G. Hamel					x	x	x	
Hincksia mitchelliae (Harvey) P.C. Silva					x	x		
Levringia brasiliensis (Montagne) A.B. Joly		x	x					
Padina gymnospora (Kützing) Sonder	x	x	x	x	x	x	x	x
Sargassum cymosum C. Agardh			x	x			x	
Sargassum stenophyllum Martius					x	x		
Sargassum vulgare C. Agardh								x
Sphacelaria furcigera Kützing					x	x		
Sphacelaria tribuloides Meneghini					x			
Chlorophyta								
Chaetomorpha aerea (Dillwyn) Kützing					x		x	
Chaetomorpha antennina (Bory de Saint-Vincent) Kützing	x		x	x		x	x	
Cladophora flexuosa (O.F. Müller) Kützing		x		x				x
Cladophora prolifera (Roth) Kützing		x			x			
Cladophora vagabunda (Linnaeus) Hoek	x	x			x	x	x	x
Cladophoropsis membranacea (Hofman Bang ex C. Agardh) Børgesen		x	x		x			
Codium isthmocladum Vickers					x			

Table 2 (continued)

	Torres		Imbituba		Florianópolis		Balneário Camboriú	
	PLE	UBE	PLE	UBE	PLE	UBE	PLE	UBE
Derbesia marina (Lyngbye) Solier			x					
Rhizoclonium riparium (Roth) Harvey				x	x			x
Ulva chaetomorphoides (Børgesen) Hayden, Blomster, Maggs, P.C. Silva, M.J. Stanhope and J.R. Waaland		x						–
Ulva lactuca Linnaeus	x	x	x	x	x	x	x	x
Ulva flexuosa Wulfen			x		x		x	

Table 3
Means of relative abundance of genera with more than 5% of coverage on at least one of the pristine like (PLE) and urbanized (UBE) beaches.

	Torres		Imbituba		Florianópolis		Balneário Camboriú	
	PLE	UBE	PLE	UBE	PLE	UBE	PLE	UBE
Centroceras	13.39	3.92	7.09	7.01	10.68	23.08	7.37	6.13
Chaetomorpha	5.68	0.00	6.93	1.08	0.30	0.17	4.00	0.00
Rocky surface	9.65	17.35	16.99	17.83	15.77	36.12	17.30	18.74
Asteronema	0.69	0.53	0.00	0.00	0.42	10.90	0.00	0.00
Gelidium	3.09	0.00	0.79	0.37	0.00	0.00	15.70	0.00
Hypnea	1.89	5.36	6.33	1.90	12.72	2.21	0.00	0.05
Jania	0.78	0.83	25.64	8.73	2.74	1.04	24.36	0.00
Sargassum	0.00	0.00	1.01	0.64	1.83	0.72	8.52	0.09
Ulva	51.13	56.72	22.33	57.67	39.66	13.58	8.90	67.61

Table 4
Permutational multivariate analysis of variance of macroalgae community data from rocky shores in urban and protected beaches (treatments) of four sites in southern Brazil.

Source of variation	df	MS	F-value	P (Monte Carlo)
Site	3	56576.58	13.720	0.001
Treatment	1	24888.40	2.013	0.013
Site × Treatment	3	12366.14	2.999	0.001
Residual	352	4123.69		

Pairwise a posteriori comparisons tests among levels of the factor Treatment within levels of the factor Site.

Site	t-Test	P (Monte Carlo)
Torres	1.573	0.037
Imbituba	1.729	0.001
Florianópolis	1.900	0.001
Balneário Cambouriú	2.379	0.001

df, degree of freedom; MS, means square.

sensitivity to pollutants (e.g. Castilla, 1996; Kindig and Littler, 1980). Moreover, several authors have linked the proportion of perennial and opportunistic species to the ecological status of ecosystems (e.g. Guinda et al., 2008; Krause-Jensen et al., 2007; Orfanidis et al., 2001). Consequently, the presence of sensitive species, i.e., those susceptible to pollution, and mainly *k* strategists, indicates a preserved environment, whereas the dominance of opportunistic species, or *r* strategists, is an indicator of poor ecological quality. Therefore, individuals close to the carrying capacity of the environment and whose ability to compete and survive is favored by natural selection are peculiar to more stable environments (MacArthur and Wilson, 1967). In this study *Chondracanthus acicularis* and *Gracilaria tepocensis*, occurred in at least two beaches classified as pristine like. Both species are considered as perennials (Guiry et al., 1987; Pinheiro-Joventino and Bezerra, 1980), i.e., their populations are stable, their colonization occurs after the settlement of opportunistic species, and their thalli, or at least part of them, persist for more than a year, satisfying the requisites of Feldmann (1966). At the other extreme, it has been suggested that certain macroalgae may be relatively resilient to organic contamination (Edwards, 1975). Some green algae are considered pioneer species in ecological succession, being able to colonize environments with high nutrient concentrations or near sewage discharges (Littler and Murray, 1975; Merrill

and Fletcher, 1991; Soltan et al., 2001). The proliferation of those species as a consequence of anthropogenic disturbance may, therefore, represent an indication of the impact of urbanization in coastal areas (Juanes et al., 2008). According to our results, the beaches under greater human pressure (UBEs) were dominated by green opportunistic species of *Ulva* or *Cladophora*. The species *Ulva flexuosa, U. chaetomorphoides* and *Cladophora flexuosa* were present only in urbanized beaches. *U. lactuca* was responsible for a coverage of 62.55% on UBEs, while on PLEs this coverage reached only 30.87%. The lower incidence of brown algae on urbanized beaches has been attributed to oil pollution at levels that may affect the reproductive processes of these algae (Jaenicke, 1977), as already suggested for the Guanabara Bay (Teixeira et al., 1987) and Santos Bay (Berchez and Oliveira, 1992), along the southeast Brazilian coast. *Sargassum cymosum*, an indicator of good environmental quality (Orfanidis et al., 2003), occurred at three sites, among which only one was classified as urbanized. In fact, species of *Sargassum* occurred mostly on pristine beaches in comparison to the urbanized ones, corresponding to 3.5% and 0.3% coverage, respectively.

The cases listed above illustrate that the benthic community structure is a result of several factors occurring simultaneously, being complex the determination of causal relations. The physical and biological heterogeneity of the rocky shore environments result
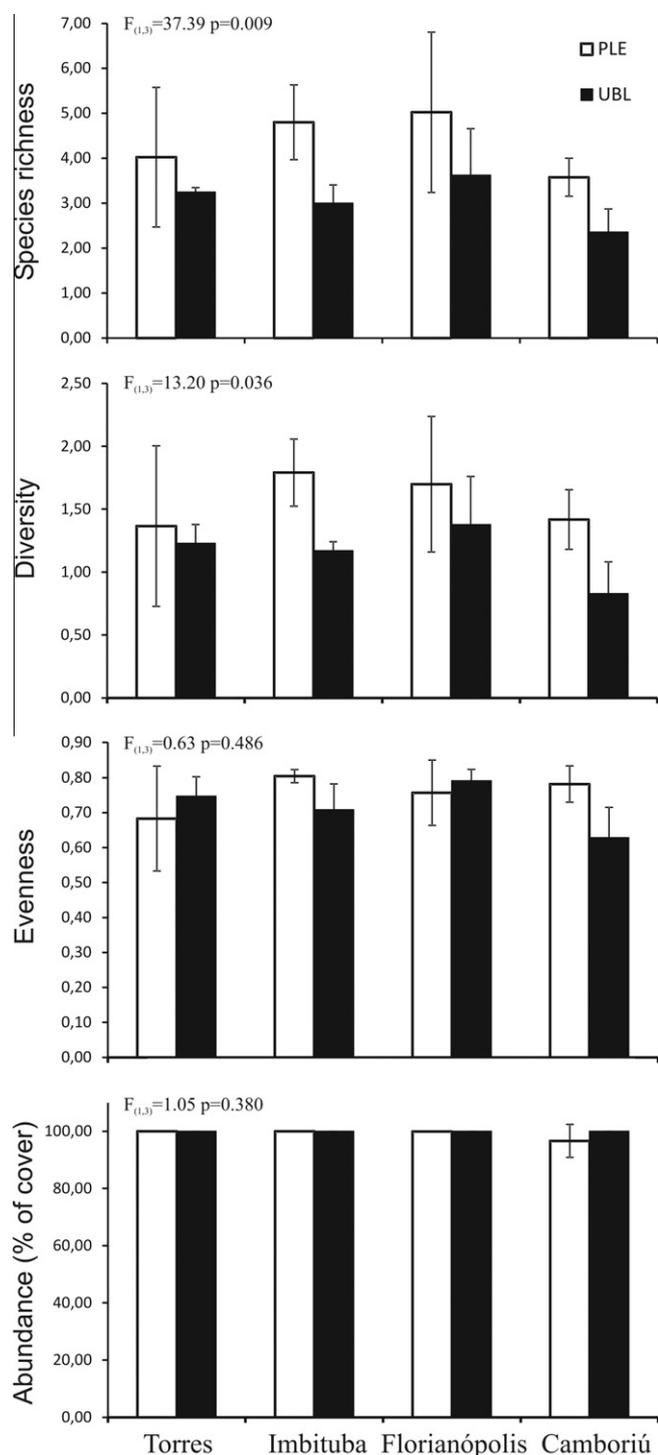
Fig. 2. Means (±SD) for species richness, Shannon–Wiener diversity, Pielou evenness, and abundance of phytobenthos communities from pristine (PLE) and urbanized (UBE) environments in southern Brazil.

shore environments also requires an analysis that takes into account quantitative approaches and utilization of synthetic ecological indices and community descriptors in a search of informative patterns of historical evolution based on statistical analysis.

The higher similarity of the urbanized sites, when compared to pristine beaches, can be explained by the dominance of a few opportunistic species of *Ulva* and *Cladophora*, present on all rocky shores sampled at more impacted areas. The pristine environments showed lower similarity, probably due to the occurrence of a greater number of exclusive species on those sites. In those environments, it is plausible that a greater heterogeneity of selective factors, as represented by herbivory and competition for substrate, light and nutrients, result in more heterogeneous communities (Britton-Simmons, 2006). In contrast, urbanized environments tend to have a smaller range of selective factors, such as the impact of urban effluents, which generally have very similar characteristics, thus making the biological communities present in those locations more homogeneous. According to multivariate and univariate analysis, the difference between phytobenthic community structure from pristine like and urbanized beaches was significant, suggesting that observed differences in community structure are due to pollution resulting from unplanned urbanization processes. The extreme anthropogenic contamination of marine habitats is often associated with the reduction of biodiversity, whether as a result of the decline in species richness, increased dominance of tolerant species (low evenness), or a combination of the two factors (Johnston and Roberts, 2009). Species diversity is one of the most important descriptors of a community, since it is related to its productivity and stability (Hooper et al., 2005; Worm et al., 2006).

Overall, human activity poses one of the major contemporary evolutionary driving forces causing changes in the composition and structure of algal communities, including the loss of diversity observed in urbanized environments in this study. Such changes have been detected along southern Brazil and other tropical and temperate coastal environments around the globe (Palumbi, 2001). Such activities also have the potential to represent important selective pressures, which favor opportunistic species, with both nitrophilous and ephemeral life cycles. Besides the introduction of high concentrations of nutrients in highly urbanized sites, other pollutants have been documented in coastal waters (Ellis et al., 1997; Haiyan and Stuanes, 2003; Kamalakkannan et al., 2004; Marsalek et al., 1997; Pagliosa et al., 2006; Sansalone and Buchberger, 1997) and their effects on algae can include a decrease in the concentration of chlorophyll-*a* and carotenoids, enzyme changes and lethality.

Considering the role of ecology in determining evolutionary processes (Schoener, 2011), we must reinforce the importance of impacts related to urbanization in determining the future of the coastal environments, given the undoubted increase in human occupation in these areas and their consequent deleterious effects. The presence or even the relative high abundance of some pollution indicator species, as *Ulva*, in pristine like areas indicates that the disturbs from human activities are no longer restricted to urban areas and that changes in water quality in coastal regions as a whole may already be producing large-scale changes. These results reinforce the need to conduct further studies on larger spatial scales, seeking pristine areas more distant from the sources of disturbances, to assess the degree of impairment of the community as a whole. We also highlight the need to produce time series data in south Atlantic to enable the evaluation of the temporal evolution of these impacts.

in high variability of combinations of physic and chemical parameters, thus contributing to environmental instability, unlike pelagic environments. Such variability provides a heterogeneous mosaic of microhabitats to rocky shores organisms (Murray et al., 2006). Following this reasoning, the presence or absence of particular species on rocky shores does not, in itself, represent an accurate descriptor of environmental quality. This is one of the main weaknesses of floristic surveys that take into account only the qualitative analysis of the species of a given location. The variability of rocky

References

Adams, S.M., 2005. Assessing cause and effect of multiple stressors on marine systems. Mar. Pollut. Bull. 51, 649–657.

Amado Filho, G.M., Barreto, M.B.B.B., Marins, B.V., Felix, C., Reis, R.P., 2003. Estrutura das comunidades fitobentônicas do infralitoral da Baía de Sepetiba, RJ, Brasil. Rev. Bras. Bot. 26, 329–342.

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecol. 26, 32–46.

Bahartan, K., Zibdah, M., Ahmed, Y., Israel, A., Brickner, I., Abelson, A., 2010. Macroalgae in the coral reefs of Eilat (Gulf of Aqaba, Red Sea) as a possible indicator of reef degradation. Mar. Pollut. Bull. 60, 759–764.

Ballesteros, E., Torras, X., Pinedo, S., García, M., Mangialajo, L., Torres, M., 2007. A new methodology based on littoral community cartography dominated by macroalgae for the implementation of the European Water Framework Directive. Mar. Pollut. Bull. 55, 172–180.

Berchez, F.A.S., Oliveira, E.C., 1992. Temporal changes in the benthic marine flora of the Baía de Santos, SP, Brazil, over the last four decades. In: Cordeiro-Marino, M., Azevedo, M.T.P., SantAnna, C.L., Yamaguishi-Tomita, N., Plastino, E. M. (Eds.), Algae and Environment, CETESB, São Paulo, pp. 120-131.

Boyce, D.G., Lewis, M.R., Worm, B., 2010. Global phytoplankton decline over the past century. Nature 466, 591–596.

Britton-Simmons, K.H., 2006. Functional group diversity, resource preemption and the genesis of invasion resistance in a community of marine algae. Oikos 113, 395–401.

Cardoso, P.G., Raffaelli, D., Lillebø, A.I., Verdelhos, T., Pardal, M.A., 2008. The impact of extreme flooding events and anthropogenic stressors on the macrobenthic communities' dynamics. Estuar. Coast. Shelf Sci. 76, 553–565.

Castilla, J.C., 1996. Copper mine tailing disposal in northern chile rocky shores: Enteromorpha Compressa (Chlorophyta) as a sentinel species. Environ. Monit. Assess. 40, 171–184.

Clarke, K.R., Warwick, R.M., 2001. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation, second ed. PRIMER-E, Plymouth.

Connell, S.D., Russell, B.D., Turner, D.J., Shepherd, S.A., Kildea, T., Miller, D., Airoldi, L., Cheshire, A., 2008. Recovering a lost baseline: missing kelp forests from a metropolitan coast. Mar. Ecol. Progr. Ser. 360, 63–72.

Cunha, P.R., Aragão P.J., Salvador, R.M., 2008. Saneamento em Santa Catarina e investimento PAC. Associação Brasileira de Engenharia Sanitária e Ambiental Seção Santa Catarina.

Edwards, P., 1975. An assessment of possible pollution effects over a century on the benthic marine algae of Co. Durham, England.. Bot. J. Linn. Soc. 70, 269–305.

Ellis, J.B., Revitt, D.M., Llewellyn, N., 1997. Transport and the environment: effects of organic pollutants on water quality. Water Environ. Manage. 11, 170–177.

FATMA, 2010. Relatório de balneabilidade da Fundação do Meio Ambiente - Governo do Estado de Santa Catarina.

Feldmann, J., 1966. Les types biologiques d'algues marines benthiques. B. Soc. Bot. Fr., 45–60.

Gorostiaga, J.M., Díez, I., 1996. Changes in the sublittoral benthic marine macroalgae in the polluted area of Abra de Bilbao and proximal coast (northern Spain). Mar. Ecol. Prog. Ser. 130, 157–167.

Grasshoff, K., Ehrhardt, M., Kremling, K., 1983. Methods of Seawater Analysis, second ed. Verlag Chemie, Weinheim.

Guinda, X., Juanes, J.A., Puente, A., Revilla, J.A., 2008. Comparison of two methods for quality assessment of macroalgae assemblages, under different pollution types. Ecol. Indic. 8, 743–753.

Guiry, M.D., Tripodi, G., Lfining, K., 1987. Biosystematics, genetics and upper temperature tolerance of Gigartina teedii (Rhodophyta) from the Atlantic and Mediterranean. Helgoländer Meeresun. 41, 283–295.

Haiyan, W., Stuanes, A.O., 2003. Heavy metal pollution in air-water-soil-plant system of Zhuzhou city, Hunan Province, China. Water, Air, and Soil Pollution 147, 79–107.

Hardy, F.G., Evans, S.M., Tremayne, M.A., 1993. Long-term changes in the marine macroalgae of three polluted estuaries in north-east England. J. Exp. Mar. Biol. Ecol. 172, 81–92.

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setälä, H., Symstad, A.J., Vandermeer, J., Wardle, D.A., 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecol. Monogr. 75, 3–35.

IBGE, Censo Demográfico 2010. Trabalho e Rendimento. Rio de Janeiro.

Jaenicke, L., 1977. Sex hormones of brown algae. Naturwissenschaften 64, 69–75.

Johnston, E.L., Roberts, D.A., 2009. Contaminants reduce the richness and evenness of marine communities: a review and meta-analysis. Environ. Pollut. 157, 1745–1752.

Juanes, J.A., Guinda, X., Puente, A., Revilla, J.A., 2008. Macroalgae, a suitable indicator of the ecological status of coastal rocky communities in the NE Atlantic. Ecol. Indic. 8, 351–359.

Kamalakkannan, R., Zettel, V., Goubatchev, A., Stead-Dexter, K., Ward, N.I., 2004. Chemical (polycyclic aromatic hydrocarbon and heavy metal) levels in contaminated stormwater and sediments from a motorway dry detention pond drainage system. J. Environ. Monit. 6, 175–181.

Kindig, A.C., Littler, M.M., 1980. Growth and primary productivity of marine macrophytes exposed to domestic sewage effluents. Mar. Environ. Res. 3, 81–100.

Kohler, K.E., Gill, S.M., 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. Comput. Geosci. 32, 1259–1269.

Krause-Jensen, D., Carstensen, J., Dahl, K., 2007. Total and opportunistic algal cover in relation to environmental variables. Mar. Pollut. Bull. 55, 114–125.

Littler, M.M., Murray, S.N., 1975. Impact of sewage on the distribution, abundance and community structure of rocky intertidal macroorganisms. Mar. Biol. 30, 277–291.

Littler, M.M., Littler, D.S., Brooks, B.L., 2010. The effects of nitrogen and phosphorus enrichment on algal community development: Artificial mini-reefs on the Belize Barrier Reef sedimentary lagoon. Harm. Algae 9, 255–263.

Liu, D., Bai, J., Song, S., Zhang, J., Sun, P., Li, Y., Han, G., 2007. The impact of sewage discharge on the macroalgae community in the Yellow Sea coastal area around Qingdao, China. Water Air Soil Pollut. 7, 683–692.

Lüning, K., 1990. Seaweeds: Their Environment, Biogeography and Ecophysiology. John Wiley, New York.

MacArthur, R.H., Wilson, E.O., 1967. The theory of island biogeography. Princeton University Press, Princeton.

Marsalek, J., Brownlee, B., Mayer, T., Lawal, S., Larkin, G.A., 1997. Heavy metals and PAHs in stormwater runoff from the Skyway bridge in Burlington, Ontario. Water Qual. Res. J. Can. 32, 815–827.

Merrill, J., Fletcher, R., 1991. Green tides cause major economic burden in Venice Lagoon, Italy. Appl. Phycol. 8, 1–3.

Murray, S.N., Ambrose, R.F., Dethier, M.N., 2006. Monitoring Rocky Shores. University of California Press, Berkeley, pp. 220.

Oliveira, E.C., Qi, Y., 2003. Decadal changes in a polluted bay as seen from its seaweed flora: the case of Santos bay in Brazil. Ambio 32, 403–405.

Orfanidis, S., Panayotidis, P., Stamatis, N., 2001. Ecological evaluation of transitional and coastal waters: a marine benthic macrophytes-based model. Medit. Mar. Sci. 2, 45–65.

Orfanidis, S., Panayotidis, P., Stamatis, N., 2003. An insight to the ecological evaluation index (EEI). Ecol. Indic. 3, 27–33.

Orfanidis, S., Papathanasiou, V., Gounaris, S., 2007. Body size descriptor of Cymodocea nodosa indicates anthropogenic stress in coastal ecosystems. Transit. Waters Bull. 2, 1–7.

Pagliosa, P.R., Fonseca, A., Barbosa, F.A.R., 2006. Evidence of systemic changes in trace metal concentrations in subtropical estuarine sediments as a result of urbanization. J. Coastal Res. SI39, 1078–1083.

Palumbi, S.R., 2001. Humans as the world's greatest evolutionary force. Science 293, 1786–1790.

Pereira, M.D., Schettini, C.A.F., Omachi, C.Y., 2009. Caracterização de feições oceanográficas na plataforma de Santa Catarina através de imagens orbitais. Rev. Bras. Geof. 27, 81–93.

Pinheiro-Joventino, F., Bezerra, C.L.F., 1980. Estudo de fenologia e regeneração de Gracilaria domingensis Sonder (Rhodophyta–Gracilariaceae) do Ceará. Arq. Cien. Mar 20, 33–41.

R development core team, 2011. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0. http://www.R-project.org. Accessed on 2 May 2011.

Reopanichkul, P., Schlacher, T.A., Carter, R.W., Worachananant, S., 2009. Sewage impacts coral reefs at multiple levels of ecological organization. Mar. Pollut. Bull. 58, 1356–1362.

Rogers, S.I., Greenaway, B., 2005. A UK perspective on the development of marine ecosystem indicators. Mar. Pollut. Bull. 50, 9–19.

Sanderson, E.W., Jaiteh, M., Levy, M.A., Redford, K.H., Wannebo, A.V., Woolmer, G., 2003. The human footprint and the last of the wild. Bioscience 52, 891–904.

Sandrini-Neto, L., Camargo, M.G., 2010. GAD: an R package for ANOVA designs from general principles. http://cran.r-project.org/web/packages/GAD. Accessed 2 May 2011.

Sansalone, J.J., Buchberger, S.G., 1997. Partitioning and first flush of metals in urban roadway storm water. J. Environ. Eng. 123, 134–143.

Schoener, T.W., 2011. The newest synthesis: understanding the interplay of evolutionary and ecological dynamics. Science 331, 426–429.

Scrosati, R., Heaven, C., 2007. Spatial trends in community richness, diversity, and evenness across rocky intertidal environmental stress gradients in eastern Canada. Mar. Ecol. Prog. Ser. 342, 1–14.

Soltan, D., Verlaque, M., Boudouresque, C.F., Fracour, P., 2001. Changes in macroalgal communities in the vicinity of a Mediterranean sewage outfall after the setting up of treatment plant. Mar. Pollut. Bull. 42, 59–70.

Strickland, J.L.H., Parsons, T.R., 1972. A Practical Handbook of Seawater Analysis, second ed. Fisheries Research Board of Canada, Ottawa.

Teixeira, V.L., Pereira, R.C., Júnior, A.N.M., Leitão Filho, C.M., Silva, C.A.R., 1987. Seasonal variations in infralitoral seaweed communities under a pollution gradient in Baía da Guanabara, Rio de Janeiro (Brazil). Cien. Cult. 39, 423–428.

Terlizzi, A., Fraschetti, S., Guidetti, P., Boero, F., 2002. The effects of sewage discharge on shallow hard substrate sessile assemblages. Mar. Pollut. Bull. 44, 544–550.

Tréguer, P., Le Corre, P., 1976. Manual d'analysis des seis nutritifs das l'eau de mer, second ed. Université de Bretagne Occidentale, Brest.

Vitousek, P.M., Mooney, H.A., Lubchenco, J., Melillo, J.M., 1997. Human domination of earth's ecosystems. Science 277, 494–499.

Wells, E., Wilkinson, M., Wood, P., Scanlan, C., 2007. The use of macroalgal species richness and composition on intertidal rocky seashores in the assessment of ecological quality under the European Water Framework Directive. Mar. Pollut. Bull. 55, 151–161.

Wilkinson, M., Wood, P., Wells, E., Scanlan, C., 2007. Using attached macroalgae to assess ecological status of British estuaries for the European Water Framework Directive. Mar. Pollut. Bull. 55, 136–150.

Worm, B., Barbier, E.B., Beaumont, N., Duffy, J.E., Folke, C., Halpern, B.S., Jackson, J.B.C., Lotze, H.K., Micheli, F., Palumbi, S.R., Sala, E., Selkoe, K.A., Stachowicz, J.J., Watson, R., 2006. Impacts of biodiversity loss on ocean ecosystem services. Science 314, 787–790.