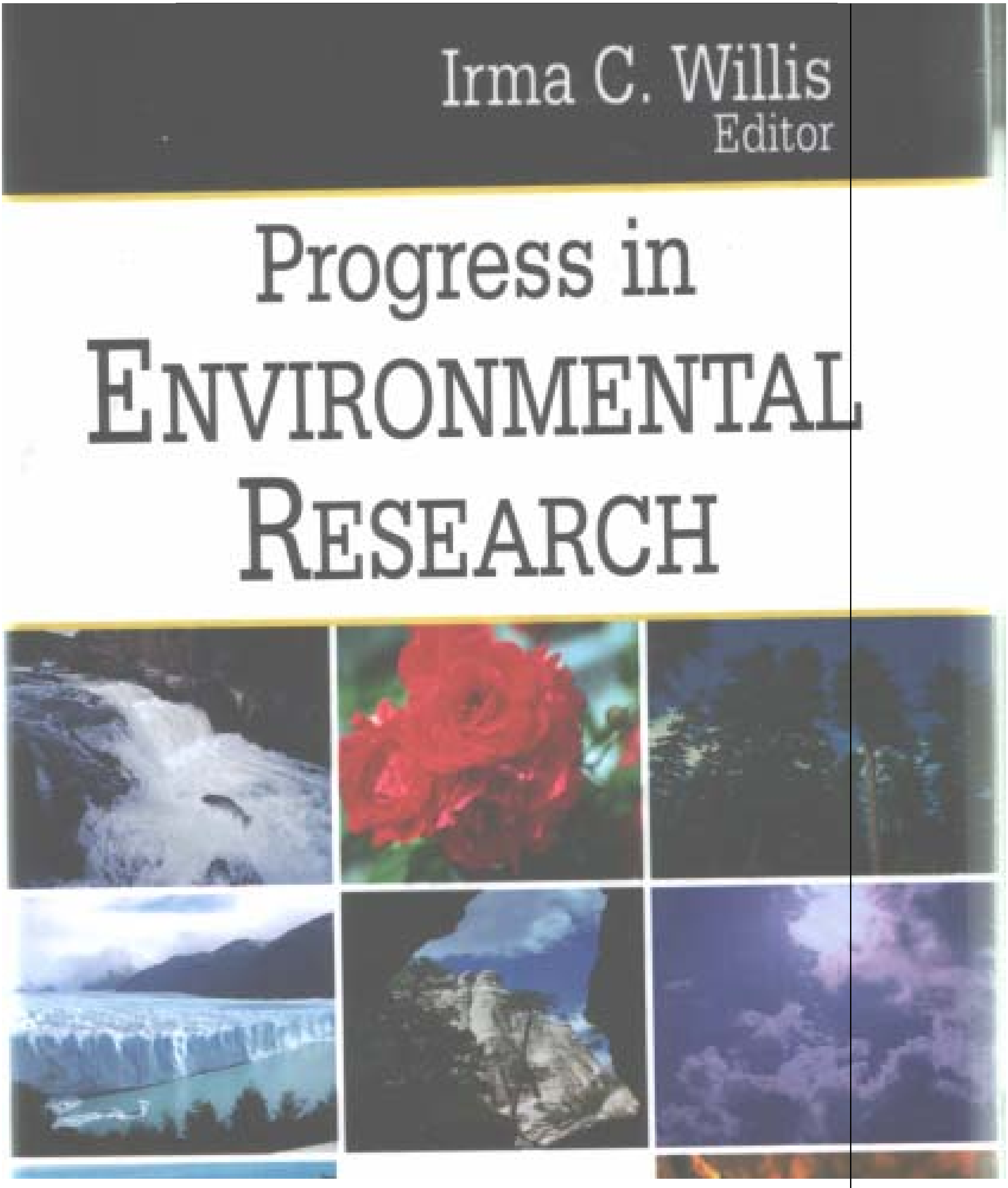
In: Progress in Environmental Research
Editor: Irma C. Willis, pp.1-
ISBN 978-1-60021-618-3



PROGRESS IN ENVIRONMENTAL RESEARCH

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Nova Science Publishers, Inc.
New York


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Progress in environmental research / Irma C. Willis (editor).
p. cm.
Includes index.
ISBN-13: 978-1-60021-618-3 (hardcover)
ISBN-10: 1-60021-618-8 (hardcover)
1. Environmental sciences--Research. I. Willis, Irma C.
GE70.P785 2007
628--dc22 2007006560

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Chapter 3

ENVIRONMENTAL PERTURBATION AND COASTAL BENTHIC BIODIVERSITY IN URUGUAY

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ABSTRACT

The Uruguayan coastal zone is bathed by the waters of the Southwest Atlantic Ocean (SWAO, 230 km) and the Río de la Plata Estuary (450 km), one of the largest estuaries in the world. The main tributaries of this estuary are the Paraná-Paraguay and Uruguay rivers, which drain the second largest basin in South America and provide the major source of freshwater runoff to the SWAO. Typical coastal ecosystems are sandy beaches with rocky points, sub-estuaries flowing along the Río de la Plata one and coastal lagoons in the Atlantic region. The estuarine portion is characterised by muddy sediments while sandy-shell debris are the dominant sediment type in the Atlantic portion. One of the most relevant features in this coastal zone is the interaction between the SWAO and the Río de la Plata waters, being that salinity is of primary importance in regulating the benthic biodiversity. In general, autochthonous fauna of the Uruguayan coast is characteristic of the temperate zone with temperate-cold and temperate-warm components, which correspond to the Patagonic biogeographic province, and show a break in the region of the Río de la Plata Estuary influence. Several studies had demonstrated that the Uruguayan coastal zone is under the effects of different kinds of human related stressors. Urbanisation, harbour, shipping and industrial activities are the main perturbation factors for the Río de la Plata portion, while, agricultural and tourism affect preferentially the Atlantic one. In addition, these studies had shown that petroleum hydrocarbons, heavy metals and the organic enrichment of bottom sediments have a direct influence on biodiversity patterns. Furthermore, some morphological anomalies have been detected in benthic foraminifera, which seem to be related to the heavy metal and organic content of the sediments. Despite the existence of a clear salinity gradient in this estuarine area, the utilization of different approaches, together with the integration of physical, chemical and biological data, demonstrated the occurrence of an environmental quality gradient with the improvement of conditions from the inner stations of Montevideo Bay to the outer coastal ones. Recent studies warned about the presence of non indigenous invasive species in both zones of the Uruguayan littoral, however, the degree of incidence seems to be greater in the estuarine portion. Their introduction would be related to the discharge of ballast water, and their distribution determined by salinity patterns. The available information about marine biodiversity and environmental perturbation in the Uruguayan coastal zone was improved during the last decades; however, it is still restricted to isolated areas along it and to some aspects of aquatic ecosystems. The implementation and development of integrative baseline studies on these topics are highly relevant, in order to contribute to the conservation of benthic biodiversity in the coastal zone of Uruguay.

INTRODUCTION

Coastal areas are the most dynamic portions of the global ecosystem, and also, the most subjected to population concentration. Natural and anthropogenic impacts, poor planning and management of the conflictive activities developed in coastal areas have major influence on the resident biota, functioning and productivity of all coastal ecosystems, which in turn, affect directly the global biodiversity and the whole ecosystem health. The consciousness about the consequences of man-induced impacts on marine ecosystems has increased during the last years, and so, the need of applying effective measures to prevent or stop them (Howarth et al., 2000). Regarding this necessity, the development of research to identify and to attenuate the impact is fundamental, aiming the prevention of socio-economical prejudices and health risks (Constanza et al., 1997). Furthermore, the search of solutions for the conflictive uses of an environment should lay down on a solid scientific base that permits to obtain information about how it works, which are its responses to external impacts and its capacity of recovery, in order to establish priorities and make the best decisions for its management and conservation.

Nowadays, an important challenge for the human beings is the maintenance of natural coastal systems that provide goods and services. Within the coastal area, estuaries are highly dynamic systems with an intrinsic strong natural variability and consequently, a high level of stress for the inhabitant organisms. Estuaries are environments where the meeting of water masses with contrasting physical and chemical properties occurs, promoting the establishment of strong horizontal and vertical gradients. Hydrographical regimes are very variable among estuaries and also, temporally, within a particular one. Then, populations and species associated with these relatively complex environments should develop physiological adaptations to deal with this high natural variability (Day et al., 1989; Lalli & Parsons, 1997). In addition, they are zones of transition between the marine and the terrestrial environments that perform essential ecological functions, including nutrient degradation and regeneration, as well as, the control of nutrient, water, particles and organisms fluxes, from and to the continental margins, rivers and oceans. The ecological and economical importance of estuaries and coastal zones in general, is worldwide recognized. Besides, they are the most complex, diverse and productive areas on earth, they are the most vulnerable ones, due to the competing demands between the natural and the socioeconomic systems. The former, is represented by the physical, chemical and biological components, whereas, the latter is represented by anthropogenic activities and the infrastructure needed to develop them (Constanza et al., 1997).

Within aquatic ecosystems, the benthic environment has an important function as an efficient natural trap for several substances, and also, as a natural regulator of sedimentary biogeochemical processes. Bottom sediments constitute a source of nutrients for the water column above them, leading to benthic-pelagic coupling and influencing primary productivity (Jorgensen, 1996). In addition, sediments accumulate natural and anthropogenic products from the overlying water, being that, they may act both, as a sink or a source of contaminants. Furthermore, marine sediments could act as a non-point source of contaminants causing adverse effects to organisms and to human health through trophic transfer. Heavy metals, hydrocarbons and other kind of pollutants derived from anthropogenic activities, produce perturbations in an ecosystem changing its abiotic conditions and affecting its biota. In these sense, the analysis of the structure of the benthic communities is an important tool to describe changes in space (with applications on point source pollution monitoring) and time (with applications on the description of changes concerning the state of the marine system) (Heip, 1992).

Benthic organisms have been used extensively as indicators of environmental status and trends. Numerous studies had demonstrated that they respond predictably to many kinds of natural and human induced stressors (López-Jamar, 1985; Ritter & Montagna, 1999; Borja et al., 2000). Benthic macrofauna can be suitable used to describe changes in a particular environment because the organisms are relatively sedentary and have comparatively long life spans (Thouzeau et al., 1991). In addition, the macrofauna consists of different species that exhibit different tolerances to stress (Dauer, 1993), allowing the monitoring of the environmental quality. Exposure to hypoxia is typically great in near-bottom waters and anthropogenic contaminants often accumulate in sediments where the benthos lives. The limited mobility of most adult macrobenthic organisms has advantages in environmental assessment, due to, unlike most pelagic fauna; their assemblages reflect local environmental conditions (Gray, 1979). The macrobenthos can also be employed to understand the incidence of certain ecological factors, such as predation and competition for space or food, which are responsible for the structure and productivity of benthic communities (Saiz-Salinas, 1997). In benthic ecosystems, the variability of environmental conditions has effects on species composition, which are well established. The distribution of invertebrate species is strongly influenced by the physico-chemical environment over a wide range of scales (Hall et al., 1994). Regionally, the species changes over geographic gradients, both latitudinal and longitudinally (Heip et al., 1992; Hillebrand, 2004). The most important variables identified, in the literature, as primary influencing factors are depth, temperature, water movement patterns and sediment type (Rees et al., 1999; Sanvicente-Anorve et al., 2002). Sediment mobility together with the amount and quality of the organic matter presented, have also been evocated as influential factors at the regional scale (Heip et al., 1992). Elsewhere, salinity can be a determinant factor of large-scale invertebrate composition (Giberto et al., 2004). On a minor scale, sediment composition and other related factors seems to be of variable importance in determining benthic composition assemblages (Brown et al., 2002; Thrush et al., 2003). At a local scale, seabed morphology is also important for structuring benthic patterns (Giménez et al., 2006), although, there may be multiple variables involved in complex species-environment relationships at this scale.

The reviews developed by Masello & Menafra (1996) and Calliari et al. (2003) provide detail information about characteristics, components and dynamics of the benthic communities in the Uruguayan coastal zone. In general, these authors coincided in the fact that there are a limited number of investigations developed, and they focused mainly on intertidal sandy beaches. Nevertheless, some recent works had analysed several aspects of the intertidal and subtidal macrobenthic communities in the Río de la Plata and the Atlantic coast providing some new knowledge, which permit recognize some distributional patterns, scales associated with this patterns and several environmental variables that regulate the structure and dynamics of these benthic communities (Rodríguez- Capítulo et al., 2003; Giménez et al., 2005, 2006; Defeo & McLachlan, 2005; Carranza et al., *in press*; Lercari & Defeo, 2006).

This chapter summarised the main natural and anthropogenic stressors for macrobenthic fauna in the Uruguayan coastal zone and its effects at the population and community levels. We focused specifically, on the shallow sublittoral benthic fauna of the coastal area of Montevideo, which is the most studied in relation to anthropogenic impacts. We also commented some published and original data about biological invasions in the Uruguayan coastal zone.

URUGUAY AND THE URUGUAYAN COASTAL ZONE

Uruguay is a small country situated in South America (30º-35º S and 53º- 58º W), between Argentina and Brazil (Figure 1). In Table 1, are presented the most important territorial, social, and economic indicators of Uruguay. The weather is subtropical temperate with an annual mean temperature of 16° C and 1000 mm of precipitation. It is situated in the lower portion of the Río de la Plata Basin, the second most important of South America after the Amazonian Basin. The Río de la Plata Basin, with an area of 3.100.000 km^2, includes five countries (Brazil, Argentina, Paraguay, Uruguay and Bolivia) and a population of ca. 120.000 million inhabitants. It is the most industrialised area of South America. The most important rivers are the Paraná (ca. 4000 km) and the Uruguay (ca.1.600 km), which flowed in the Río de la Plata Estuary, with an annual river flow of 16.000 m^{3}s^{-1} and 3.900 m^{3}s^{-1}, respectively (Tundisi et al., 1999). With a total extension of 680 km, the Uruguayan coast has 450 km lie on the Río de la Plata and 230 km on the Atlantic Ocean (Figure 2). One of the most important aspects in this coastal ecosystem is the interaction between the freshwater from the Río de la Plata Estuary and the Atlantic Ocean, which promotes a saline gradient along the coast. These coastal zone can be grossly divided in three different sub-environments according to the prevalent haline conditions: freshwater (between Punta Gorda and Montevideo corresponding to the upper and middle Río de la Plata, with a salinity between 1 and 10), estuarine (between Montevideo and Punta del Este in the lower Río de la Plata, with a highly salinity variation, between 1 and 33) and oceanic (between Punta del Este and Barra del Chuy in the Atlantic Ocean) (Figure 2) (Guerrero et al., 1997; Nagy et al., 1997, 2002).

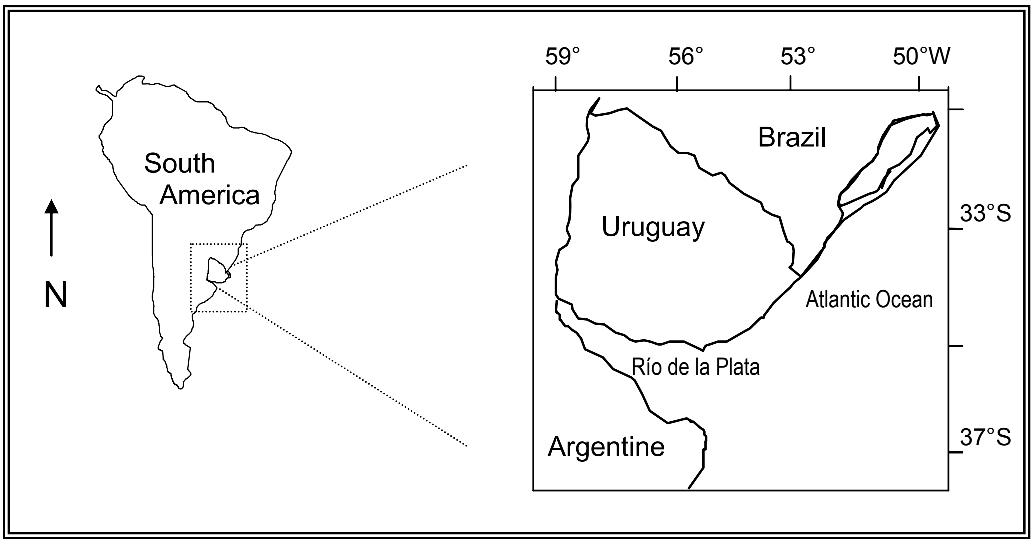


Figure 1. Geographic position of Uruguay

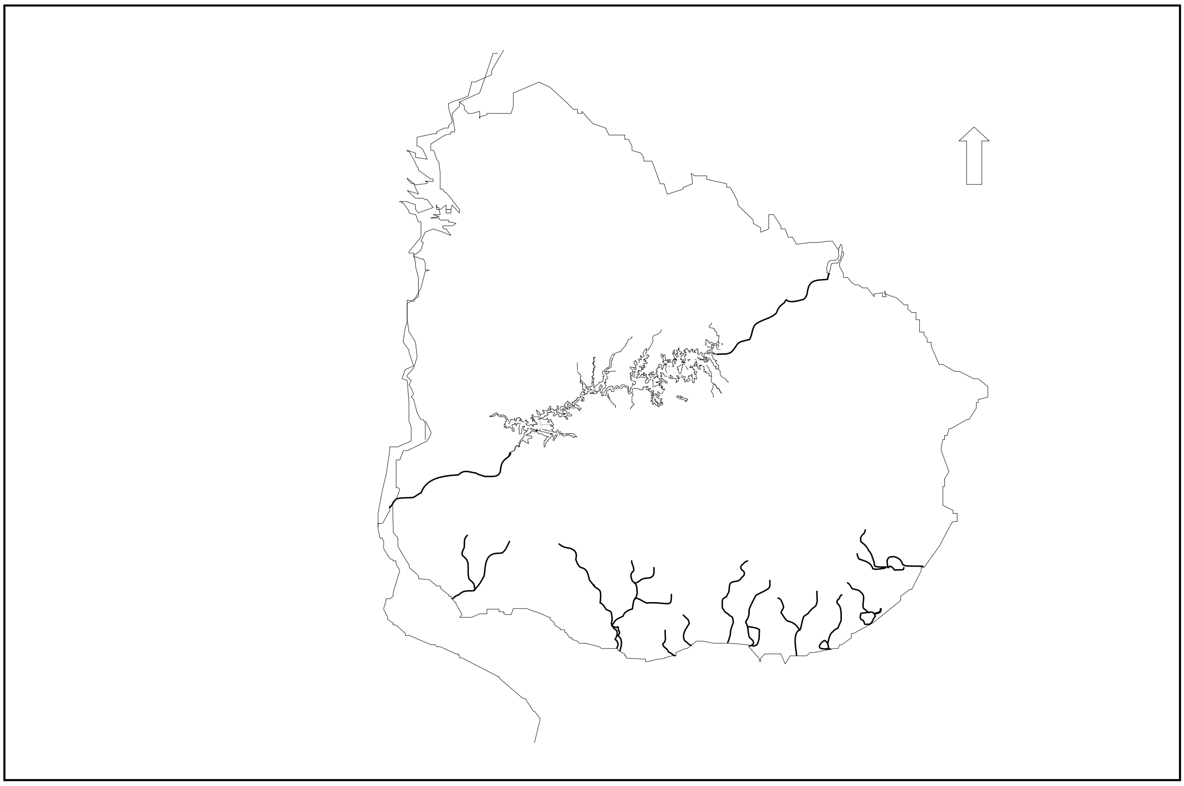


Figure 2. Uruguayan coastal zone and main coastal rivers, streams and lagoons. a)- San Juan river, b)- Santa Lucía river, c)- Pando stream, d)- Solís Chico stream, e)- Solís Grande stream, f)- del Potrero stream, g)- Maldonado stream, h)- José Ignacio Lagoon, i)- Rocha Lagoon, j)- Valizas stream.

Table 1. Main territorial and social-economical characteristics of Uruguay

Location	30º to 35 º S – 53º to 58º W
Area*	176.000 km^2 (Terrestrial surface) 137.567 km^2 (jurisdictional waters)
Borders	Argentine (W) and Brazil (E, NE).
Extension coastal zone**	680 km
Population***	3.305.723 inhabitants (2005)
Pop. distribution at coastal zone	73% live in coastal zones
Pop. density in the country****	19 inhabitants km^2
Pop. density in the costal zone****	74 inhabitants km^2
Pop. density in Montevideo city****	2600 inhabitants km^2
PBI*****	12.329
PBI distribution	Fish and agriculture (9%), Industry (17%), Services (58%), Others (16%)

* Include terrestrial surface and jurisdictional waters;
** Include Río de la Plata and Atlantic coast;
*** www.ine.org.uy;
**** Populations projections to 2001 (www.ine.org.uy);
**** millions of US$ (year 2003).

The Río de la Plata (35°00´-35°10´ S and 55°00´-58°10´ W) may be defined as a funnel-type coastal plain tidal river with a semi-enclosed shelf sea at the mouth and a surface area ca. 36.000 km^2. The Paraná and Uruguay Rivers feed freshwater into the Río de la Plata Estuary with a seasonal and interannual discharge variation between 22.000-28.000 m^3 s^{-1}, and with extreme values during El Niño (> 30.000 m^3 s^{-1}) and La Niña (< 20.000 m^3 s^{-1}) (Nagy et al., 2002). Tides are semidiurnal with amplitude of about 40 cm on the Uruguayan coast. Features such as salinity, depth of the halocline and vertical mixing vary with astronomic tidal oscillation on an hourly basis, while axial winds influence water height and salinity variations on a daily basis. The river flow governs monthly to interannual variations. The mean water temperature (1981-1987) is 15 °C (CARP, 1990). Fine grained sediments are confined to the upper and middle Río de la Plata, while sand covers almost the entire outer Río de la Plata and the adjacent continental shelf (López-Laborde & Nagy, 1999). In this area, the dominant coastal features consist of sandy beaches and estuaries (López-Laborde et al., 2000). At least 67% of the Uruguayan population lives in this area, so, the main industrial and urban activities are concentrated there. Several polluted sub-estuaries are located in it and consequently habitat degradation is growing with an accelerated rate. Sedimentation and final deposition of dredging sediments are other important problems recently perceived. Among the main pollutants introduced in the aquatic environment we should mentioned heavy metals, petroleum hydrocarbons, surfactants, DDT´s and PCB´s (Moyano et al., 1993, Moresco & Dol, 1996; Cranston et al., 2002; Muniz et al., 2002, 2004a, b, 2005a; Viana et al., 2005). In addition, this region shows conditions of moderate eutrophication (Nagy et al., 2002). Recently, the increment of harbour activities and the transoceanic transport have called the attention for another problem, the introduction of exotic species with ballast water (Clemente & Brugnoli, 2002; Brugnoli et al., 2005; Muniz et al., 2005b; Brugnoli et al., 2006).

The most relevant characteristics of the Atlantic zone are the large sandy beaches and the presence of coastal lagoons. Coastal lagoons in Uruguay are usually shallow water bodies receiving variable amounts of fresh water. Due to their geomorphological and hydrological characteristics, environmental conditions in the lagoons undergo frequent fluctuations on a daily and seasonal basis. This instability causes changes in the distribution of benthic species and the structure of communities, which sometimes are accentuated by anthropogenic influences. These ecosystems are important sites, functioning as nursery and feeding areas for several aquatic species of economic value. In this area, is localized the “Bañados del Este” Biosphere Reserve (MAB, UNESCO), the most important natural protected area of the country. Recently, a significant demographic growth was detected in the basins of these lagoons; therefore, some environmental problems were reported such as eutrophication, habitat degradation, the occurrence of exotic species and the increment in agriculture activities (Nión et al., 1979; Rodríguez-Gallego et al., 2003; Muniz et al., 2005b; Borthagaray et al., 2006). Sandy beaches located on the Atlantic coastal zone are interrupted by rocky points or freshwater systems, both, natural and artificial ones. The mainly artificial freshwater system that flows directly into the Atlantic Ocean is the Andreoni Channel, situated 800 m south of La Coronilla (Defeo & De Alava, 1995). It discharges pesticides, high concentration of suspended solids (Mendez & Anciaux, 1995) and freshwater from rice plantations that affect the benthic intertidal community adjacent to its mouth. For a review of these effects and detailed information of the effects of the channel over the intertidal fauna see: Defeo & de Alava (1995, 2002); Defeo et al. (1996); Lercari & Defeo (1999); Lercari et al. (2002); Calliari et al. (2003); Lercari & Defeo (2003).

Some Biogeographical Aspects of the Uruguayan Region

According to the influence of temperature, latitudinal gradients, local circulation patterns and water properties combined with the spatial arrangement of the continents and oceans, is possible to divide the oceans into a series of provinces or biogeographic regions with characteristic biological assemblages. These regions show high degree of endemism of its flora and fauna (Balech, 1954, 1964; Boschi, 1976, 2000) being characterised by specific organism assemblages that are based on qualitative data about the distribution of plant and/or animal species, primary, production, biomass or trophic relationship. The latitudinal gradient of species diversity is the most robust pattern in biogeography. Specifically, the gradient in species richness, a negative correlation between the number of species and latitude (Pianka, 1966; Rhode, 1992; Brown & Lomolino, 1998) is seen in most of taxonomic groups: vascular plants, algae, birds, mammals, reptiles, amphibians, fishes, arthropods, fungi and many others (e.g. Fischer 1959; Gaines & Lubchenco, 1982; Willig & Selcer, 1989; Currie, 1991; Rex et al., 1994; Blakburn & Gaston, 1996).

From Trinidad (~ 10°N) to Cape Horn (~ 55°S) the South American margin changes from tropical to cool temperate. It includes a great range of ecosystems including the enormous Amazonian delta, the coral reef of Fernando de Noronha Island, mangrove swamps and tidal marshes, hypersaline and brackish lagoons as well as narrow and wide continental shelves. Among the first studies about zoogeography in South America we can mentioned those developed by d´Orbigny (1835-43) and Dana (1853), who realised the occurrence of faunistic regions, centres of dispersion and speciation, and also the relationship between the regional marine fauna and different ocean temperature zones. Dana (1853), include within zoogeographic regions associated to the Uruguayan coastal zone, the Uruguayan Province (temperate, located 30º S northwards from la Plata Cape) and the Platense Province (sub-temperate, located in the mouth of the Río de la Plata Estuary), indicating a essential faunistic change in the Río de la Plata region. Moreover, Balech (1954), recognize 4 zoogeographic regions in the Southwest Atlantic (Antillana, Sur de Brasil, Argentina, Magallánica). Boltovskoy (1970) published the major biogeographic divisions in the Southwestern Atlantic included four domains in the neritic zone: a tropical domain (from equator to 20º S), subtropical (20º to 30 - 35º S), transition (30 - 35º S to 46 - 48º S) and subantartic (46 - 48º S to 55 -60º S). Recently, Sealey & Bustamante (1999), mentioned the warm-temperate Southwestern Atlantic Province, defined to the south by the Valdes Peninsula (41ºS) and to the north by Cabo Frío (Brazil 23º S), comprising a coast line of 8.154 km length. This province has a warm temperate climate that constitutes a transition between the cold temperate South America Province and the Tropical Southwestern Atlantic Province. The transitional, that is ascribed to the Argentina Biogeographical Province (and its boundaries with subtropical and subantartic zones) is the most dynamic, and their limits vary widely seasonally and multiannually. This zone is also defined as an area of mixing of subtropical and subantartic fauna (Boltovskoy et al., 1999), being characterised by a high diversity of fishes and invertebrates, numerous colonies of sea mammals and birds that belong to South Brazilian and Sub Antartic regions, so, the expected degree of endemism is very low.

The marine domain of Uruguay is made up by the Río de la Plata and the adjacent shelf and shares ecosystems with Brazil and Argentina. According to several authors, the Uruguayan coastal zone belongs to the transitional zoobiogeographic division, located between subtropical and subantartic zones (Balech, 1954; Palacio, 1982; Boltovskoy, 1970) (Figure 3). In the classification of Sealey & Bustamante (1999), the Uruguayan coastal zone includes the Uruguayan - Buenos Aires coastal shelf and the Río de la Plata coastal biogeographic province constituted by a warm-temperate biota, rather than a cold-temperate one. The Río de la Plata represents a major geographic feature of the Argentine Biogeographical Province, located approximately in its centre. Its presence imposes a not completely proved role of biogeographical barrier. According to Boltovskoy & Wright (1976), considering the benthic foraminifera communities, the Argentine Province is an area of temperate water characterized by the dominance of *Buccella peruviana.* This species also occurs along the Pacific coast of South America but its true domain is in the Argentina Province. Other species also characteristics of this province are *Bolivina compacta*, *Bulimina patagonica*, *Buliminella elegantissima*, *Elphidium* spp. *Epistominella exigua*, *Milolinella subrotunda*, *Quinqueloculina seminulum*, *Pyrgo rigens* and others, but the dominant species is *Bucella peruviana*. Murray (1991) divided the estuary of the Rio de la Plata in three zones according to the occurrence of living foraminifera: fluvial (salinity < 10) with *Haynesina germanica*, fluvial marine (salinity 10 - 30) with *Ammonia beccarii* and marine (salinity > 30) with *Buliminella elegantissima* (Boltovskoy & Lena, 1974).

Furthermore, the great majority of the intertidal fauna species have in the Uruguayan coastal zone their dispersion limits, restricting their latitudinal distribution. This characteristic is principally due to the water mean temperature, with presence of temperate cold and temperate warm faunal limits, and the existence of the Río de la Plata Estuary (salinity variation) that appears to act as an ecological barrier (Maytia & Scarabino, 1979; Escofet et al., 1979). Costal communities off Río de la Plata are characterised by the absence of the typical cold-temperate south Atlantic forms, the absence of significant invertebrate predators from the rocky intertidal zone, as well as the presence of a community dominated by the yellow clam (*Mesodesma mactroides*) in exposed sandy beaches (Sealey & Bustamante, 1999). Some authors ascribe these attributes to the river while others consider that this barrier does not exist. Recent information suggest that rivers act as an “intermittent barrier” for the distribution of flora and fauna (see Mianzan et al., 2001), such information must be taken into consideration when analysing species richness spatial patterns. The subtidal macrofauna of the Río de la Plata Estuary was less diverse than that of adjacent marine areas but shows higher densities and biomass. Bottom type, salinity and the presence of a turbidity front are the main physical variables structuring benthic communities of this estuary and coastal adjacent zone (Giberto et al., 2004). Table 2 synthesise the most representative taxa (abundance, biomass, with ecological or economic relevance) of the macrofaunal benthic community (intertidal and subtidal) of the Uruguayan coastal zone, according to studies developed in the last 30 years.

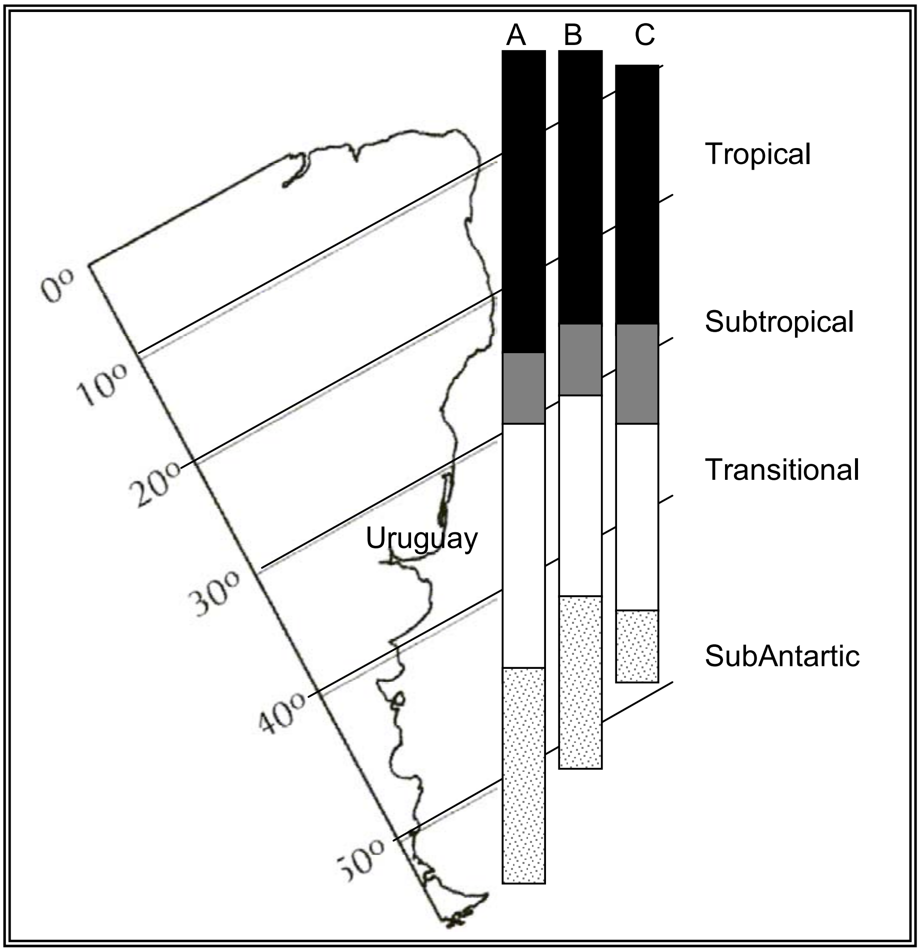


Figure 3. Biogeographic divisions of coastal South America according to Dana, 1853 (A), Boltovskoy, 1970 (B) and Palacio, 1982 (C) (after Boltovskoy et al., 1999).

Table 2. Benthic taxa most representative of coastal Uruguayan zone

Species	Intertidal (*)		Subtidal (**)	Río de la Plata				Atlantic coast		
	SS	HS		E	SE	RP	SB	CL	RP	SB
Plantae										
Juncus acutus	X				X		X			
Ruppia maritima	X							X		
Spartina longispina	X				X		X			
Spartina montevidensis	X				X		X			
Algae										
Codium sp.		X							X	
Enteromorpha sp.		X				X			X	
Hypnea musciformis		X							X	
Polysiphonia sp		X				X			X	
Porphyra sp.		X							X	
Ulva lactuca		X							X	
Artrophoda										
Crustacea										
Artemesia longinaris	X		X							
Chtamalus bisinuatus		X				X			X	
Chasmagnathus granulata	X				X			X		
Cyrtograpsus angulatus		X			X	X		X		
Eubalanus amphitrite		X							X	
Eubalanus improvisus		X				X		X	X	
Emerita brasiliensis	X									X
Excirolana armata	X						X			X
Excirolana brasiliensis	X									X
Lygia exotica (Esp.)		X		X						
Neomysis americana (Csp.)			X	X	X					
Ocypode cuadrata	X									X
Panaeus paulensis								X		
Seriolis marplatensis	X		X							
Annelida (Polychaeta)										
Ficopmatus enigmaticus (Esp.)		X	X		X			X		
Heteromastus similis	X				X			X		
Hemipodus olivieri	X									X
Laeonereis acuta	X				X					
Alitta succinea	X				X		X		X	X
Nephtys fluviatille	X				X			X		
Onophis tenuis			X							
Mollusca										
Adelomenon brasiliana (Fsp.)			X					X		
Brachidontes darwinianus		X		X		X			X	
Brachidontes rodriguezi		X		X		X				
Buccinanops cochildium			X	X						
Corbicula fluminea (Esp.)	X		X	X						

Corbula patagonica			X							
Donax hanleyanus (Fsp.)	X									X
Erodona mactroides	X		X		X		X	X		
Heleobia australis	X		X	X	X			X		
Limnoperna fortunei (Esp.)		X		X	X					
Littoridina australis		X				X				
Mactra isabelleana	X		X	X	X		X			
Mesodesma mactroides (Fsp.)	X									X
Mytella charruana		X		X	X	X				
Mytilus edulis platensis (Fsp.)		X		X		X			X	
Olivancillaria vesica auricularia	X				X					X
Perna perna		X							X	
Rapana venosa (Esp.)		X	X	X		X				
Siphonaria lessoni		X								
Tagelus plebeius	X				X		X	X		
Echinodermata										
Encope emarginata	X		X							

* supra, meso and infralitoral;

** under infralittoral zone (Río de la Plata and adjacent shelf waters);

SS: Soft substrate; HS: Hard Substrate, E: Estuary; SE: Sub-estuary; RP: Rocky point; SB: Sandy beach; CL: Coastal Lagoon.; Fsp: Fisheries species; Esp: Exotic species; Csp. Criptogenic species.

From Scarabino et al. (1975), Escofet et al. (1979), Maytia & Scarabino (1979), Nión (1979), Boschi (1988), Riestra et al. (1992), Masello & Menafra (1996), Muniz & Venturini (2001), Orensanz et al. (2002), Giberto et al. (2004), Calliari et al. (2001); Brugnoli et al. (2005), Giménez et al. (2005), Carranza et al. (*in press*).

THE MONTEVIDEO COASTAL ZONE: ITS BIOTA AND MAIN STRESSORS

Montevideo Bay (MB) and the adjacent coastal zone (AZC) are located in the middle portion of the Río de la Plata Basin. Data about surface water temperature indicate the existence of a hot season from December to March and a cold season from June to September with a temperature difference between them of approximately 10 °C (Guerrero et al., 1997). Mean surface salinity of the Río de la Plata waters near Montevideo varies between 5 and 10. Hourly variations in salinity, halocline depth and vertical mixing, are coupled with tidal oscillations, which are semi-diurnal. The river discharge governs monthly and interannual variations of salinity and turbidity, being that the maximum of turbidity is generally associated with the limit of saline intrusion. Furthermore, it is related to gravitational circulation and flocculation of clay particles, and has a high suspended organic matter load (López-Laborde & Nagy, 1999). The MB with an area of 10 km^2 has a mean depth of 5 m, excepting in the navigation channels. In relation to geological features, it is characterized by the presence of Precambric outcrops with more recent material derived from them in some regions (Cardellino & Ferrando, 1969). Modern sediments consist of silt and clay (Muniz et al., 2002). Water circulation within the MB is clockwise, being determined mainly by winds from NE and W-SW. This area has a great economic importance for Uruguay as a navigation and commercial route. Its waters are used for sport activities and the extraction of artisanal and industrial fish resources, among other demands.

The BM and its ACZ are under the influence of different anthropogenic impacts such as the input of urban and industrial effluents, oil refining processes and maritime traffic. Water quality of Montevideo Bay is highly deteriorated due to several point and non-point sources and harbour activities. This subsystem, of 10 km^2 of area, receives most of its water from the Río de la Plata, with minor contribution from three little creeks and several sewage outlets heavily charged in heavy metals, nutrients and BOD (Muniz et al., 2002, 2004a, b). Within the bay is the ANCAP oil refinery, La Teja dock, the Central Batlle electric power plant (UTE) and the Montevideo Harbour. Untreated or partially treated municipal effluents, produced by one and a half million people are discharged towards the east portion of BM, in the Punta carretas zone, where the largest sewage pipe of the country is located. The standing stock of particulates generally attains its peak concentration in the late summer and early fall, reflecting a trend towards a unimodal seasonal pattern in the development of autotrophic biomass, i.e. chlorophyll-*a* concentration. In summer, water conditions are clearly hypertrophic with high concentrations of ammonium (up to 120 micro M), chlorophyll *a* (> 100 µg/g chl *a* l^{-1}), and hypoxia (0-20 % of oxygen saturation) as a consequence of domestic effluent discharges and the relative high residence time of the water mass. Depth in the navigation channels that permit the access to Montevideo Harbour can reach between 9 and 11 m because they are frequently dredged. Is estimated that 700000 m^3 of sediment are removed every year, which are deposited in the Río de la Plata some kilometers off the bay. There is no information about the behavior of these sediments after their deposition (Muniz et al., 2004a).

The knowledge about local pollution levels and its effects is limited, mainly in relation to the biota. The first studies concerning heavy metal and hydrocarbon pollution in the Montevideo coastal zone were published by Moyano et al. (1993) and Moresco & Dol (1996). These authors found near the mouth of the Pantanoso Stream sediments severely contaminated by chromium derived from leather fabrics. Altogether, these activities promote the input of different contaminant classes that can modified environmental conditions and have detrimental effects on the ecosystem biota. These effects should be identified and quantified. In view of this background information, the following section presents results generated during 1997 and 1999 under a series of research projects developed by our group. The projects were developed to evaluate the degree of contamination and its effects on the ecosystem health through the study of the benthic communities and its environment in a portion of the Montevideo coastal zone, specifically between Punta Carretas and Punta Yeguas, including Montevideo Bay and Montevideo Harbour (Figure 4). The main goal was to create scientific-technical basis for the correct instauration of an environmental monitoring plan for the Montevideo coastal zone. With this objective we collected field information in a series of sampling stations concerning environmental and biotic variables.

In the Montevideo Bay (MB) and the adjacent coastal zone (ACZ) samples were collected monthly, in 24 stations during one year. Stations A to E were localised in the most inner portion of the bay, near the harbour, the electric power plant (Central Batlle), the oil refinery (ANCAP) and the mouths of Miguelete and Patanoso Streams (Figure 4). The west portion of the bay included stations F, G and I and the most external one included stations H and J. Stations K to Q were placed toward the east of the bay and in Punta Carretas zone, where the main submarine outfall of Uruguay is located. Stations R to X were positioned in the western part of the bay, in Punta Yeguas zone. The rationale of this sampling design was to evaluate and compare the environmental characteristics of these three zones, related with the influence of the urban effluents discharged by the Punta Carretas submarine sewage pipe and the two streams that flow into the bay. In the harbour area, samples were collected in winter and summer in 8 stations distributed to cover different portions inside the harbour area (stations A* to H*; Figure 4). Surveys were carried out onboard vessels provided by the Uruguayan Army and the Government of Montevideo City (IMM).

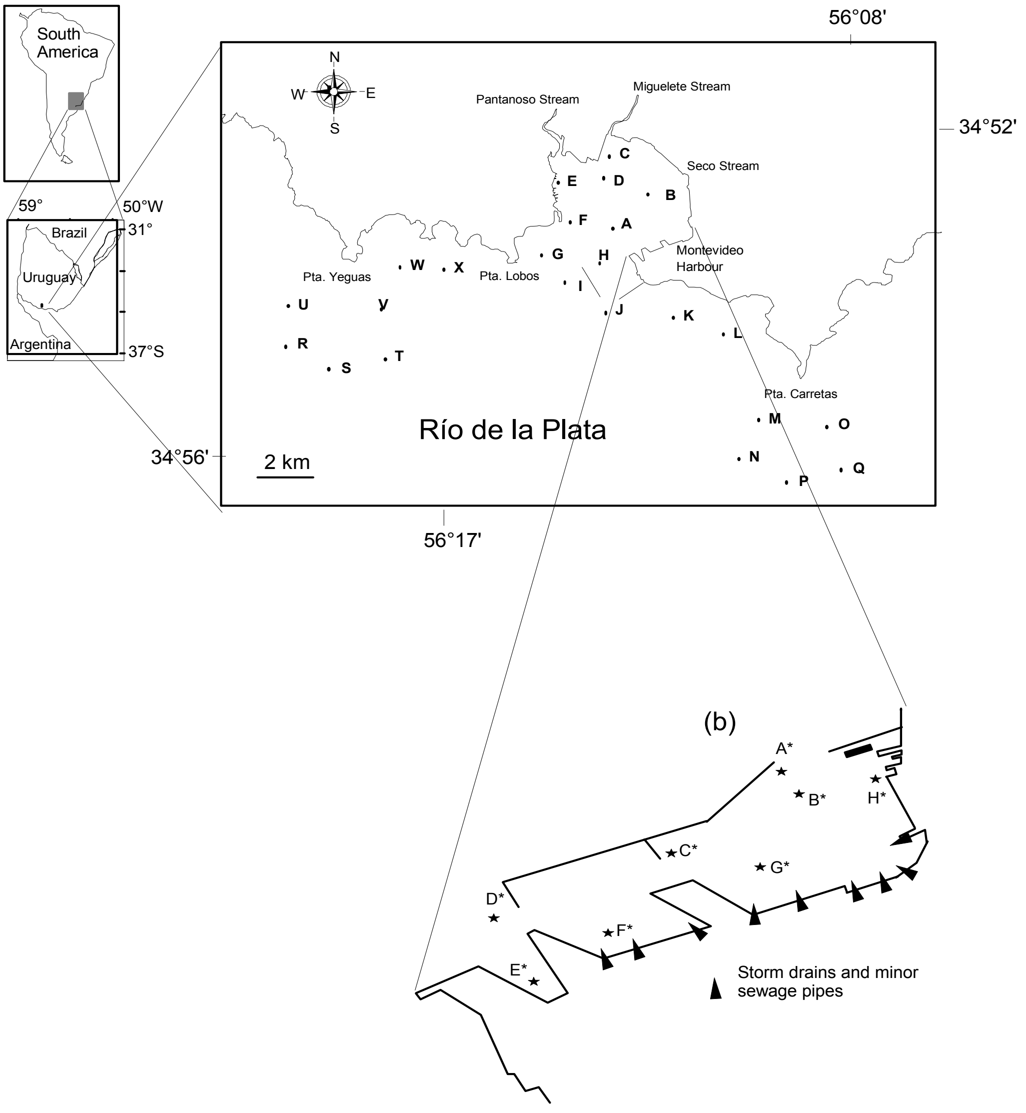


Figure 4. Location of (a) Montevideo Bay with the ten stations (A to J), and (b) Montevideo harbour with the eight stations (*) marked A to H. Arrows indicate the principal sewage pipes and storm drains that discharge into the harbour basins.

Salinity, temperature and pH of the water column were measured *in situ* using a YSI® multiparameter. Water was collected with HYDRO-BIOS water samplers to analyse oxygen dissolved concentration according to Winkler titration method (Grasshoff, 1983). At each station, 11 sediment samples were collected with manual Kajac corers of 4.5 cm internal diameter. Granulometric analysis was performed on three corers, the upper centimetre of three other corers, stored in a PE vial at 20°C, served for the quantification of organic matter and photosynthetic pigments. Sediments contained in three metallic corers were transferred to a tin foil and frozen in aluminium bottles until the analysis of hydrocarbons. Two acrylic corers were stored in vertical position in the cold; upon arrival at the laboratory, one of them was used for heavy metal analysis and the other one was used to measure redox potential. Danulat et al. (2002) and Muniz et al. (2002, 2004a) describe in detail the laboratory methods for granulometric, organic matter, chlorophyll *a*, heavy metals and hydrocarbons determinations, as well as redox measurements.

Biological samples for the study of macrobenthic communities' structure were collected in triplicate with an Ekman grab (0,053 m^2). They were sieved through a 0.4 mm mesh and preserved in 4% formol. After their separation from the sediment, macrobenthic organisms were preserved in 70% ethanol, counted and identified in most of the cases to the species level. To study the foraminiferal fauna, samples were collected with a Kajak corer (5 cm internal diameter) and the uppermost 3 cm layer of the sediment was taken at each station forming a volume of about 60 cm^3 per sample. Immediately after sampling, the material was stained with buffered rose Bengal dye (1 g of rose Bengal in 1000 ml of alcohol) for 48 h to differentiate between living and dead foraminifera (Walton, 1952). In the laboratory, the wet samples were carefully washed through 0.500 mm, 0.250 mm and 0.062 mm sieves to separate the size fractions. All living specimens in each sample were picked and identified following the generic classification of Loeblich & Tappan (1988).

RELATIONSHIPS BETWEEN MACROFAUNA AND MAIN STRESSORS IN THE MONTEVIDEO COASTAL ZONE

Montevideo Harbour

In the Montevideo Harbour area, in summer, only four benthic species were found, the gastropod *Heleobia australis*, the polychates *Alitta succinea* and *Nephtys fluviatilis* and the bivalve *Erodona mactroides*. Macrobenthic fauna was absent in the most inner part of the harbour (St. H*). *H. australis* was the only zoobenthic species found at the two sites in the inner harbour (Sts. B* and G*) and contributed to more than 94% of total abundance (Table 3). Very small organisms (< 2mm total length) made up a significant portion (ca. 21%) of the total number of specimens. In contrast, in winter the community was composed only by two species, the dominant *H. australis* and *N. fluviatilis*. The very small size of the *H. australis* individuals and the highly variable percentage of small specimens could be indicating that the recolonisation, settlement of juveniles, and their further survival are greatly altered, probably due to dredging and frequent sediment resuspension. Larvae of the polychaete *Heteromastus similis*, a small opportunistic species adapted to variable environmental conditions, poor oxygenation and high concentrations of organic matter of the sediment (Pearson &

Rosenberg, 1978), were recorded in plankton samples (Danulat et al., 2002). However, no adults were found colonising the sediments. These polychaete species that inhabit the harbour area are considered as tolerant to variable environmental conditions (Méndez et al., 1998; Muniz & Pires, 2000), and in Brazil, *Alitta succinea* is commonly found in eutrophic coastal areas (Amaral et al., 1998).

Table 3. Spatial distribution of four macrozoobenthos species (individuals m^2) in Montevideo Harbour, in March compared to July of 1998.

Location	< 2 mm	*Heleobia* 2-4 mm	6 mm	total	*Nephthys*	*Alitta*	*Erodona*
March 1998							
A*	14500	14187	438	29125	0	63	0
B*	3750	8000	1125	12875	0	0	0
C*	5063	13250	11000	29313	0	0	63
D*	4313	16375	15437	36125	63	125	13
E*	187	938	312	1437	63	63	0
F*	2063	11625	15375	29063	63	0	0
G*	1687	6313	5875	13875	0	0	0
H*	0	0	0	0	0	0	0
July 1998							
A*	125	1250	125	1500	0	0	0
B*	250	562	125	937	0	0	0
C*	187	17000	34750	51937	0	0	0
D*	6250	169937	90000	266187	125	0	0
E*	250	813	250	1313	0	0	0
F*	375	2250	2750	5375	63	0	0
G*	312	1500	438	2250	0	0	0
H*	0	0	0	0	0	0	0

For the gastropod *Heleobia cf. australis australis*, total number of specimens as well as abundance of three size classes is presented, while only the total number of individuals is indicated for the polychaetes *Nephthys fluviatilis* and *Alitta succinea*, and the pelecipod *Erodona mactroides*. Results represent mean values based on three replicate sediment samples retrieved by grab and five obtained by corer.

Related with the abiotic benthic environment, sediments showed little variation of the grain size. The predominant fraction was silt (up to 85%) and the only area where sand reached 15% was that near the fluvial dock (St. E*). Organic matter content of the sediments was high, with a clear spatial gradient from the inner area (St. H* = 16.5 %) to the outer one of the harbour (St. G = 9.6%). Accordingly, redox potential evidenced the lack of oxygen in surface sediments at most of the stations. Reduced conditions were detected at 1cm depth within the sediment column with values ranging between – 90 and + 100 mV, only at St. D* the top 1 cm of the sediment tend to be oxygenated (+210 mV) (Figure 5).

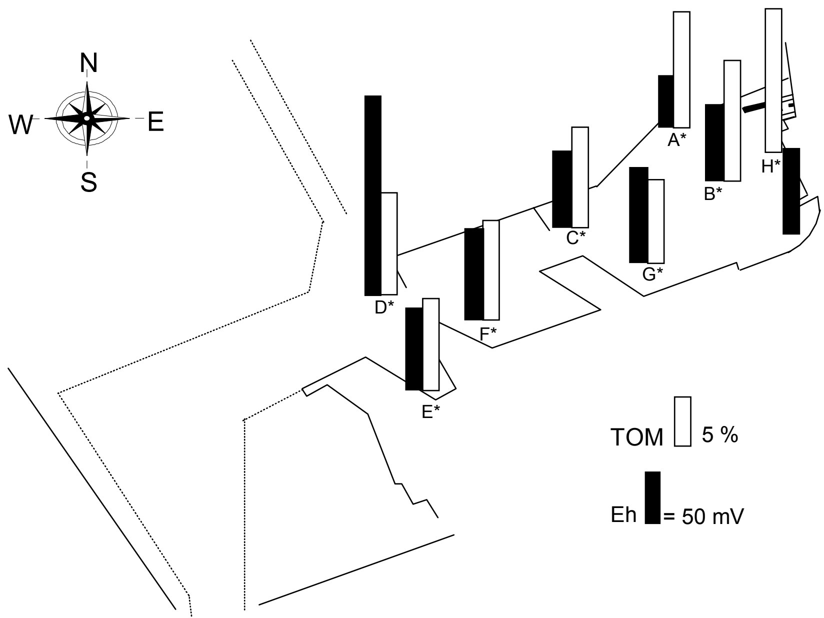


Figure 5. Total organic matter (TOM, in %) and redox potential (Eh, in mV) at the eight stations in Montevideo harbour area.

Table 4 present data of heavy metals in different marine and estuarine environments of Latin America, including the harbour area, MB and the adjacent coastal zone. The comparison of the results with those reported for other regions of Latin America indicates that levels and ranges of variation of our data are similar to those reported from sites with high anthropogenic disturbance. These data and those derived from the analysis of hydrocarbons were evaluated for potential adverse effects on biological organisms using available sediment quality values and sediment quality guidelines (Muniz et al., 2004a). Concerning heavy metals and considering that toxicity rarely occurs below the TEL and frequently above PEL (MacDonald et al., 1996), the authors concluded that: i) the inner region of the harbour showed heavy metal concentrations that may cause major adverse biological effects, except for Ni and Hg; and almost all the metals were above PEL level; the exceptions were Cd, Ni and Ag that were between TEL and PEL; ii) stations D*, E* and F* presented Pb and Cr levels between TEL and PEL; iii) only at St. H* Ag would cause major adverse biological effects on benthic fauna. Related to PAHs, at least one of the analysed contaminants was present in excess of the TEL level, and most of them (ca. 81%) had at least one compound in excess of PEL. This situation is more severe in summer, when all compounds listed by the U.S. Environment Protection Agency were above PEL (Muniz et al., 2004a). These results were coincident with those observed for the benthic fauna abundance patterns (Danulat et al., 2002).

Table 4. Range of variation of the heavy metals of sediments in different environments of Latin America n.a = not analyzed, concentrations are all in mg kg^{-1}, (bl) = background level

Location	Cd	Zn	Cu	Cr	Ni	Pb	Ag	Hg	Reference
Uruguay									
Montevideo Harbour	<1.0-1.6	183-491	58-135	79-253	26-34	44-128	<1.0-2.3	0.3-1.3	Muniz et al. (2004a)
Montev. Coastal Zone	41-231	2.4-105	1.3-4.0	n.a	n.a	40-148	n.a	n.a	Moyano et al. (1993)
Montev. Bay	0.1-0.2	n.a	10-112	n.a	n.a	40-148	n.a	n.a	Moresco & Dol (1996)
Montev. Bay	109	300	150	300	n.a	81	n.a	n.a	Cranston et al. (2002)
Montev. Bay	n.a	n.a	n.a	68-1062	n.a	99-365	n.a	n.a	Muniz et al. (2002)
Montev. Coastal Zone	n.a	n.a	n.a	37-50	n.a	38-56	n.a	n.a	Muniz et al. (2002)
Carrasco Creek				10-807		17-73			Lacerda et al. (1998)
Brazil									
Patos Lagoon	0.1-20	20-214	0.8-20	8-337	n.a	8-267	n.a	n.a	Baisch et al.(1988)
Patos lagoon	n.a	n.a	n.a	n.a	n.a	n.a	n.a	0.06 (bl)	Mirlaen et al. (2001)
Coastline of RJ State	n.a	14-795	11-166	18-121	11.2	10-83	n.a	n.a	Lacerda et al. (1982)
Jurujuba Sound	n.a	15-337	5-213	10-223	15-79	5-123	n.a	n.a	Baptista Neto et al. (2000)
Mangroves in RJ	n.a	26-610	18-80	n.a	6-12	20-130	n.a	n.a	Machado et al. (2002)
Argentine									
Bahia Blanca	0.44	35.5	7.3	n.a	n.a	17	n.a	n.a	Villa et al. (1988)
Venezuela									
Coral reef sediments	n.a	36-77	6-40	18-32	n.a	17-36	n.a	0.2-0.33	Bastidas et al. (1999)
Chile									
South Fjords	0.1-0.5	91-122	16-22	49-82	24-31	26-29	n.a	n.a	Ahumada &Contreras (1999)
Mexico									
Baja California	n.a	n.a	n.a	n.a	n.a	n.a			Carreón-Martínez et al. (2001)

The biological benthic diversity of the Montevideo Harbour is extremely poor. Macrobenthic fauna consists of only four species of small size and is dominated strongly by the gastropod *H. australis*, an opportunistic species that feed on the surface sediment. This general pattern of low diversity and high abundance of a single species is common to estuaries world-wide, including those in the same geographic region (Benvenuti, 1997; Ieno & Bastida, 1998; Muniz & Venturini, 2001; Giménez et al., 2005, 2006). Diversity and species richness, however, are substantially lower in Montevideo Harbour than at the locations in the immediate vicinity. Scarabino et al. (1975) found seven macrobenthic species just outside the harbour area that were considered typical for the entire Montevideo coastal zone. In addition to the dominant *H. australis* the authors found three bivalves (*E. mactroides*, *Tageleus plebius* and *Mactra isabelleana*) and one species of Tubificidae (oligochaete). In the contiguous Montevideo Bay Muniz et al. (2000, 2005a, b) and Venturini et al. (2004) recorded ten species, nevertheless, the dominant was *H. australis*. Further than the fact that low biodiversity found in estuaries can be merely the result of highly variable salinity and unstable sediments (Tenore, 1972; Wilson, 1994), it has also been established that chronic contamination produces and additional reduction of macrofaunal diversity (Heip, 1995). Data analysed for the Montevideo Harbour clearly suggest the latter case.

Montevideo Bay

In Montevideo Bay, considering the ten sampling stations (Figure 4) in an annual cycle, the total mean abundance determined for macrobenthic species was of 30,118 individuals belonging to the Phyla Arthropoda, Nematoda, Mollusca and Annelida. As in the neighbouring area of Montevideo Harbour, the most frequent and dominant species was *Heleobia australis*, following by *Nephtys fluviatilis*, *Erodona mactroides*, *Heteromastus similis*, *Alitta succinea, Goniadides sp., Glycera sp., Sigambra grubii,* an unidentified isopod as well as unidentified Nematoda and Ostracoda.

Density was very variable at each station on a month scale. Over all stations, the highest values occurred in May, June and July 1997 and the lowest in March 1998 (Figure 6a). In general, stations B, C and D presented a lower number of individuals per unit area than the remaining stations. Ignoring unidentified Nematodes, Ostracods and Barnacles a total of 10 species were recorded. Maximum species richness occurred between April and August 1997 (Figure 6b). Stations F and G showed the highest species richness, and the lowest values were found at stations B, C and D. Species richness was low throughout the study. A higher number of individuals did not always correspond to a higher number of species. Shannon diversity was also very low, reflecting the high dominance of *Heleobia australis* that occurs frequently in very high abundance in several regions of the bay. The maximum diversity was registered in September 1997 at station G (1.63) and the minimum at station B (0) where only Nematodes were present. Between April and August 1997 the highest diversity values were registered in the majority of the sampling stations. In November of the same year diversity decreased notably and in February 1998 it increased again (Figure 6b). As for abundance, biomass was also variable in all stations during the period of study. The highest biomass values of the dominant species *Heleobia australis* were occasionally exceeded by those corresponding to the second most abundant species, the bivalve *Erodona mactroides*. The highest biomass was recorded in July 1997 and the lowest in March 1998 (Figure 6a).

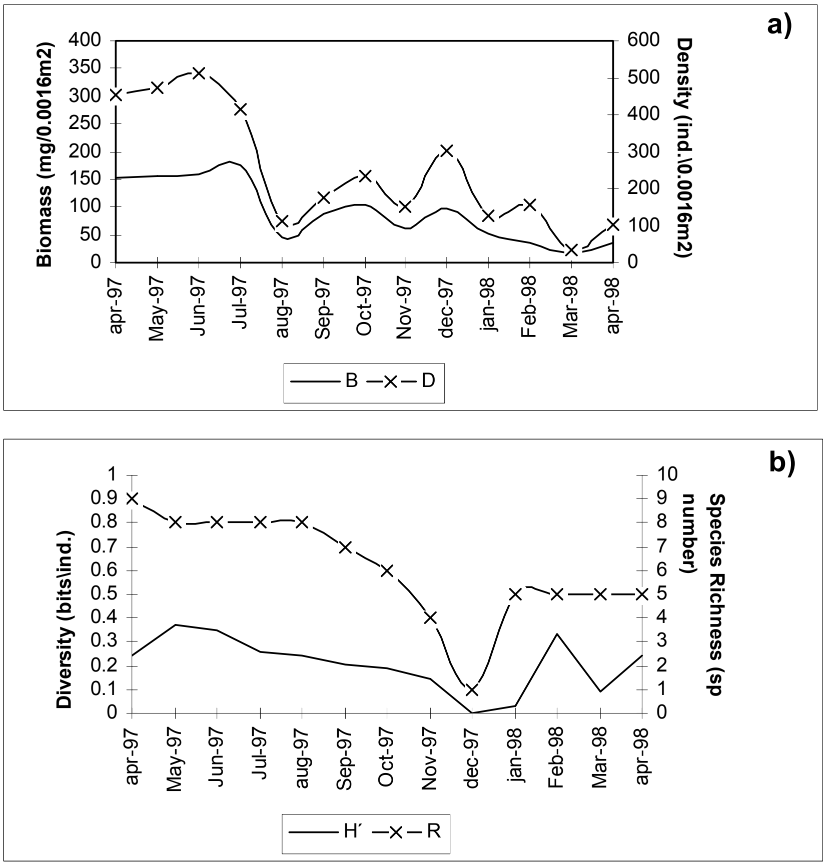


Figure 6. Mean monthly values of (a) density and biomass; and (b) diversity and species richness of macrobenthic fauna for Montevideo Bay.

The decline in the number of *H. australis* during some months of the studied period is reflecting that it is a short-lived species. The relation among the rise of organic matter concentration, the reduction in species number, diversity and the increment of the abundance of one or two species of small size have been well reported in previous studies (see for example Pearson & Rosenberg, 1978; Méndez et al., 1998; Oug et al., 1998; Sánchez-Mata et al., 1999). These species are generally considered as indicators of organically enriched sediments. In such communities, perturbed by organic contamination, the frequency of disturbance is higher than the recovery rate. Thus, opportunistic species of small size and short lifetime will be favoured and could colonise such habitats with any type of biological competition. Then, such species can be adapted to a high frequency of continuous disturbance. Even though, *Heleobia australis* was the most abundant (80% of the total abundance) and the dominant macrobenthic species, many of the other species, especially the polychaetes *Nephtys fluviatilis, Allita succinea, Heteromastus similis* and *Goniadides sp.*, have been also reported in environments with a high organic load elsewhere (e.g. Dauer & Conner, 1980; Amaral et al., 1998). The high frequency of occurrence of these species, in addition to the presence of large-bodied nematodes, which were retained in the 0.4 mm sieve, would be related to the high organic content of the sediments. Cluster analysis of abundance and biomass data (annual arithmetic mean of pooled data) showed two groups of stations at near 60% of similarity (Figure 7). One group was composed by the most-inner stations B, C and D, whereas the remaining stations constituted the other. The same two groups appeared in the MDS ordinations (Figure 7).

Although the bay was a very variable system, it was possible to differentiate, by means of the cluster analyses and the MDS ordination, discrete faunal associations, in regions with particular environmental characteristics. The cluster formed by Sts B, C and D that showed less abundance and biomass of benthic organisms, corresponds to the inner part of the bay where environmental conditions are very unfavourable. Water circulation is limited; there is a high percentage of organic matter, chromium, lead and PHAs in the sediments and a tendency to the presence of reduced sediments (Muniz et al., 2002). The other cluster formed by the remaining stations A, E, F, G, H, I and J corresponds to regions of the bay, which are heterogeneous, but in general have more favourable environmental conditions than the inner region. At Sts F and G particularly, the high water circulation and oxygenation of the sediment column, together with the small percentage of organic matter, may be responsible for the great abundance and biomass of benthic organisms recorded.

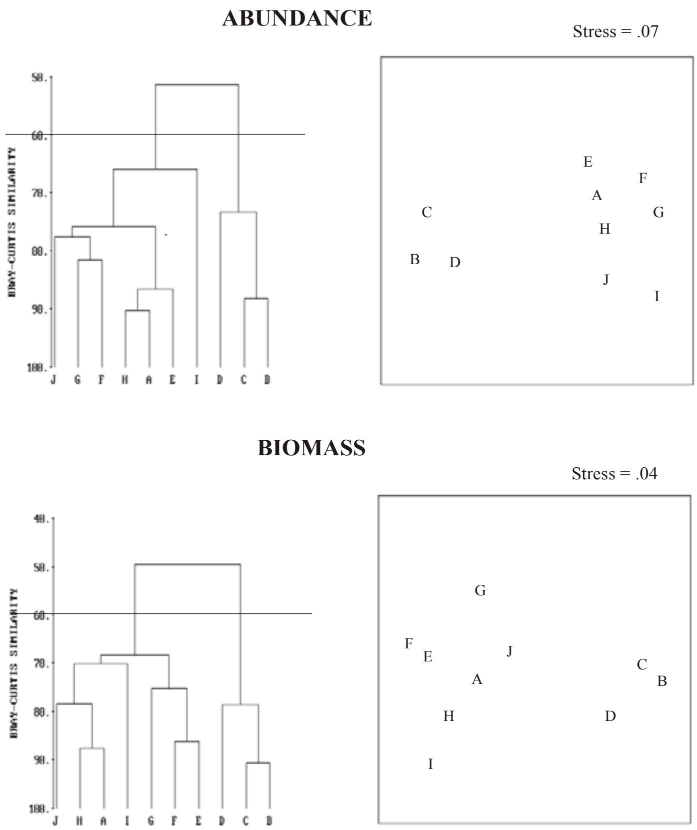


Figure 7. Dendrograms and MDS ordinations diagrams showing the results of grouping the sample stations (Q-mode). (a) by abundance. (b) by biomass. Sampling stations of Montevideo Bay (A to J).

Montevideo bay and the Adjacent Coastal Zone: Their Degree of Perturbation and Ecological Status

When considered Montevideo Bay and the adjacent coastal zone (Stations A to X, Figure 4) the main results using multivariate statistical techniques showed that the region can be divided in three zones with different abiotic (Figure 8) and macrofaunal patterns (Figure 9). The inner portion is the most heterogeneous according to sedimentological composition. It has a high organic load and is highly contaminated by chromium, lead and oil derived hydrocarbons. The outer portion of the bay and the ACZ showed a moderate contamination level. Despite the dominance of *Heleobia australis* in the whole area, the difference in environmental quality among the three regions was reflected in their macrobenthic community structure. In the inner portion of MB, benthic communities showed a very simple structure, being dominated by nematodes, organisms that belong to the meiofauna and generally are associated to organically enriched environments (Warwick, 1986) and also some individuals of *Heleobia* (Muniz et al., 2000; Venturini & Muniz, 2001; Venturini et al., 2004). In the other two zones, benthic communities showed a more complex structure with a higher number of species and diversity. Moreover, in the inner portion of the bay the individuals of the dominant species presented epibiontic parasites (Ciliophora of the family Vortecellidae: *Zoothanium elegans*), a smaller size and thinner shells than those individuals of the outer portion and the ACZ. Mean biomass values for Gastropoda in the group 1 of stations, were 20 times lower than those in group 3, and 6 times lower than in group 2.

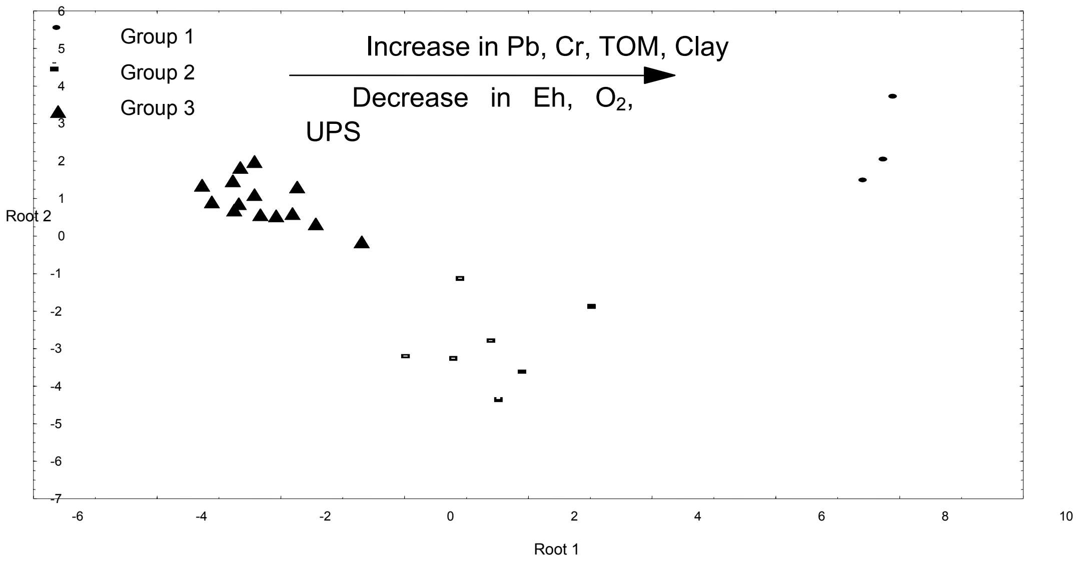
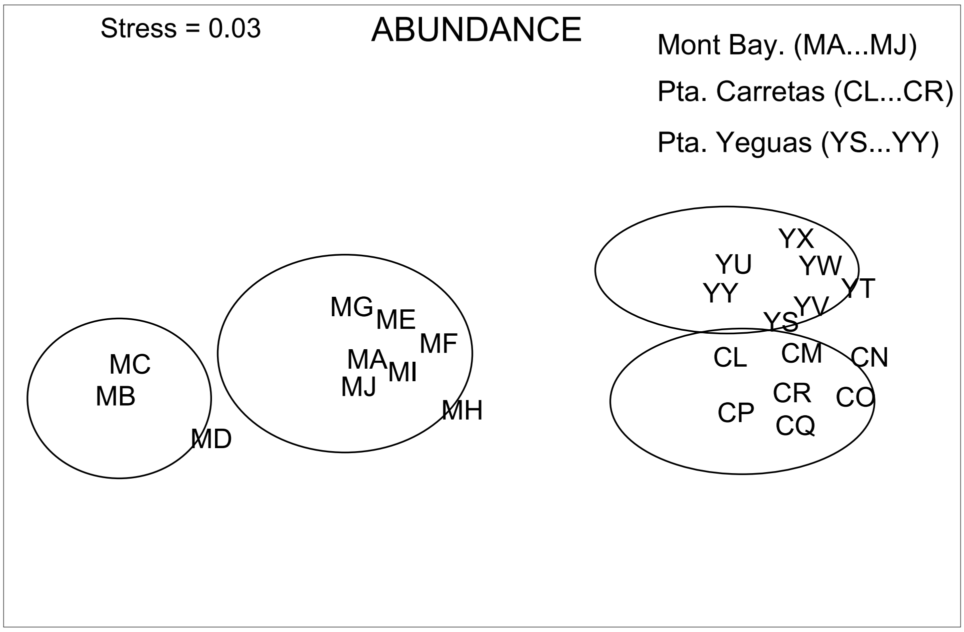


Figure 8. Plot of location groups produced by the multiple discriminant analysis. Pb = lead content; Cr = chromium content; TOM = total organic matter content; Eh = redox potential; O^2 = bottom water dissolved oxygen content; UPS = bottom water salinity.



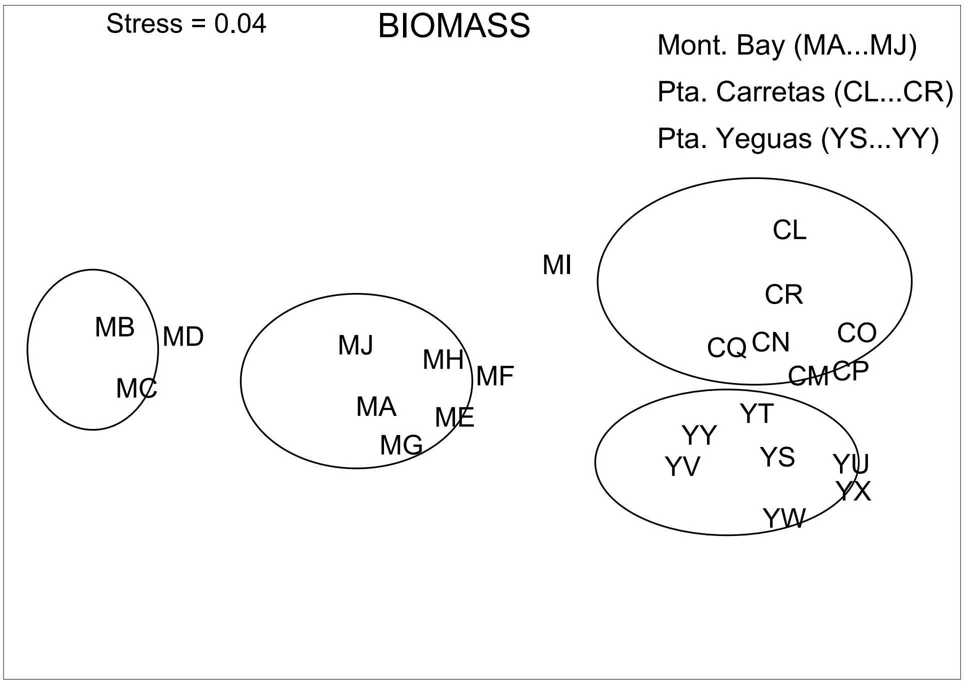


Figure 9. Multi-dimensional scaling ordination diagram of the abundance and biomass (bottom) data from Montevideo Bay (M stations), Punta Carretas (C stations) and Punta Yeguas (Y stations).

The environmental variables that best explained the macrofauna distributional pattern were lead and polycyclic aromatic hydrocarbon concentrations and salinity (Venturini et al., 2004). It is known that coastal regions and estuaries are very dynamic environments characterised by great variations in their abiotic parameters and subjected to continuous natural disturbances. This natural variability can represent the main cause of stress for organisms (Turner et al., 1995), however, the input of nutrients, organic matter and human-derived contaminants can alter environmental conditions in a different manner from that expected by natural causes alone (Pearson & Rosenberg, 1978; Mucha et al., 2002).

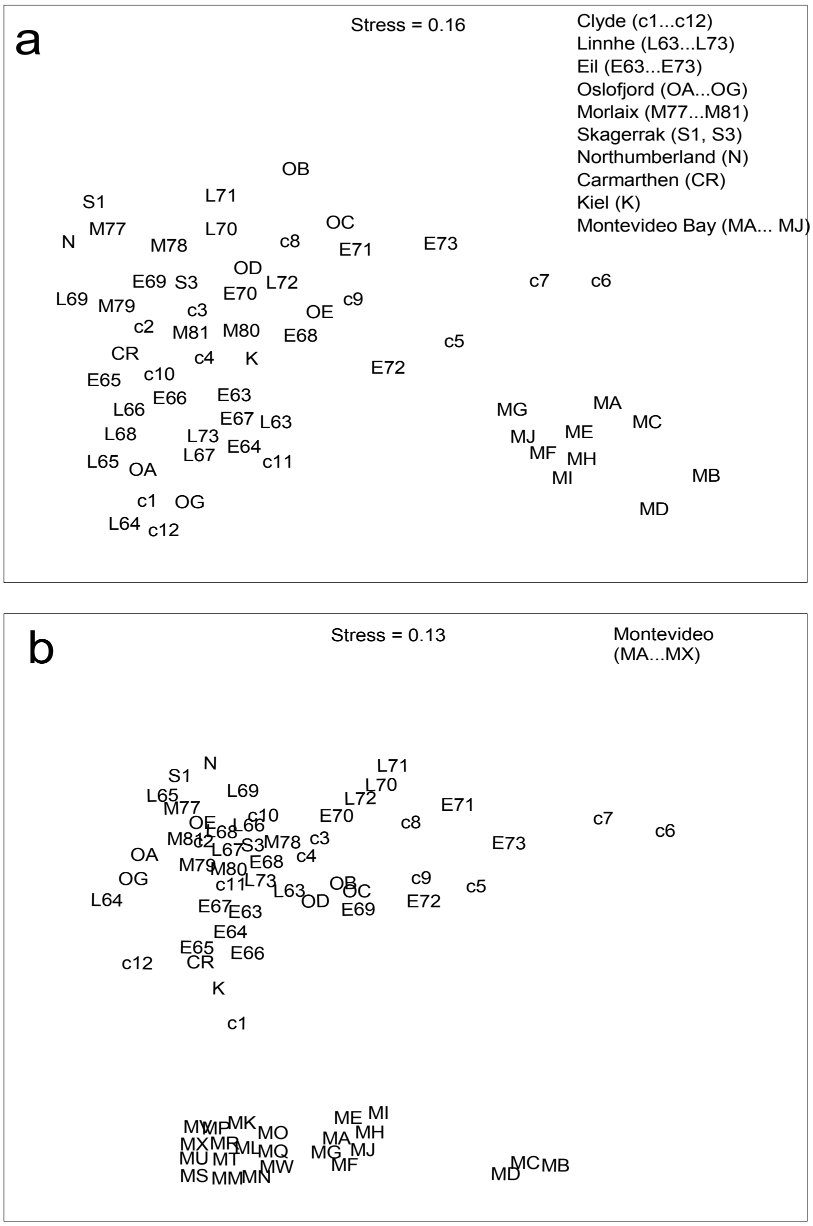


Figure 10. Multi-dimensional scaling ordination diagram of the phylum-level production data from: a the ten Montevideo Bay samples combined with the 50 original NE European shelf samples (Warwick and Clarke 1993) and b the 24 Montevideo coastal zone samples combined with the 50 original NE European shelf samples. Nomenclature of locations in panel b is the same as in panel a, plus Montevideo coastal stations (MK to MX).

Warwick & Clarke (1993) created a method denominated "Phylum-level Meta-analysis", which using abundance and biomass data at the phylum level, allows the evaluation of the degree of perturbation of a particular benthic community in a global scale of anthropogenic impact. Applying this method to data obtained in the Montevideo coastal zone, it was observed that the inner stations of MB were located in the right side of the diagram (Figure 10), indicating a high degree of perturbation of these communities (Warwick & Clarke, 1993). The other two groups formed by the stations of the outer portion of MB and the ACZ, were located in a gradient from lower to higher impact, respectively. The position of stations B, C and D in the right side of the ordination diagram is related to presence of large size nematodes, coupled to the great organic load of the sediments. The vertical separation of Montevideo stations from the others is the result of the high dominance of the gastropoda *H. australis*, which would tolerate the instability of environmental conditions, as well as, the elevated organic and heavy metal loads. Within this framework, stations B, C and D can be classified as highly contaminated, station A and stations E to J as contaminated and the others (K to X) as moderately contaminated (Venturini et al., 2004).

All previous studies indicated the occurrence of a vertical separation between the samples studied and the NE Atlantic samples. This separation was attributed to the alteration of the balance between echinoderms and crustaceans due to estuarine characteristics of the coast of Trinidad (Agard et al., 1993), to the higher proportion of annelids due to oxygen-deficient conditions in central Chile (Tam & Carrasco, 1997) and to the higher proportion of crustaceans and annelids and the lower proportion of molluscs and echinoderms in the Gulf of Cádiz (Drake et al., 1999). Non-mined samples from southern Africa exhibited a lower proportion of molluscs and echinoderms but a larger proportion of crustaceans than the NE Atlantic samples (Savage et al., 2001). In the Montevideo coastal zone data, part of the separation could be explained by the absence of echinoderms due to the salinity gradient that would prevent the presence of these osmoconformer organisms (Agard et al., 1993). It could also be attributed to the lower proportion of annelids and the higher proportion of crustaceans in the Montevideo coastal zone samples with regard to those from the NE Atlantic. However, the prominent separation of the Montevideo coastal zone samples seems to be mainly due to the high proportion of molluscs. Similar results were obtained in mined samples from the southern African coast, which were ascribed to the capacity of some gastropods to withstand the physical disturbance caused by the mining process (Savage et al., 2001). In this case the high proportion of molluscs is the consequence of the high dominance of the gastropod *Heleobia australis* that seems to tolerate the environmental instability typical of estuarine areas and the high organic and inorganic loads existing in the Montevideo coastal zone. According to Rakocinski et al. (2000), hydrobiid gastropods are among many opportunistic and/or tolerant estuarine taxa associated with sites moderate or high contaminated with both metals and organic chemicals.

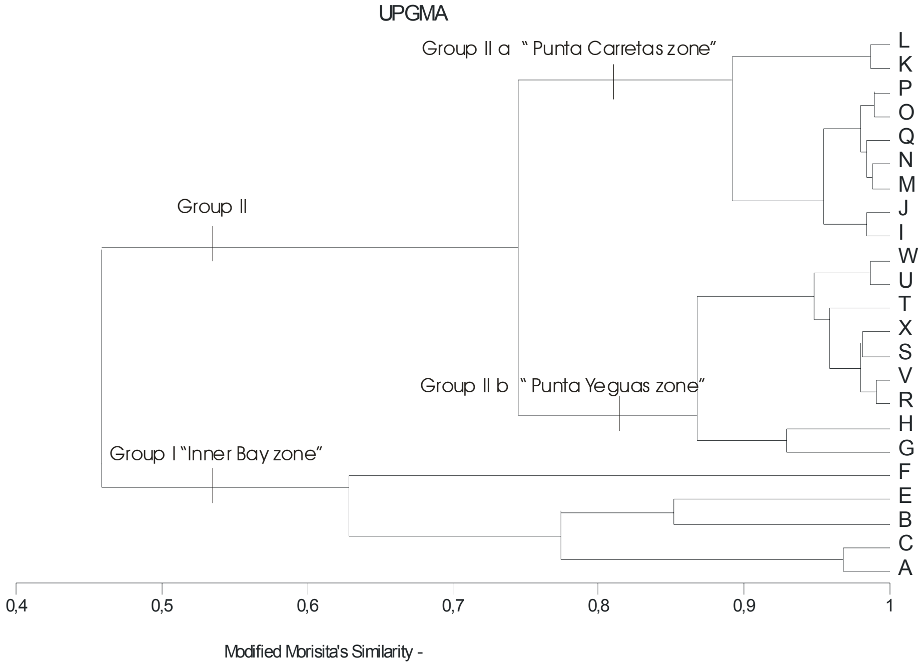


Figure 11. Dendrogram classifications showing the station groups with abundance data of foraminifera species.

A conceptual framework, which has been the basic reference in the literature concerning the effects of organic enrichment on benthic communities, has been established by Pearson and Rosenberg (1978). Within this context, several investigators have developed biotic indices to estimate macrobenthic community disturbance level and to establish the ecological status of soft-bottom benthos (Hilly et al., 1986; Grall & Glémarec, 1997). All such studies have emphasised the importance of biological indicators, to measure the ecological quality of a marine environment (Engle et al., 1994; Grall & Glémarec, 1997; Weisberg et al., 1997). Recent approaches have developed a biocriteria-based predefined reference condition and, upon this, several deviations (disturbance classes) were established (Dauer & Alden, 1995; Weisberg et al., 1997; Van Dolah et al., 1999). Borja et al. (2000) have proposed the adoption of AZTI´s Marine Biotic Index (AMBI), using macrobenthic organisms as bio-indicators. These authors have explored the response of soft-bottom communities, to natural and man-induced changes in water quality. Such approach has integrated the long-term environmental conditions in several European estuarine and coastal environments. This index is based essentially upon the distribution of 5 ecological groups, of soft-bottom macrofauna (Grall & Glémarec, 1997); these are in relation to their sensitivity to an increasing stress gradient. Such an approach has the advantage of being simple, in terms of calculation, compared to those adopted previously; it is based upon a formula which permits the derivation of a coefficient (AMBI, Borja et al., 2003, 2004), allowing statistical analysis of the results. Further, Borja et al. (2003) have established that benthic samples subjected to different impact sources e.g. organic enrichment, physical alterations of the habitat, heavy metal inputs, etc., along the European coast, were classified correctly according to the Marine Biotic Index (see Muxika et al., 2005 and Borja et al., 2006, for a review of the increasing use of this index, within Europe). In this context, Muniz et al. (2005a) applied AMBI to the data set of the Montevideo coastal zone, including Montevideo Bay area. For Montevideo Bay and the adjacent coastal zone, Muniz et al. (2002) and Venturini et al. (2004) had classified the innermost stations of the bay (B, C and D) as grossly polluted, Stations A and E to J as polluted and the remainder as moderately polluted. The inner part of Montevideo Bay: (a) was associated with high concentrations of chromium, lead and polycyclic aromatic hydrocarbons in the sediments; (b) presented anoxic conditions, with negative values of redox potential; and (c) the benthic communities were dominated by large-sized nematodes, which were not considered in the AMBI calculations (see discussion below).

Although, the dominance of the second-order opportunistic species *Heleobia australis* was apparent over the whole area, in most of the outer stations of Montevideo Bay and in the adjacent coastal zone, the benthic communities were richer, more diverse and the bottom conditions were less severe. This general trend was observed clearly with high AMBI values in the innermost stations of Montevideo Bay; these decreased throughout the outermost part of the bay and the adjacent coastal zone (Table 5). Based upon the AMBI (without considering the nematodes), the innermost stations were classified as moderately disturbed (Table 5). These stations were dominated, both in terms of abundance and biomass, by large nematodes (retained in 0.4 mm sieve mesh). For this reason, in a previous study when data were analysed with the phylum-level meta-analysis approach, they presented the status of grossly polluted (Venturini et al., 2004). According to the ecological group assignment followed to the calculation of AMBI and BI values, nematodes belong to EG III (Borja et al., 2000); this will result in the final classification of these inner stations as being slightly disturbed, instead of grossly polluted, sites. Perhaps, these large nematodes, probably oncholaimid, should be assigned to ecological group V, since their dominance could be a symptom of polluted conditions (Warwick, 1986). Conversely, Stations K and L, which were classified previously as moderately polluted (based upon chemical data, Muniz et al., 2002, 2004b), revealed AMBI values which are similar to those presented by the innermost stations of Montevideo Bay. Along this sector of the coastline, there is an important sewage outfall that carries also the city runoff and other untreated industrial effluents (Moyano et al., 1993); this promotes, probably, adverse effects on the benthic fauna, reflected in AMBI values. Venturini et al. (2004) did not identify any difference in the macrobenthic communities between the two coastal zones adjacent to Montevideo Bay. However, these investigators argued that Punta Carretas should be more perturbed than Punta Yeguas, due to the effluents released by the largest and most important sewage outfall of Uruguay, situated in this area. With the use of the AMBI and BI, it was possible to detect differences between these two zones (Table 5). Punta Yeguas was classified as an area with an unbalanced benthic community health (slightly disturbed) and Punta Carretas as moderately disturbed, with a transitional to pollution benthic community health. Although the dominance of *Heleobia australis* was evident in the two zones, the crustacean species in Punta Yeguas (mainly EG I) had an important contribution, in terms of abundance (between 25 and 49 % of the total abundance); this suggests that the AMBI appeared to be more sensitive in this respect, than the meta-analysis approach previously applied to this data set.

Table 5. Values of AMBI, BI, total abundante (ind/0.053 m^2), total organic matter content (%TOM), chlorophyll *a* content (ug/g), chromium (mg/kg) and lead (mg/kg) content of the sediments, benthic community health (BCH) and site disturbance classification (SDC) in the 24 stations of the Montevideo Coastal Zone

St	AMBI	BI	Abund.	TOM	Chl *a*	Cr	Pb	BCH	SDC
A	4.4	4	396	6.6	6.2	131.5	215.1	Polluted	Moderately disturbed
B	4.5	4	3	11.3	11.6	81.1	246.7	Polluted	Moderately disturbed
C	4.5	4	1	8.3	3.5	91.5	369.6	Polluted	Moderately disturbed
D	4.5	4	1	12.0	8.8	657.1	352.2	Polluted	Moderately disturbed
E	4.3	3	134	12.8	0.8	368.1	64.9	Polluted	Moderately disturbed
F	4.2	3	144	3.5	0.5	43.7	44.7	Transitional to pollution	Moderately disturbed
G	3.9	3	32	7.5	0.3	30.9	39.1	Transitional to pollution	Moderately disturbed
H	4.2	3	65	9.4	0.5	83.7	65.4	Transitional to pollution	Moderately disturbed
I	3.6	3	17	6.2	0.2	42.1	38.5	Transitional to pollution	Moderately disturbed
J	4.1	3	17	9.5	0.3	56.2	41.7	Transitional to pollution	Moderately disturbed
K	4.4	4	3091	6.8	0.1	38.9	56.4	Polluted	Moderately disturbed
L	4.3	3	3375	6.9	0.5	42.5	57.9	Polluted	Moderately disturbed
M	3.2	2	388	4.8	0.5	40.1	58.5	Unbalanced	Slightly disturbed
N	3.8	3	153	4.5	0.5	40.3	58.9	Transitional to pollution	Moderately disturbed
O	3.9	4	867	6.1	0.8	39.3	57.9	Transitional to pollution	Moderately disturbed
P	3.7	3	338	4.6	0.4	36.7	55.1	Transitional to pollution	Moderately disturbed
Q	3.8	4	305	5.9	0.2	39.3	55.2	Transitional to pollution	Moderately disturbed
R	2.7	2	2412	6.1	0.3	38.8	54.5	Unbalanced	Slightly disturbed
S	2.3	2	1804	5.6	0.3	36.9	55.4	Unbalanced	Slightly disturbed
T	2.6	2	1679	5.2	0.2	37.4	55.1	Unbalanced	Slightly disturbed
U	3.1	2	537	5.5	0.4	38.3	56.2	Unbalanced	Slightly disturbed
V	2.9	2	2267	6.6	0.6	38.9	56.7	Unbalanced	Slightly disturbed
W	3.2	2	49	6.5	0.1	42.1	54.8	Unbalanced	Slightly disturbed
X	3.2	2	322	5.4	0.3	38.1	54.9	Unbalanced	Slightly disturbed

Table 6. Density of foraminifera species present in Montevideo Coastal Zone at the 24 stations

Species/Stations	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T	U	V	W	X
Ammobaculites exiguus							25			2			1		21			6	17	9	4	2	1	
Ammonia parkinsoniana							152	51			66	42	11	15	48	2	1	11	42	65		31		56
Ammonia rolshauseni																								5
Ammonia tepida	3	8	12		12	2	241	159	112	117	308	137	911	1120	6390	1230	1000	157	665	600	379	242	329	530
Ammotium salsum							38	12			2		3	1	3		1				1		8	
Bolivina pulchella															6									
Brizalina striatula													17	8	1	4	16		12		9			5
Bulimina marginata															3		1							
Buliminella elegantissima	7	1	5		8															5				
Cibicides variabilis												1	1	1	6	1	5							
Discorbis williamnsoni														2	4	3								
Elphidium excavatum					5	2	16	7	8	5	10	3	9	25	12	1	3	1	11	30	1		3	2
Milammina fusca						8	287	116			2		2	6	34	5	24	93	600	222	191	174	202	415

Pararotalia cananeiaensis								7	1	3	5	1	11		21	1	1							
Pseudononion atlanticum								8					4	2	15	8	12		5					
Psammosphaera sp.	3	2	5		7																			
Rosalina sp.														1	1									
Trochammina sp.						1	5													1				
Total	13	11	22		32	13	764	360	121	127	393	184	970	1181	6565	1255	1064	268	1352	939	585	449	543	1026

RELATIONSHIPS BETWEEN FORAMINIFERAL FAUNA AND MAIN STRESSORS IN THE MONTEVIDEO COASTAL ZONE

Studies carried out by Burone et al. (2004) and Burone et al. (2006) in the Montevideo coastal zone to study the foraminifera responses to polluted sediments allowed the identification of a total of 18 species and 18,341 individuals of benthic foraminifera at the 24 stations sampled (see Figure 4 and Table 6) belonging to the suborders Rotaliina (13 species) and Textulariina (five species). According to the results of these works, three different sub-environments based on foraminiferal assemblages and population parameters can be distinguished (Figures 11, 12 and 13).

In one hand, Montevideo Bay, the inner portion in particular, showed an extremely poor foraminiferal fauna, including a totally azoic station, evidencing the high degree of local contamination. *Psammosphaera* sp. and *Buliminella elegantissima* were the representative spices recorded in this region, however, other species like *Ammonia tepida* and *Elphidium excavatum* were also observed but in a very low density. In addition, high percentages of abnormal test were observed, being that most of them were classified as complex (Figure 14). Intermediate values of diversity (between 0.760 and 1.337) and high evenness, with values approaching 1 were registered. This sub-environment is under stressed environmental conditions due to the different pollutants that come from different sources and strongly affect this zone, such as the high concentration of organic matter and heavy metals (Muniz et al., 2002, 2004a, b).

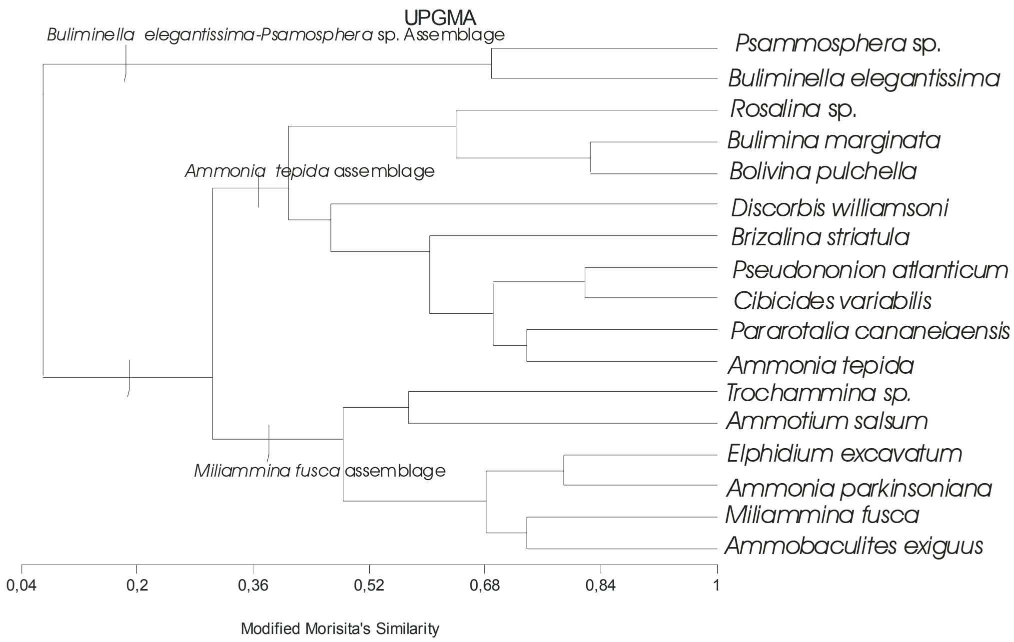


Figure 12. Dendrogram classifications showing the species assemblages with abundance data of foraminifera species.

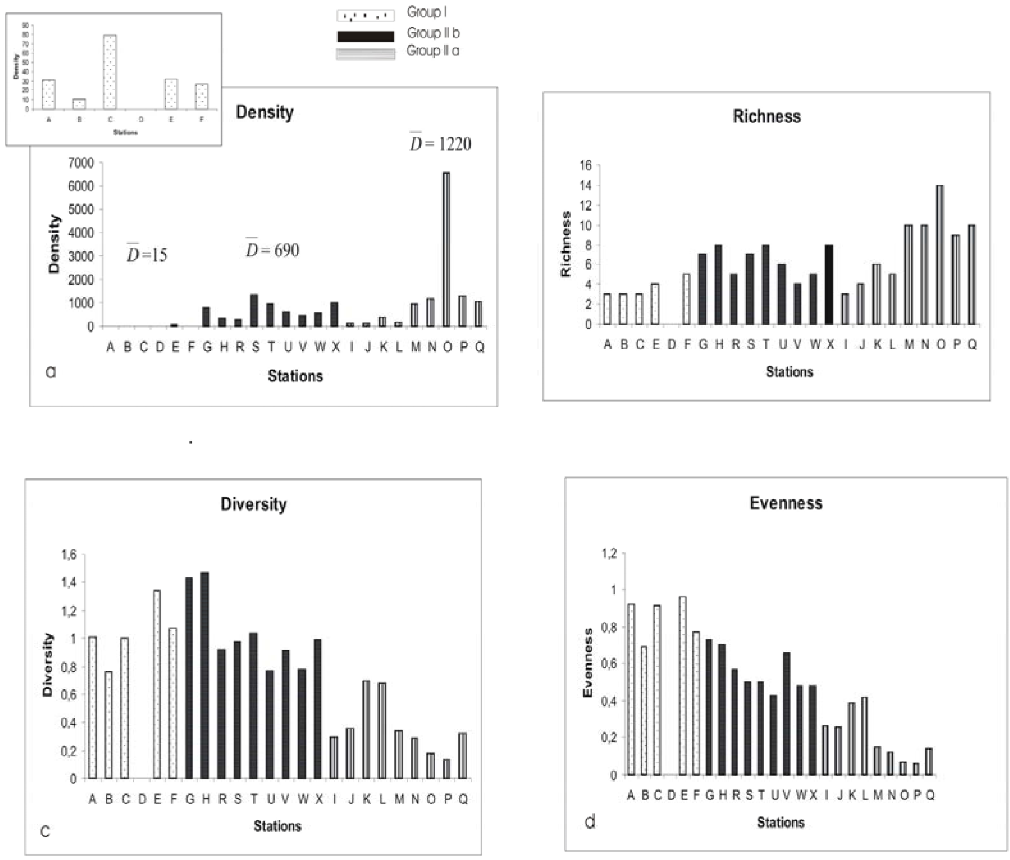


Figure 13. Populational parameters used to relate the foraminiferal assemblages to the environment conditions. $\overline{D} = \text{mean density}$.

On the other hand, Punta Carretas region is characterised by *Ammonia tepida* assemblage that consisted of nine hyaline species (Figure 12) with *A. tepida* represented 99.1 % of the total assemblage. This sub-environment showed the highest mean density (D = 1220) and the diversity was low ranged between 0.134 and 0.918. The species richness showed the highest values of the studied area with a maximum of 14 species. A positive effect on density of foraminifera, especially on *A. tepida* in this local was noted, which seems to be related to the discharges of the major sewage pipe of Uruguay that lies in this region, therefore, with a more pure organic contamination.

The third sub environment identified corresponding to Punta Yeguas zone was represented by *Miliammina fusca* assemblage (73.3 %) that was constituted by six species (two hyaline and four agglutinated). This zone showed values of pH and Cr slightly lower than those recorded in Punta Carretas zone (Burone et al., 2006), presented intermediate values of mean density (D = 600), as well as intermediate species richness and evenness, which ranged between 4 and 8, and from 0.428 to 0.734, respectively.

A high number of agglutinates species was recorded in the stations of Punta Yeguas, while calcareous species dominated in Punta Carretas zone. This pattern can be related to the great marine influence on the latter region as indicated by the presence of *Pararotalia* *cananeiaensis.* This is a small marine species easy to be transported by currents and used as an indicator of marine influence (Debenay et al., 2001). Moreover, the presence of *P. cananeiaensis* in this area can be the response to the entrance of more saline waters, considering that the mean upstream limit of the saline intrusion is located just at the transverse section of Punta Yeguas (Nagy et al., 2002).

Out of the 18 species found in the study area, three of them exhibited morphological abnormalities (*A. tepida*, *B. elegantissima* and *Elphidium excavatum*). Basically, all the hyaline specimens observed in the inner portion of MB showed at least one type of abnormality. Besides, more than 58% of the hyaline specimens showed morphological deformities in this region. When it is consider the total population at each sampling station, this number reaches 72.7 %. Morphological abnormalities in this zone were manifested as protuberances of one chamber (Figure 14 F), aberrant chamber shape and size (Figure 14 B and I), additional chambers (Figures 14 J and K), overdeveloped chambers (Figure 14 C) double apertures (Figure 14 M) and Siamese twins (Figure 14 E). Some specimens accumulate more than one type of deformation and were named as complex deformities (Figure 14 D). In the outermost regions (Punta Carreta and Punta Yeguas zones), the percentages of deformed specimens were much lower (between 0.03 % and 0.08%). In these zones, the morphological deformities were manifested basically by aberrant chamber size (Figure 14 B).

A strong relationship between organic matter, oxygen and heavy metal concentrations, as well as redox potencial and pH values with the mean density of each sub-environment was detected. The diverse pollution sources and the complex mixture of different contaminants in the sediments make difficult to identify the effect of a single stressor on benthic foraminifera, even more, in high variable environments such as estuaries. Based in the results obtained it was concluded that such an extreme harmful condition in the Montevideo Bay is a consequence of the combined action of all polluted factors presented in the bay. Moreover, the high percentages of abnormal tests in Montevideo Bay seem to be related with the high contamination level.

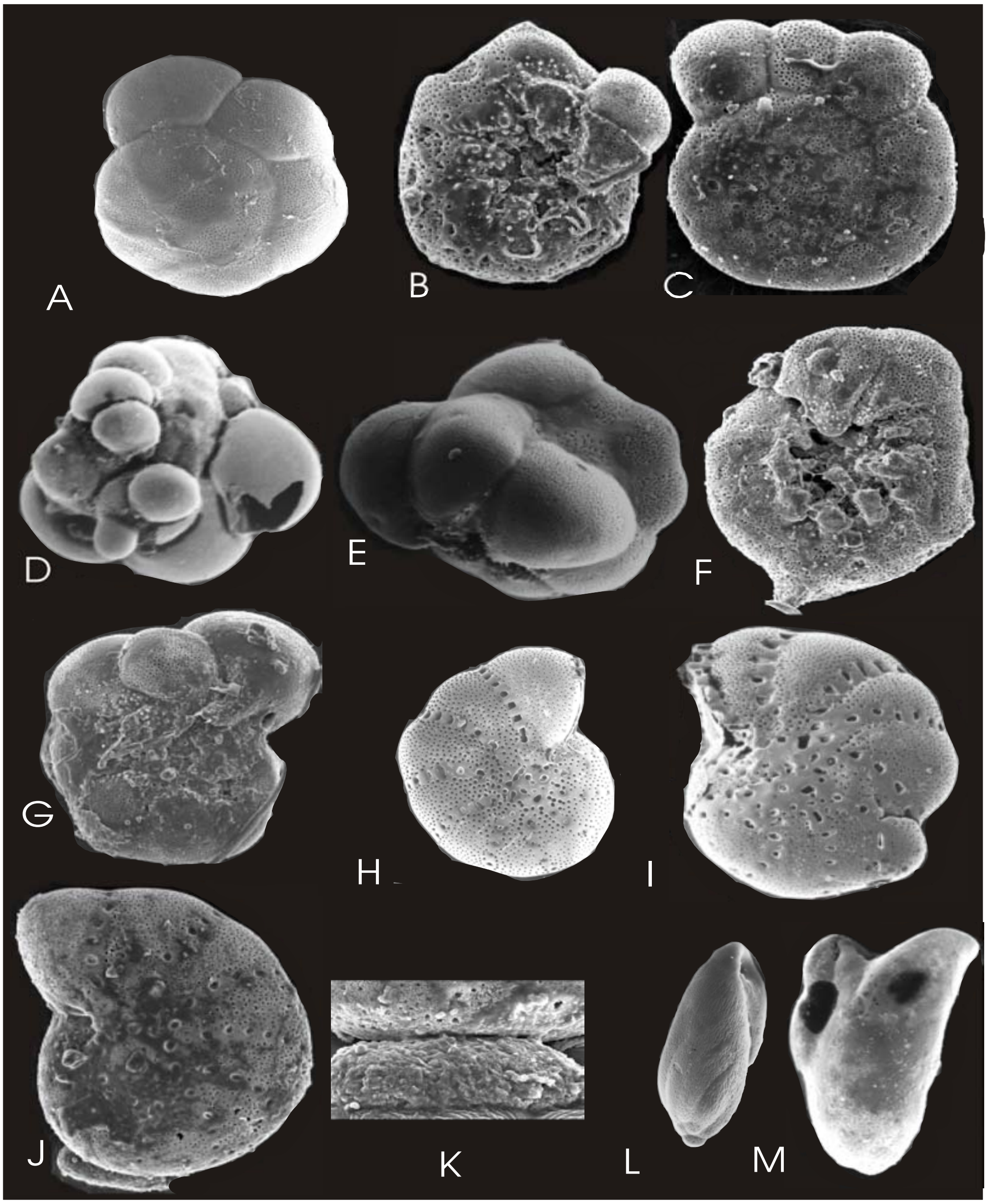


Figure 14. A – Normal specimen x 230 (*Ammonia tepida*); B – Aberrant chambers shape and size x 300 (*Ammonia tepida*); C – Overdeveloped chambers x 270 (*Ammonia tepida*); D – Complex abnormality x 250 (*Ammonia tepida*); E – Siamese twins x 380 (*Ammonia tepida*); F – Reduced size of some chambers and presence of pustule x370 (*Ammonia tepida*); G – Aberrant chamber arrangement x 170 (*Ammonia tepida*); H – Normal specimen x 220 (*Elphidium excavatum*). I – Aberrant chamber shape (the last chamber is broken) x160 (*Elphidium excavatum*); J –Additional chambers x 220 (*Elphidium excavatum*); I – Detail of figure I x 1200 (*Elphidium excavatum*); L –Normal specimen x 500 (*Buliminella elegantissima*); K – Double aperture x 330 (*Buliminella elegantissima*).

Summarising, through the integration of abiotic and biotic data, it is possible to classify the coastal area of Uruguay near Montevideo, in at least three zones with different environmental quality and degree of anthropogenic impact (Figure 15). The first one corresponds to the inner portion of Montevideo Bay and includes the Montevideo Harbour. This zone is the most impacted by heavy metals and oil derived hydrocarbons with several compounds present in concentrations potentially harmful for benthic organisms, has the highest organic load and sometimes showed evidences of lack of oxygen. As a result of these conditions, benthic diversity was very poor and species richness very low with the dominance of only one opportunistic species and nematodes, and also with high percentages of abnormal foraminifera. The second zone corresponds to the outer portion of Montevideo Bay, which is more heterogeneous. There, environmental conditions seem to be more favorable for the establishment and development of benthic organisms, due to higher hydrodynamic energy, oxygenation of the sediments, lower organic matter and contaminant concentrations than in the inner part. The greater abundance, biomass and diversity of benthic organisms recorded confirm this trend. In a global scale, the less impacted is the adjacent coastal zone of Montevideo, as was shown by the "Phylum-level Meta-analyses" approach. However, according to the AMBI results we can distinguish between Punta Yeguas and Punta Carretas, being that the former can be classified as slightly disturbed and the latter as moderately disturbed, probably, as a consequence of the sewage input through the submarine pipe. Is relevant to remark that despite a clear salinity gradient exists from the inner stations of Montevideo Bay to the outer coastal stations; through the utilization of different approaches, the occurrence of an environmental quality gradient (with the same direction) was effectively established.

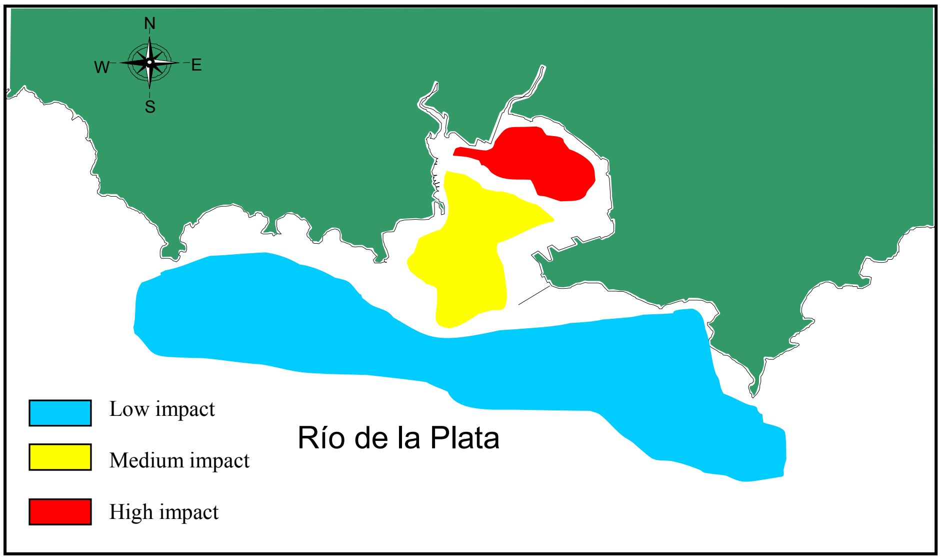


Figure 15. Global environmental classification of Montevideo Bay and the adjacent coastal zone considering abiotic and biotic variables.

BIOLOGICAL CONTAMINATION BY INVASIVE EXOTIC SPECIES

The introduction of species into a new ecosystem represents an economic and environmental serious risk. Under favourable environmental conditions, without predators, parasites and/or natural competitors these new species can reach high densities and are difficult to eliminate once established (Carlton, 1989). After their introduction into the aquatic environment invasive species may promote several changes. They can, alter the local hydrological regime, produce loss of biodiversity with the elimination of native species, changes in the trophic web, habitat modification, and also cause negative economic impacts to human populations (de Poorter 1999; Darrigran, 2002; Silva et al., 2004). In the last two centuries, introduced species have caused considerable changes in the biogeography of coastal areas, with an important decline in local biodiversity (Raffaelli & Hawkins, 1997).

Table 7. Aquatic exotic species reported in the coastal zone of Uruguay (after Orensanz et al., 2002, and Brugnoli et al. 2006)

Species	Biogeographic Origen
Artrophoda	
Crustacea, Amphipoda	
Monocorophium insidiosum	N Atlantic
Crustacea, Cirripedia	
Amphibalanus amphitrite	Cosmopolitan
Crustacea, Isopoda	
Lygia exotica	Cosmopolitan
Synidotea laevidorsalis	Japan y China
Annelida	
Polychaeta	
Boccardiella ligerica	W Europe
Ficopomatus enigmaticus	Cosmopolitan
Mollusca	
Bivalvia	
Corbicula fluminea	SE Asia
Corbicula largillierti	SE Asia
Limnoperna fortunei	SE Asia
Gastropoda	
Myosotella myosotis	Europe
Rapana venosa	Japan
Chordata	
Ascidiacea	
Styela plicata	Asia
Piscies	
Cyprinus carpio	Europe-Asia
C.carpio var. especularis	

Exotic species are aloctone organisms that can be considered as biological contamination (Ricciardi & Atkinson, 2004). According to the introduction manner they can be divided in intentional and accidental. Important vectors are intercontinental shipping and the commercial transport of aquaculture and aquarium products (Carlton, 1996; Ruiz et al., 2000; Naylor et al., 2001; Semmens et al., 2004). The intentional introduction includes aquaculture and aquarium human activities, while fouling and ballast waters are the principal vector for the accidental introductions. Shipping ballast water is known as the highest risk vector for international introductions, causing a coastal community homogenization. At any given moment it is estimated that 10,000 different species are being transported between biogeographic regions in ballast tanks alone (Carlton, 1999).

As in other areas of the world, the South-Western Atlantic contains many exotic aquatic species (Schwindt, 2001; Orensanz et al., 2002; Silva & Souza, 2004) and Uruguayan coastal ecosystems are not an exception (Maytia & Scarabino, 1979; Nión et al., 1979; Scarabino & Verde, 1995; Amestoy et al., 1998; Brugnoli, et al., 2005, Muniz et al., 2005b; Brugnoli et al., 2006). According to Brugnoli et al. (2006), Uruguay has 12 exotic invertebrate aquatic species (Table 7) accidentally introduced by ballast water. These benthic species have in common an estuarine or marine distribution in the Río de la Plata or Atlantic coastal ecosystems that is related with their salinity tolerance (Brugnoli et al., 2005; Muniz et al., 2005b; Brugnoli et al. 2006). There are reports concerning this kind of organisms in Uruguay from the beginning of this century, however, this ecological problem and its perception by the society has been increased during the last years (Masello & Menafra 1996; Orensanz et al., 2002; Muniz et al., 2005b; Brugnoli et al., 2006). Some studies about three exotic organisms that occur in the coastal zone of Uruguay are following described.

Annelida, Polychaeta, Serpulidae

Ficopomatus enigmaticus (Fauvel, 1923) is an exotic reef-building polychaete distributed in most brackish waters in temperate zones throughout the world (ten Hove & Weerdenburg, 1978) and originates in Australia (Allen, 1953). *Ficopomatus* have different sexes, external fecundation and trochophore larvae that once settled in hard substrata produces a calcareous tube that is secreted by the collar glands (Obenat & Pezzani, 1994; Obenat, 2001). Tube size and shape varies according to environmental variables, but in general the reefs are circular structures that can reach 2.5 m in diameter and grow with an estimated rate of 1.6 cm/month (Schwindt et al., 2004a). Calcareous reefs can produce large extensions in shallow water and low energy environments. These reefs can act as efficient traps for sediments, generating topographic heterogeneity that promotes changes in the abundance and distribution of the benthic associated communities (Schwindt & Iribarne, 2000). As a species producing habitat creation and modification, it is known as an ''ecosystem engineer'' (Schwindt et al., 2004b).

In Uruguay, *F. enigmaticus* has been recorded in Las Brujas stream (Monro, 1938), the outlet of Coronilla, El Bagre, Pando and Pantanoso Streams (Scarabino et al., 1975, Brugnoli et al., 2006 and refers), Montevideo Bay (Muniz et al., 2005b), Castillos, Garzón and José Ignacio coastal lagoons (Nión, 1979; Orensanz et al., 2002; Brugnoli et al., 2006), in the most inner region of the Solís Grande Stream estuary (Muniz & Venturini, 2001) (Figure 16). More recently, in the innermost region of Rocha coastal lagoon, calcareous tubes were recorded (Borthagaray et al., 2006). In the Solís Grande Stream estuary the reefs are of small size (approx. 20 cm) and are generally attached to dead and live valves of the native mollusc *Tagelus plebeius* (Muniz & Venturini, 2001). The species has a negative effect, making the benthic community health classified as heavily perturbed (Muniz et al., 2005a). In the Rocha

Lagoon, tubes of this species were found only in the shallow innermost region, where the concentration of suspended matter is high and the current speed low (Conde et al., 1999; Borthagaray et al., 2006). Borthagaray et al. (2006) developed a prediction analysis of the potential invasion rate of *Ficopomatus* in Rocha coastal lagoon, identifying ecological characteristics of different coastal ecosystems in Uruguay that promote it presence and colonisation. This species also causes negative economic impacts in Uruguay (Brugnoli et al., 2006). Recently, large tubes (twice in size to those found in natural environments) were recorded obstructing the cooling system of the ANCAP oil refinery situated in the Montevideo Bay (Muniz et al., 2005b).

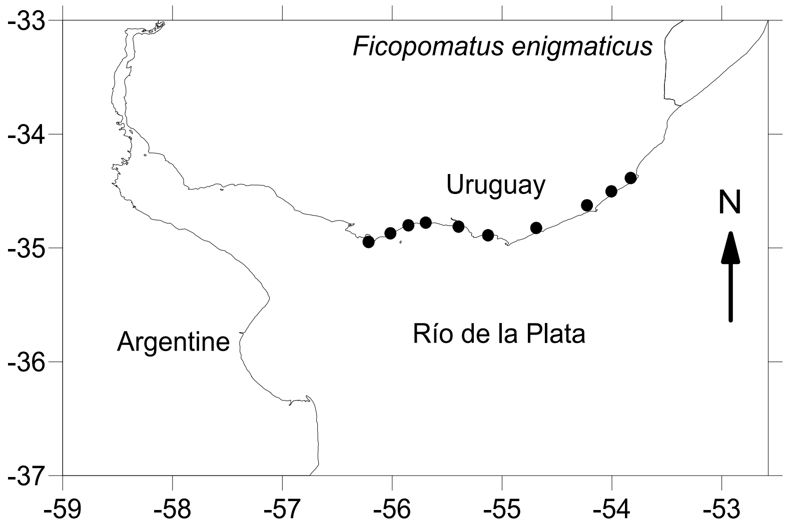


Figure 16. *Ficopomatus enigmaticus* distribution in the Uruguayan coastal zone.

Mollusca, Bivalvia, Mytilidae

Limnoperna fortunei Dunker 1857, also known as golden mussel, is a mytilid invasive species of the Río de la Plata Basin that is native from freshwater systems of China, Southeast Asia (Morton, 1997). The golden mussel was first found in South America in the coast of the Río de la Plata, Buenos Aires province (Pastorino et al., 1993). It was introduced accidentally in the region in 1991 with ballast waters (Darrigran & Pastorino, 1995). It has an epifaunal aggregate behaviour (Cataldo & Boltovskoy, 2000; Darrigran & Ezcurra de Drago, 2000) and occurs in fresh and brackish water systems, until a salinity of 3 (Darrigran, 2002). It is a dioeciously species with external fecundity and a swimming larvae phase (Cataldo & Boltovskoy, 2000). Since its arrival to the Río de la Plata Basin, this species had been found in several kinds of hard substrates, natural or artificial, showing an increase in its population abundance and changing the benthic community composition. It has been recorded in the main hydrologic freshwater systems of the region (Paraguay, Paraná, Salado and Uruguay rivers) (Darrigran & Ezcurra de Drago, 2000; Darrigran, 2002; Brugnoli et al., 2005), Los Patos Lagoon (Mansur et al., 1999, 2003) and in coastal areas of the Río de la Plata (Scarabino & Verde, 1995; Darrigran et al., 1998; Brugnoli et al., 2005).

In Uruguay, it has been recorded in five of the six main hydrographical basins: Río de la Plata, Santa Lucía, Negro and Uruguay rivers (Scarabino & Verde, 1995; Brugnoli et al.,

2005) and recently in the Merín Lagoon (Langone, 2005). According to Brugnoli et al. (2005) and Dabezies et al. (2005) salinity could be the most important environmental variable in determining its distributional limit in the Uruguayan costal zone (Figure 17). Although there are any records of this mussel in the Atlantic Basin, Brugnoli et al. (2005) suggested that it could access this basin through the San Gonzalo Channel that connects Los Patos and Merín lagoon systems. After their settlement larvae would be disperse by different vectors (ships or birds) to other aquatic ecosystems in the Atlantic Basin. Langone (2005) recently reported the occurrence of the golden mussel in the Merín Lagoon and Río Branco River and mentioned its potential dispersion by tourism boats.

The golden mussel had also been reported as a new item in the diet of native fishes in the Río de la Plata Basin (López-Armengol & Casciotta, 1998; Montalto et al., 1999; Darrigran & Ezcurra de Drago, 2000; Penchaszadeh et al., 2000), having a potential impact on the native freshwater malacofauna of Uruguay with special reference to endemic species (Scarabino, 2004) and causing problems of macrofouling in hydraulic installations (Clemente & Brugnoli, 2002; Mansur et al., 2003; Muniz et al., 2005b; Brugnoli et al., 2006).

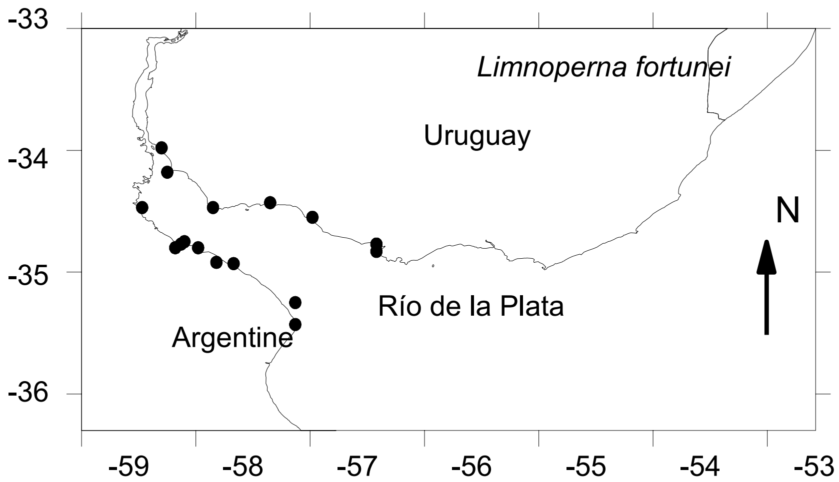


Figure 17. *Limnoperna fortunei* distribution in the coastal zone of Uruguay-Argentine.

Mollusca, Gastropoda, Muricidae

Rapana venosa (Valenciennes, 1846) (rapa whelk) is a predatory mollusc native of the Sea of Japan, Yellow Sea, Bohai Sea, and from the East China Sea to Taiwan (ICES, 2004). Nowadays, it has a distribution out of its native biogeographic range, being that ballast water transport was claimed as the main vector for the introduction of this species (ICES, 2004). This species was discovered in the Black Sea in 1947 (Drapkin, 1953), and has subsequently spread throughout the Sea of Azov, the Adriatic Sea (since 1973) (Ghisotti, 1974), the Aegean Sea (since 1990) (Koutsoubas & Voultsiadou-Koukoura, 1999) and recently in America (since 1998) (Mann & Harding, 2000; Pastorino et al., 2000). The first record of *R. venosa* in America was made in the Chesapeake Bay on the East Coast of the United States in 1998 (Harding & Mann, 1999). In 1999 was found in the Río de la Plata Estuary at the

Samborombón Bay (Argentina) (Pastorino et al., 2000). Furthermore, in the Uruguayan coastal zone was detected in Maldonado Department (intertidal zone) (Scarabino et al., 1999), in the subtidal Río de la Plata area (Rodríguez-Capítulo et al., 2003), and recently colonising successfully the outer estuarine Río de la Plata area (Carranza et al., *in press*) (Figure 18). This gastropod is a successful invader of marine coastal/brackish ecosystems, being tolerant to wide variations in temperature, salinity and oxygen concentration (Chung et al., 1993; Mann & Harding, 2003). Their high fertility (Chung et al., 1993), dispersion assisted by a planktonic larvae (Mann & Harding, 2003), and fast growth (Harding & Mann, 1999) make *R. venosa* a potentially successful invader worldwide (Savini & Occhipinti-Ambriogi, 2004). This species is a broad predator of subtidal Mollusca, it usually feeds on bivalves of economic interest like oysters, mussels and clams (Harding & Mann, 1999; Savini et al., 2002; Savini & Occhipinti-Ambriogi, 2006), and has been identified as the main reason for the collapse of several banks of mussels and oysters in the Black Sea (ICES, 2004).

This species colonised the Río de la Plata Estuary (Pastorino et al., 2000) and is successfully breeding in this area (Carranza et al., *in press*). There, it was reported in a salinity range of 14.93 and 28.24, between temperatures from 18.12 to 22.04 ºC and in a depth range of 4-12 m (Carranza et al., *in press*). Furthermore, Giberto et al. (2006) suggests the potential predation of rapa whelk on autochthonous species of the Río de la Plata such as *Mactra isabelleana* and *Ostrea puelchana* based on their coexistence with *Rapana venosa* in this area.

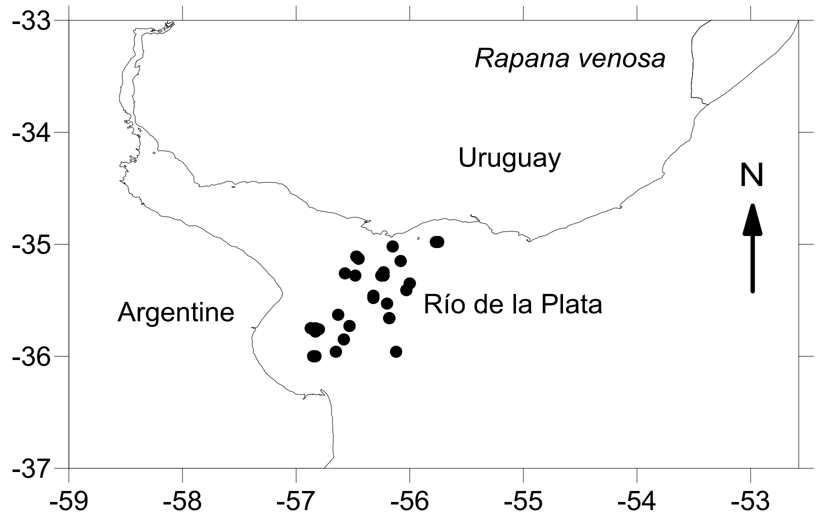


Figure 18. *Rapana venosa* distribution in the Río de la Plata area.

CONCLUSION

Although the available information about marine biodiversity and environmental perturbation in the Uruguayan coastal zone was improved during the last decades, it is still restricted to isolated areas and to some aspects of aquatic ecosystems only. The increase in the number of studies concerning benthic processes and faunal community patterns (including macro, meio and microfauna), principally in subtidal coastal zones, the continental shelf, slope and abyssal plains, compartments were they are almost non existent, is of primary importance. Thus, the implementation and development of integrative baseline studies on these topics are highly relevant, in order to contribute to the conservation of benthic biodiversity in the coastal zone of Uruguay.

As invader species are often main factors in ecological degradation, the establishment of national and international research programmes to minimise the impacts of exotic species, and to develop models that predict the introduction of new invaders is crucial. It is clear that once established in a new environment, the eradication of exotic species is very difficult. With the knowledge of the distribution, ecology, life history and impacts of alien species, both in a national and regional scale, it would be possible to improve management and control. Since shipping traffic appears to be the most important introduction vector, we emphasise the need for effective controls on ballast water discharges in the harbours of the region. The prevention of such a problem is always easier, less expensive and more effective than its eradication.

ACKNOWLEDGMENTS

These studies were partially funded by the Administración Nacional de Puertos (ANP) Uruguay, Comisión Sectorial de Investigación Científica (CSIC) and Intendencia Municipal de Montevideo (IMM) of uruguay. Also thanks to IAI-SACC, INCOFISH and IABIN (I3N) projects. The Grupo de Buceo de la Armada Nacional (GRUBU) helped with the sampling. Special thanks are also due to our numerous field collaborators from the Facultad de Ciencias, Montevideo, the Dirección Nacional de Tecnología Nuclear (DINATEN) and the Dirección Nacional de Recursos Acuáticos (DINARA). The authors are grateful for the kindly invitation of Dr. Irma C. Willis to participate as collaborators in the book.

REFERENCES

Agard, JBR; Gobin, J; Warwick RM. Analysis of marine macrobenthic community structure in relation to pollution, natural oil seepage and seasonal disturbance in a tropical environment (Trinidad, West Indies). *Marine Ecology Progress Series,* 1993 92: 233-243.

Ahumada, R; Contreras S. Contenido de metales (Ba, Cd, Co, Cr, Cu, Ni, Pb, Sr, V y Zn) en sedimentos de los fiordos y canales adyacentes a Campos de Hielo Sur. *Ciencia y Tecnología Marina*, 1999 22: 47-58.

Allen, FE. Distribution of marine invertebrates by ships. *Australian Journal of Marine and Freshwater Research,* 1953 4, 307-316.

Amaral, ACZ; Morgado, EH; Salvador, LB. Poliquetas indicadores de poluição orgânica em praias paulistas. *Revista Brasileira de Biologia,* 1998 58: 307-316.

Amestoy, F; Spinetti, M; Fabiano, G. Aquatic species introduced in Uruguay. *Verhandlungen International Vere in Limnology,* 1988 26: 2170-2173.

Baisch, PRN; Niencheski, LFH; Lacerda, LD. Trace metal distribution in sediments of the Patos Lagoon estuary, Brazil. In: *Metals in Coastal Environments of Latin America,* Seeliger, U.; Lacerda, L.D.; Patchineelam, S.R. (eds.). Springer Verlag, Berlin, 1988.

Balech, E. División zoogeográfica del litoral sudamericano. Revista Biología Marina, 1954 4: 184-195.

Balech, E. Caracteres biogeográficos de la Argentina y Uruguay. *Boletín del Instituto Biología Marina* 1964 7: 107-112.

Bastidas, C; Boné, D; García, EM. Sedimentation rates and metal content of sediments in a Venezuelan Coral Reef. *Marine Pollution Bulletin* 1999 38: 16-24.

Baptista Neto, JA; Smith, BJ; McAllister, JJ. Heavy metal concentrations in surface sediments in a nearshore environment, Jurujuba Sound, Southeast Brazil. *Environmental Pollution,* 2000 109: 1-9.

Benvenuti, CE. Benthic invertebrates. In: *Subtropical convergence marine ecosystem. The coast and the sea in the warm temperate southwestern Atlantic*, Seeliger, U; C. Odebrecht and J.P. Castello (eds.), pp. 43-46. New York: Springer Verlag, Heidelberg, 1997.

Blackburn, TM; Gaston, KJ. Spatial patterns in the species richness of birds in the *New World. Ecography,* 1996 19: 369-376.

Boltovskoy, E. Masas de agua (características, distribución, movimiento) en la superficie del Atlántico sudoeste según indicadores biológico-foraminíferos. Servicio Hidrografía naval, Buenos Aires, 1970 (H643), 99p.

Boltovskoy, E; Wright, R. *Recent Foraminífera*. Junk, The Hague, 1976.

Boltovskoy, E; Lena, H. Foraminíferos del Río de la Plata: Argentina, Servicio Hidrografía Naval, Argentina, 1974 660:1- 30.

Boltovskoy, D; Gibbons, MJ; Hutchings, L; Binet, D. General biological features of South Atlantic. South Atlantic. In: *Zooplankton*. Boltovskoy, D. (ed.), 1-42 p. Backhyus Publishers, Leiden, The Netherlands, 1999.

Borja, A; Franco, J; Pérez, V. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. *Marine Pollution Bulletin,* 2000 40: 1100-1114.

Borja, A; Muxika, I; Franco, F. The application of a marine Biotic Index to different impact sources affecting soft-bottom benthic communities along European coasts. *Marine Pollution Bulletin*, 2003 46: 835-845.

Borja, A; Franco, J; Muxika, I. The Biotic Indices and the Water Framework Directive: the required consensus in the new benthic monitoring tools. *Marine Pollution Bulletin,* 2004 48: 405-408.

Borja, A; Muxica, I; Franco, J. Long-term recovery of soft-bottom benthos following urban and industrial sewage treatment in the Nervión estuary (southern Bay of Biscay). *Marine Ecology Progress Series,* 2006 313: 43-55.

Borthagaray, A; Clemente, JM; Boccardi, L; Brugnoli, E; Muniz P. Impacto potencial de invasión de *Ficopomatus enigmaticus* (Fauvel) (Polychaeta: Serpulidae) en la Laguna de Rocha, Uruguay. *Pan-American Journal of Aquatic Sciences,* 2006 1: 57-65.

Boschi, EE. Nuevos aportes al conocimiento de la distribución geográfica de los crustáceos decápodos del Mar Argentino. *Physis, Buenos Aires, 1976* 35A: 59-68.

Boschi, EE. El ecosistema estuarial del Río de la Plata (Argentina y Uruguay). Anuario del Instituto de Ciencias del Mar y Limnología Universidad Nacional Autónoma de México, 1988 15: 159-182.

Boschi, EE. Species of decapod crustaceans and their distribution in the American marine zoogeographic provinces. Revista de Investigación y Desarrollo Pesquero (Chile), 2000 13: 7-136.

Brown, JH; Lomolino, MV. Biogeography, 2nd edn,. Sinauer Associates, Sunderland, MA, 1998, p. 691.

Brown, CJ; Cooper, KM; Meadows, WJ; Limpenny, DS; Rees, EIS. Small-scale mapping of sea-bed assemblages in the eastern English Channel using sidescan sonar and remote sensing sampling techniques. Estuarine, *Coastal and Shelf Science,* 2002 54: 263-274.

Brugnoli, E; Clemente, J; Boccardi, L; Borthagaray, A; Scarabino, F. Update and prediction of golden mussel (*Limnoperna fortunei*): distribution in the principal hydrographic basin of Uruguay. *Anais da Academia Brasileira de Ciências*, 2005 77: 235-244.

Brugnoli, E; Clemente, J; Riestra, G; Boccardi, L; Borthagaray, A. Especies acuáticas exóticas en Uruguay: situación, problemática y gestión. In: *Bases para la conservación y manejo de la costa uruguaya.* Menafra, R; Rodríguez, L; Scarabino, F; Conde, D (eds.). 351-361 p. Vida Silvestre Uruguay, Montevideo. 2006.

Burone, L; Muniz, P, Venturini, N; Sprechmann, P. Estúdio de la fauna de foraminíferos bentônicos como herramienta en la interpretación de los disturbios antropogénicos de la zona costera del departamento de Montevideo – Uruguay. In: II Simpósio Brasileiro de Oceanografia, 2004. São Paulo, Brasil.

Burone, L; Venturini, N; Sprechmann, P; Valente, P; Muniz, P. Foraminiferal responses to polluted sediments in the Montevideo coastal zone, Uruguay. *Marine Pollution Bulletin,* 2006 52: 61-73.

Burone, L; Pires-Vanin, AMS. Foraminiferal assemblages in the Ubatuba Bay, Southeastern Brazilian coast. *Scientia Marina,* 2006 70: 203-217.

Calliari, D; Cervetto, G; Gómez, M. Short-term variability in abundance and vertical distribution of the opossum shrimp *Neomysis americana* in the Solis Grande river estuary, Uruguay. Atlantica, Rio Grande, 2001 23: 117-125.

Calliari, D; Defeo, O; Cervetto, G; Gómez, M; Giménez, L; Scarabino, F; Brazeiro, A; Norbis, W. Marine life of Uruguay: critical update and priorities for future research. *Gayana,* 2003 67: 341-370.

Cardellino, R; Ferrando L. Carta geológica del Uruguay, segundo segmento (Montevideo). Sector XCVII. Universidad de la República Depto. de Publicaciones Col Ciencias Serie Especial, Montevideo, Uruguay, 1969 82pp.

Carlton, JT. Man´s role in changing the face of the ocean: biological invasions and implications for conservation of nearshore environments. *Conservation Biology*, 1989 3: 265-273.

Carlton JT. Pattern, process, and prediction in marine invasion ecology. *Biological Conservation,* 1996 78: 97-106.

Carlton, JT. Molluscan invasions in marine and estuarine communities. *Malacologia*, 1999 41: 439-454.

CARP (Comisión Administradora del Río de la Plata). Estudio para la Evaluación de la Contaminación en el Rïo de la Plata. Informe de Avance, Montevideo, 1990, 422 pp.

Carranza, A; Scarabino, F; Ortega L. Distribution of Large Benthic Gastropods in the Uruguayan Continental Shelf and Río de la Plata Estuary. *Journal of Coastal Research* (in press).

Carreón-Martínez, LB; Huerta-Díaz, MA; Nava-López, C; Siqueriros-Valencias A. Mercury and silver concentrations in sediments from the Port of Ensenada, Baja California, Mexico. *Marine Pollution Bulletin,* 2001 42: 415-418.

Cataldo, DH; Boltovskoy, D. Yearly reproductive activity of *Limnoperna fortunei* (Bivalvia) as inferred from the occurrence of its larvae in the plankton of the lower Paraná river and the Río de la Plata estuary (Argentina). *Aquatic Ecology,* 2000 34: 307-317.

Chung, EY; Kim, SY; Kim, YG. Reproductive ecology of the purple shell *Rapana venosa* (Gastropoda: Muricidae), with special reference to the reproductive cycle, depositions of egg capsules and hatchings of larvae. *Korean Journal of Malacology,* 1993 9: 1-15.

Clemente, J; Brugnoli, E. First record of *Limnoperna fortunei* (Dunker 1857) (Bivalvia: Mytilidae) in continental waters of Uruguay. *Boletín Sociedad Zoológica,* 2002 29-33.

Conde, D; Bonilla, S; Aubriot, L; de León, R; Pintos, W. Comparison of the areal amount of chlorophyll a of planktonic and attached microalgae in a shallow coastal lagoon. *Hydrobiologia,* 1999 408/409: 285-291.

Constanza, R; D'arge, R; Degroot, R; Farber, S; Grasso, M; Hannon, B; Limburg, K; Aeem, SN; O'neill, RV; Paruelo, J; Raskin, RG; Sutton, P; Van Den Belt, M. The value of the world's ecosystem services and natural capital, *Nature,* 1997 387: 253-260.

Cranston, RE; Masello, AM; Kurucz, AP. Comparing anthropogenic metal distribution in three urban regions: Montevideo (Uruguay), Halifax and Vancouver (Canada). *EcoPlata II Scientific Papers,* 2002 1:1-8.

Currie, DJ. Energy and large scale patterns of animal and plant species richness. *American Naturalist,* 1991 137: 27-49.

Dabezies, M; Gómez-Erache, M; Sans K; Brugnoli, E. Distribución de larvas planctónicas de la especie invasora *Limnoperna fortunei* en el estuario del Río de la Plata. Proceedings of the 8vas Jornadas de Zoología de Uruguay, II Encuentro de Ecología de Uruguay, Montevideo, 2005.

Dana, JD. On an isothermal oceanic chart, illustrating the geographical distribution of marine animal. American Journal Sciences. *Articles series 2*, 1853 16: 153-167.

Danulat, E; Muniz, P; García-Alonso, J; Yannicelli B. First assessment of the highly contaminated harbour of Montevideo, Uruguay. *Marine Pollution Bulletin,* 2002 44: 551-576.

Darrigran, G. Macroinvertebrates associated with *Limnoperna fortunei* (Dunker, 1857) (Bivalvia, Mytilidae) in Río de la Plata, *Argentina. Hydrobiologia,* 1998 367: 223–230.

Darrigran, G. Potential impact of filter-feeding invaders on temperate inland freshwater environments. *Biological Invasions,* 2002 4: 145-156.

Darrigran, G; Pastorino, G. The recent introduction of asiatic bivalve, *Limnoperna fortunei* (Mytilidae) in to South America. *The Veliger,* 1995 38: 183-187.

Darrigran, G; Ezcurra de Drago, I. Invasion of the exotic freshwater mussel *Limnoperna fortunei* (Dunker 1857) (Bivalvia: Mytilidae) in South America. *The Nautilus,* 2000 114: 69-73.

Dauer, DM. Biological criteria, environmental health and estuarine macrobenthic community structure. *Marine Pollution Bulletin,* 1993 26: 249-257.

Dauer, DM; Alden, RV. Long-term in the macrobenthos and water quality of the Lower Chesapeake bay. *Marine Pollution Bulletin,* 1995 30: 840-850.

Dauer, DM; Conner, WG. Effects of moderate sewage input on benthic polychaete populations. *Estuarine and Marine Sciences,* 1980 10: 335-346.

Day, JW; Hall, CAS;Kemp, JM; Yañez-Arancibia, A. Estuarine ecology. Wiley & Sons publications, 1989, 558 pp.

de Poorter, M. Borrador de Guías para la prevención de pérdidasde diversidad biológica ocasionadas por invasion biológica. Cuarta Reunión del Órgano Subsidiario de Asesoramiento Científico, Técnico y Tecnológico. Documento de Base. Unión Internacional para laConservación de la Naturaleza (UICN). 1999, 15 pp.

Debeney, J. P; Duleba, W; Bonetti, C; Melo e Souza, SH; Eichler, B. Pararotalia cananeiaensis N. SP.: indicator of marine influence and water circulation in Brazilian coastal and paralaic environments. *Journal of Foraminiferal Research,* 2001 31: 152-163.

Defeo, O; de Alava, A. Effects of human activities on long-term trends in sandy beach populations: the wedge clam *Donax hanleyanus* in Uruguay. *Marine Ecology Progress Series,* 1995 123: 73-82.

Defeo, O; de Alava, A. Impacto humano en la biodiversidad de playas arenosas. Un estudio de caso. In: Manejo de zonas costeras y su impacto en la biodiversidad: estudios de caso. Moreno-Casasola P., M.L. Martínez & O. Defeo (eds.). Red Iberoamericana de Biodiversidad de Ecosistemas Costeros CYTED-Instituto de Ecología, Xalapa, Veracruz, 2002: pp. 169-179.

Defeo, O; McLachlan, A. Patterns, processes and regulatory mechanisms in sandy beach macrofauna: a multi-scale analysis. *Marine Ecology Progress Series*, 2005 295: 1-20.

Defeo, O; Brazeiro, A; Riestra, G. Impacto de la descarga de un canal artificial en la biodiversidad de gasterópodos en una playa de arena de la costa atlántica uruguaya. *Comunicaciones de la Sociedad Malacológica del Uruguay*, 1996 8: 13-18.

D´Orbigny, AD. 1835-43. Voyage dans l´Ameriwue Méridionale execute pendant les annés 1826, 1827, 1828, 1829, 1830, 1831, 1832 et 1833. Tome V. partie 3. Mollusques. Paris et Strasbourg, 1845-55. Mollusques viviant et fossils. Paris. 1853. histoire physique, politique et naturalle de l´ Ile de Cuba par M. Ramón de la Sagra. Mollusques para A. D. d´Orbigny. Paris.

Drake, P; Baldó, F; Sáenz, V; Arias, AM. Macrobenthic Community Structure in Estuarine Pollution Assessment on the Gulf of Cádiz (SW Spain): is the Phylum-level Meta-analysis Approach Applicable?. *Marine Pollution Bulletin,* 1999 38: 1038-1047.

Drapkin, E. Novii mollusc V. Cemom more. Prirada, 1953 8: 92-95.

Engle, VD; Summers, JK; Gaston, GR Benthic indices of environmental condition of Gulf of Mexico estuaries. *Estuarios*, 1994 17: 372-389.

Escofet, A; Gianuca, N; Maytía, S; Scaraino, V. Playas arenosas del Atlántico Sudoccidental entre los 298 y 438 LS.: consideraciones generales y esquema biocenológico. Memorias del Seminario sobre Ecología Bentónica y Sedimentación de la Plataforma Continental del Atlántico Sur, 1979 1, 245–258.

Fischer, AG. Latitudinal variations in organic diversity. *Evolution*, 1959 14: 64-81.

Gaines, SD; Lubchenco, J. A united approach to marine plant herbivore interactions. II. Biogeography. *Annual Review in Ecology and Systematics,* 1982. 13: 111 -138.

Ghisotti, F. *Rapana venosa* (Valenciennes) nuova ospite adiratica? Conchiglie Milano, 1974 10: 125-126.

Giberto, DA; Bremec, CS; Acha, EM; Mianzan, H. Large-scale spatial patterns of benthic assemblages in the SW Atlantic: the Río de la Plata estuary and adjacent shelf waters. *Estuarine*, *Coastal and Shelf Science,* 2004 61: 1-13.

Giberto, DA; Bremec, CS; Schejter, L; Schiariti, A; Mianzán, H; Acha, EM. The invasive Rapa Whelk *Rapana venosa* (Valenciennes 1846): status and potential ecological impacts in the Río de la Plata estuary, Argentina-Uruguay. *Journal of Shellfish Research*, 2006 25:919-924

Giménez, L; Borthagaray, A; Rodríguez, M; Brazeiro, A; Dimitriadis, K. Scale-dependent patterns of macrofaunal distribution in soft-sediment intertidal habitats along a large-scale estuarine gradient. *Helgoland Marine Research,* 2005 59: 224-236.

Giménez, L; Dimitriadis, C; Carranza, A; Borthagaray, AI; Rodríguez, M. Unravelling the complex structure of a benthic community: A multiscale-multianalytical approach to an estuarine sandflat. *Estuarine, Coastal and Shelf Science*, 2006 68: 462-472.

Grall, J; Glémarec, M. Using biotic indices to estimate macrobenthic community perturbations in the bay of Brest. *Estuarine, Coastal and Shelf Science,* 1997 44: 43-53.

Grasshoff, K. Determination of oxygen. In: *Methods of Seawater Analysis.* Grasshoff E & Kremling (eds.), 1983, pp 61-72. 2nd ed. Verlag Chemie, Weinheim.

Gray, JS. Pollution-induced changes in populations. Phil Transactions of the Royal Society of London B, 1979 286: 545-561.

Guerrero, RA; Acha, EM; Framiñan, MB; Lasta, CA. Physical oceanography of the Rio de la Plata estuary, Argentina. *Continental Shelf Research,* 1997 17: 727-742.

Hall, JA; Raffaelli, DG,;Thrush, SF. Patchiness and disturbance in shallow water benthic assemblages. In: Giller P Hildrew A raffaelli D (eds*.), Aquatic ecology: Scale, pattern and Process.* Blackwell Science, 1994, Oxford.

Harding, JM; Mann, R. Observations on the biology of the veneid rapa whelk, *Rapana venosa* (Valenciennes, 1846) in the Chesapeake Bay. *Journal of Shelfish Research,* 1999 18:9-17.

Heip, C. Benthic studies: summary and conclusions. *Marine Ecology Progress Series,* 1992.91: 265-269.

Heip, C. Eutrophication and zoobenthos dynamics. *Ophelia*, 1995 41: 113-136.

Heip, C; Basford, DJ; Craeymeersch, JA; Dewarumez, JM; Dorjes, J; de Wilde, PAJ; Duineveld, GCA; Eleftheriou, A; Herman, PMJ; Niemann, U; Kunitzer, A; Rachor, E; Rumohr, H; Soetaert, K; Soltwedel, T. Trends in biomass, density and diversity of North Sea macrofauna. *ICES Journal of Marine Science,* 1992 49: 13-22.

Hillebrand, H. Strenght, slope and variability of marine latitudinal gradients. *Marine Ecology Progress Series*, 2004 273: 251-276.

Hilly, C; Le Bris, H; Glémarec, M. Impacts biologiques des emissaries urbains sur les écosystèmes benthiques. *Oceanis*, 1986 12: 419-426.

Howarth, RW; Boyer, EW; Pabich, WJ; Galloway, JN. Nitrogen use in the United States from 1961-2000 and Potencial Future Trends. *A Journal of the Human Environment,* 2000 31: 88-96.

ICES. Alien Species Alert: *Rapana venosa* (veined whelk). Edited by Roger Mann, Anna Occhipinti, and Juliana M. Harding. ICES, 2004, Cooperative Research Report N° 264. 14 pp.

Ieno, EN; Bastida, RO. Spatial and temporal patterns in coastal macrobenthos of Samboronbon Bay, Argentina: a case study of very low diversity. *Estuaries*, 1998 21: 690-699.

Jørgensen, B. Material flux in the sediment. In: Jørgensen B and Richardson K. (eds.). Coastal and estuarine studies. *American Geophysical Union.* 1996, p: 115-135.

Koustsoubas, D; Voultsiadou-Koukoura, E. The occurrence of *Rapana venosa* (Valeciennes, 1846)(Gastropoda, Thaididae) in the Aegean Sea. *Bolletino Malacologico*, Milano, 1999 26: 201-204.

Lacerda, LD; Huertas, R; Moresco, HF; Carrasco, G; Viana, F; Lucas, R; Pessí, M. Trace metal concentrations and geochemical partitioning in arroyo Carrasco wetlands, Montevideo, Uruguay. *Geochimica Brasiliensis,* 1998 12: 63-74.

Lacerda, LD; Pfeiffer, WC; Fiszman, M. Níveis de metais pesados em sedimentos marinhos da Baía da Ribeira, Angra dos Reis. *Ciência e Cultura,* 1982 34: 921-924.

Langone, JA. Notas sobre el mejillón dorado *Limnoperna fortunei* (Dunker 1857)(Bivalvia, Mytilidae) en Uruguay. Publicación extra Museo Nacional de Historia Natural y Antropología, Montevideo, 2005 1:1-18.

Lalli, CM; Parsons, TR. Biological Oceanography. An introduction. Butterworth Heinemann, Oxford, 1997, 314 pp.

Lercari, D; Defeo, O. Effects of freshwater discharge in sandy beach populations: the mole crab *Emerita brasiliensis* in Uruguay. *Estuarine, Coastal and Shelf Science*, 1999 49: 457-468.

Lercari, D; Defeo, O. Variation of a sandy beach macrobenthic community along a human-induced environmental gradient. *Estuarine, Coastal and Shelf Science,* 2003 58: 17-24.

Lercari, D; Defeo, O. Large-scale diversity and abundance trends in sandy each macrofauna along full gradients of salinity and morphodynamics. *Estuarine, Coastal and Shelf Science*, 2006 68: 27-35.

Lercari, D; Defeo, O; Celentano, E. Consequences of a freshwater canal discharge on the benthic community and its habitat on an exposed sandy beach. *Marine Pollution Bulletin,* 2002 44: 1392-1399.

Loeblich, A R; Tappan, H. *Foraminiferal genera and their classification.* Van Nostrand Reinold, New York, 1988 vol. 1 e 2, 270p. + 212p, 847 pls.

López-Armengol, MF; Casciotta, J. First record of the predation of the introduced freshwater bivalve *Limnoperna fortunei* (Mytilidae) by the native fish Micropogonias (Scianidae) in the Río de la Plata estuary, South America. *Iberus,* 1998, 105-108.

López-Jamar, E. Distribución espacial del poliqueto *Spiochaetopterus costarum* en las Rías Bajas de Galicia y su posible utilización como indicador de contaminación orgánica en el sedimento. *Boletín del Instituto Español de Oceanografía,* 1985 2: 68-76.

López-Laborde, E; Nagy, G. Hydrography and sediment transport characteristics in the Río de la Plata. In: Perillo Piccolo & Pino-Quivira (eds.). Estuaries of the South America: their geomorphology and dynamics. Springer-Verlag, Berlin, 1999.

López-Laborde, J; Perdomo, A; Gómez-Erache, M. Diagnóstico Ambiental y Socio-Demográfico de la Zona costera Uruguaya del Río de la Plata. Compendio de los principales resultados. ECOPLATA. Montevideo, 2000, 177 pp.

MacDonald, DD; Carr, SR; Calder, FD; Long, ER; Ingersol, CG. Development and evaluation of sediment quality guidelines for Florida coastal waters. *Ecotoxicology,* 1996 5: 253-278.

Machado, W; Silva-Filho, EV; Oliveira, RR; Lacerda, LD. Trace metal retention in mangrove ecosystems in Guanabara Bay, SE Brazil. *Marine Pollution Bulletin*, 2002 44: 1277-1280.

Mann, R; Harding, JM. Salinity tolerance of larval *Rapana venosa*: implications for dispersal and establishment of an invading predatory gastropod on the North American Atlantic coast. *Biological Bulletin,* 2003 204: 96-103.

Mann, R; Harding, JM. Invasion of the North American Atlantic coast by a large predatory Asian mollusk. *Biological Invasions,* 2000 2: 7-22.

Mansur, MC; Valer, RM; Aires, N. *Limnoperna fortunei* (Dunker 1857) molusco bivalve invasor na bacia do Guaíba, Río Grande do Sul, Brasil. *Biociencias*, 1999 7: 147-149.

Mansur, MC; Santos, CP; Darrigran, G; Heydrich, I; Callil, CT; Cardoso, FR. Primeiros dados quali-quantitativos do “mexilhao dourado”, *Limnoperna fortunei* (Dunker, 1857), no lago Guaíba, Bacia da laguna dos Patos, Rio Grande do Sul, Brasil e alguns aspectos de sua invasao no novo ambiente. Revista Brasileira de Zoologia, *Curitiba,* 2003 22: 75-84.

Masello, A; Menafra R. Macrobenthic community of the uruguayan coast zone and adjacent areas. In: EcoPlata Team (Eds.) *The Río de la Plata. An environmental overview.* Dalhousie University, Halifax, Nova Scotia, 1996 pp:113- 162.

Maytia, S; Scarabino, V. Las comunidades del litoral rocoso del Uruguay: zonación, distribución local y consideraciones biogeográficas. Memorias del Seminario sobre Ecología Bentónica y Sedimentación de la Plataforma Continental del Atlántico Sur. UNESCO, 1979, pp: 149-160.

Méndez, SM; Anciaux, F. Efectos en las características del agua costera provocados por la descarga del Canal Andreoni en la playa de La Coronilla (Rocha, Uruguay). *Frente Marítimo*, 1995 8:101-107.

Méndez, N; Flos, J; Romero, J. Littoral soft-bottom polychaete communities in a pollution gradient in front of Barcelona (Western Mediterranean, Spain). *Bulletin of Marine Science*, 1998 63: 167-178.

Mianzán, H; Lasta, C; Achas, M; GUERRERO, R; Macchi, G; Bremec, C. The Río de la Plata estuary, Argentina-Uruguay: In: Seeliger, U. & B. Kjerfve (eds.) Coastal Marine ecosystems of Latin America, 2001. Springer-Verlag, Berlin*. Ecological Studies Series* 144: 185-204.

Mirlaen, N; Baraj, B; Niencheski, LF; Baisch, P; Robinson, D. Effect of accidental sulphuric acid on metal distributions in estuarine sediment of Patos Lagoon. *Marine Pollution Bulletin,* 2001 42: 1114-1117.

Montalto, L; Oliveros, O; Ezcurra de Drago, I; Demonte, l. Peces del río Paraná medio, predadores de una especie invasora *Limnoperna fortunei* (Bivalvia: Mytilidae). *Revista FABICIB*, 1999 3: 85-101.

Monro, CCA. On a small collection of Polychaeta from Uruguay. *Annals and Magazine of Nature History,* 1938 2: 311-314.

Moresco, H; Dol, I. Metales en sedimentos de la Bahía de Montevideo. *Revista de la Asociación de Ciencias Naturales del Litoral,* 1996 27: 1-5.

Morton, B. Freshwater fouling bivalves. Proceed. First International *Corbicula* Sympossium. Texas Univ, 1997 p: 1-14

Moyano, M; Moresco, H; Blanco, J; Rosadilla, M; Caballero, A. Baseline studies of coastal pollution by heavy metals, oil and PAHs in Montevideo. *Marine Pollution Bulletin,* 1993 26: 461-464.

Mucha, AP; Vasconcelos, MTDS; Bordalo, AA. Macrobenthic community in Douro estuary: relations with heavy metals and natural sediment characteristics. *Environmental Pollution,* 2002 121: 169-180.

Muniz, P; Pires, AMS. Polychaete associations in a subtropical environment (São Sebastião Channel, Brazil): a structural analysis. *Marine Ecology* PSZN, 2000 21: 145-160.

Muniz, P; Venturini, N. Spatial distribution of the macrozoobenthos in the Solís Grande Stream estuary (Canelones-Maldonado, Uruguay). *Brazilian Journal of Biology,* 2001 61: 409-420.

Muniz, P; Venturini, N; Rodríguez, M; Martínez, A; Lacerot, G; Gómez, M. Benthic communities in a highly polluted urban bay. In: Milón Delgado Paredes Paredes & Benavides (eds.). Ecología y Desarrollo Sostenible: Reto de América Latina para el Tercer Milenio. Arequipa, Perú, 2000 274pp.

Muniz, P; Venturini, N; Martínez, A. Physico-chemical characteristics and pollutants of the benthic environment in the Montevideo coastal zone, Uruguay. *Marine Pollution Bulletin,* 2002 44: 962- 968.

Muniz, P; Danulat, E; Yannicelli, B; Garcia-Alonso, J; Medina, G; Bícego, MC. Assessment of contamination by heavy metals and petroleum hydrocarbons in sediments of Montevideo Harbour (Uruguay). *Environment International,* 2004a 29: 1019-1028.

Muniz, P; Venturini, N; Gómez-Erache, M. Spatial distribution of chromium and lead in the benthic environment of coastal areas of the Río de la Plata estuary (Montevideo, Uruguay). *Brazilian Journal of Biology,* 2004b 64: 103-116.

Muniz, P ; Venturini, N ; Pires-Vanin, A M S ; Tommasi, L R ; Borja, A. Testing the applicability of a Marine Biotic Index (AMBI) to assessing the ecological quality of soft-bottom benthic communities, in the South America Atlantic region. *Marine Pollution Bulletin*, 2005a 50: 624-637.

Muniz, P; Clemente, J; Brugnoli, E. Benthic invasive pests in Uruguay: a new problem or an old one recently perceived? *Marine Pollution Bulletin,* 2005b 50: 1014-1018.

Murray, JW. Ecology and Paleoecology of Benthic Foraminifera. Longman Scientific and Technical/Wiley U K/ New York, 1991 397pp.

Muxika, I; Borja, A; Bonne, W. The suitability of the marine biotic index (AMBI) to new impact sources along European coasts. *Ecological Indicators*, 2005 5:19–31.

Nagy, G; Martínez, C; Caffera, MR; Pedrosa, G; Forbes, EA; Perdomo, A. The hydrological and climatic setting of the Río de la Plata. In: Wells & Daborn (eds.). The Río de la Plata, an environmental overview. An EcoPlata project background report, pp: 17-70. Dalhousie University, Halifax, Nova Scotia, 1997.

Nagy, GJ; Gómez-Erache, M; López, CH; Perdomo, AC. Distribution patterns of nutrients and symptoms of eutrophication in the Rio de la Plata estuary. *Hydrobiologia*, 2002 475/476: 125-139.

Naylor, RL; Williams, SL; Strong, DR. Aquaculture—a gateway for exotic species. *Science,* 2001 294:1655–1656.

Nión, H. Zonación del macrobentos en un sistema lagunar litoral oceánico. Seminario sobre Ecología Bentónica y Sedimentación de la Plataforma Continental del Atlántico Sur. (Montevideo, Uruguay), Memória UNESCO, 1979 1: 225-235.

Obenat, SM; Pezzani, EE. Life cycle and population structure of the polychaete *Ficopomatus enigmaticus* (Serpulidae) in Mar Chiquita coastal lagoon, Argentina. *Estuaries*, 1994 17: 263-270.

Obenat, S. Biología del anélido introducido *Ficopomatus enigmaticus* (Polychaeta, Serpulidae). In: Iribarne, O (ed). Reserva de Biósfera Mar Chiquita: Características físicas, biológicas y ecológicas. Editorial Martin. Mar del Plata, 2001.

Orensanz, JM; Schwindt, E; Pastorino, G; Bortolus, A; Casas, G; Darrigran, G; Elías, R; López-Gappa, JJ; Obenat, S; Pascual, S; Penchaszadeh, P; Piriz, ML; Scarabino, F; Spivak, ED; Vallarino, E. No longer the pristine confines of the world ocean: a survey of exotic marine species in the southwestern Atlantic. *Biological Invasions,* 2002 4: 115-143.

Oug, E; Kristoffer, N; Brage, R. Relationship between soft bottom macrofauna and polycyclic aromatic hydrocarbons (PAH) from smelter discharge in Norwegian fjords and coastal waters. *Marine Ecology Progress Series,* 1998 173: 39-52.

Palacio, FJ. Revisión zoogeográfica marina del sur de Brasil. Boletim do Instituto Oceanográfico, São Paulo, 1982 31: 69-92.

Pastorino, G; Darrigran, G; Martin, S; Lunaschi, G. *Limnoperna fortunei* (Dunker 1857) (Mytilidae), nuevo bivalvo invasor en aguas del Río de la Plata. *Neotrópica*, 1993 39: 34.

Pastorino, G; Penchaszadeh, PE; Schejter, L; Bremec, C. *Rapana venosa* (Valenciennes, 1846) (Mollusca: Muricidae): a new gastropod in south Atlantic waters. *Journal of Shelfish Research,* 2000 19: 897-899.

Pearson, TH; Rosenberg, R. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology Annual Review,* 1978 16: 29-331.

Penchaszadeh, PE; Darrigran, G; Angulo, C; Averbuj, A; Brignoccoli, N; Brogger, M; Dogliotti, A; Pirez, N. Predation on the invasive freshwater mussel *Limnoperna fortunei* (Dunker 1857) (Mytilidae) by the fish *Leporinus obtusidens* Valenciennes 1846 (Anostomidae) in the Rio de la Plata, Argentina. *Journal of Shellfish Research,* 2000 19: 229-231.

Pianka, ER. Latitudinal gradients in species diversity: a review of concepts. *American Naturalist,* 1966 100: 33-46.

Raffaelli, D; Hawkins, S. Intertidal ecology. Chapman & Hall. London, 1997, 356pp.

Ricciardi, A; Atkinson SK. Distinctiveness magnifies the impact of biological invaders in aquatic ecosystems. *Ecology Letters,* 2004 7: 781-784.

Rakocinski, CF; Brown, SS; Gaston, GR; Heard, RW; Walker, WW; Summers, JK. Species-abundance-biomass responses by estuarine macrobenthos to sediment chemical contamination. *Journal of Aquatic Ecosystem Stress and Recovery*, 2000 7: 201-214.

Rees, HL; Pendle, MA; Waldock, O; Limpenny, DS; Boyd, SE. A comparison of benthic biodiversity in the North Sea, English Channel and Celtic Seas. *ICES Journal of Marine Science,* 1999 56: 228-246.

Rex, MA; Stuart, CT; Hessler, RR; Allen, JA; Sanders, HL; WILSON, GDF. Global scale latitudinal patterns of species diversity in the deep sea benthos. *Nature*, 1994 365: 636-639.

Riestra, G; Giménez, LJ; Scarabino, V. Análisis de la comunidad macrobentónica infralitoral de fondo rocoso en Isla Gorriti e Isla de Lobos (Maldonado, Uruguay). Frente Marítimo (Uruguay), 1992 11: 123-127.

Rohde, K. Latitudinal gradients in species diversity: the search for the primary cause. *Oikos*, 1992 65, 514-527.

Ritter, C; Montagna, PA. Seasonal hypoxia and models of benthic response in a Texas Bay. *Estuaries*, 1999 22: 7-20.

Rodriguez-Capitulo, A; Cortelezzi, A; Paggi, AC; Tangorra, M. Phytoplankton and Benthos of the environmental survey of the Río de la Plata. N° 2. Benthos. Technical report United Nations Development Programme-Global Environmental Facilities. PNUD Project/Gef RLA/99/G31, 2003. 48 pp. (In Spanish). http://www.freplata.org/documentos/tecnico.asp.

Rodríguez-Gallego, L; Conde, D; Rodríguez-Graña, L. Las Lagunas Costeras de Uruguay: estado actual del conocimiento. Simposio CARP-COFREMAR (V Jornadas de Ciencias del Mar). Mar del Plata, Argentina, 2003.

Ruiz, GM; Fofonoff, PW; Carlton, JT; Wonham, MJ; Hines, AH. Invasion of coastal marine communities in North America: apparent patterns, processes, and biases. *Annual Review of Ecology and Systematics,* 2000 31: 481-531.

Saiz-Salinas, JI. Evaluation of adverse biological effects induced by pollution in the Bilbao Estuary (Spain). *Environmental Pollution,* 1997 96: 351-359.

Sánchez-Mata, A; Glémarec, M; Mora, J. Physico-chemical structure of the benthic environment of a Galician ría (Ría de Ares-Betanzos, north-west Spain). *Journal of the Marine Biological Association of the United Kingdom,* 1999 79: 1-21.

Sanvicente-Anorve, L; Lepretre, A; Davoult, D. Diversity of benthic macrofauna in the eastern English Channel: comparison among and within communities. *Biodiversity and Conservation,* 2002 11: 265-282.

Savini, D; Harding, JM; Mann, R. Rapa whelk *Rapana venosa* (Valencienne, 1846) predation rates on hard clams *Mercenaria mercenaria* (Linnaeus, 1758). *Journa of Shellfishes Research,* 2002 21: 777-779.

Savini, D; Occhipinti-Ambriogi, A. Spreading potential of an invader: *Rapana venosa* in the Northern Adriatic sea. *Rapp Comm Int Mer Médit.,* 2004 37: 548.

Savini, D; Occhipinti-Ambriogi, A. Comsumption rates and prey preference of the invasive gastropod *Rapana venosa* in the Northern Adriatic Sea. *Helgoland Marine Research,* 2006 60: 153-159.

Savage, C; Field, JG; Warwick, RM. Comparative meta-analysis of the impact of offshore marine mining on macrobenthic communities versus organic pollution studies. *Marine Ecology Progress Series,* 2001 221: 265-275.

Scarabino, V; Maytía, S; Caches, M. Carta binómica litoral del departamento de Montevideo. I. Niveles superiores del sistema litoral. *Comunicaciones de la Sociedad Malacológica del Uruguay*, 1975 4: 117-129.

Scarabino, F; Menafra, R; Etchegaray, P. Presencia de *Rapana venosa* (Valenciennes, 1846) (Gastropoda: Muricidae) en el Río de la Plata. Boletín de la Sociedad Zoológica del

Uruguay (Segunda Epoca) 11 (Actas de las V Jornadas de Zoología del Uruguay), Montevideo, 1999 11: 40.

Scarabino, F; Verde, M. *Limnoperna fortunei* (Dunker 1857) en la costa uruguaya del Río de la Plata (Bivalvia: Mytilidae). *Comunicaciones de la Sociedad Malacológica del Uruguay*, 1995 7: 374-375.

Scarabino, F. Conservación de la malacofauna uruguaya. *Comunicaciones de la Sociedad Malacológica de Uruguay*, 2004 8: 267-273

Schwindt, E. Impacto de un poliqueto exótico y formador de arrecifes. In: Iribarne, O. (Ed), Reserva de la Biosfera mar Chiquita: Características físicas y ecológicas. Ed. Martin, Mar del Plata, 2001: 109-113.

Schwindt, E; Iribarne, O. Settlement sites, survival and effects on benthos of an introduce reef-building polychaete in a SW Atlantic coastal lagoon. *Bulletin of Marine Science,* 2000 67: 73-82.

Schwindt, E; Iribarne, O; Isla, F. Physical effects of an invading reef-building polychaete on an Argentina estuarine environment. *Estuarine Coastal and Shelf Science,* 2004a 59: 109-120.

Schwindt, E; De Francesco, C; Iribarne, O. Individual and reef growth of the invasive reef-building polychaete *Ficopomatus enigmaticus* in a south-western Atlantic coastal lagoon. *Journal of the Marine Association of the United Kingdom*, 2004b 84: 987-993.

Sealey, K S; Bustamante, G. Setting geographic priorities for marine conservation in Latin America and the Caribbean. *The Nature conservancy,* Arlington, Virginia, 1999, 125 pp.

Semmens, BX; Buhle, ER; Salomon, AK; Pattengill-Semmens, CV. A hotspot of non-native marine fishes: evidence for the aquarium trade as invasion pathway. *Marine Ecology Progress Series,* 2004 266: 239-244.

Silva, JSV da; Souza, RCCL de. Água de lastro e bioinvasão. Editora Interciência, Rio de Janeiro, Brazil, 2004.

Silva, J SV da; Costa, F; Correa, RC; Sampaio, KT; Danelon, OM. Agua de lastro e bioinvaşao. In: Silva, JVS da; Souza, RCCL (eds.). Agua de lastro e bioinvaşao. pp 1-10. Interciencia, Río de Janeiro, 2004.

Tam, J; Carrasco, FD. Macrobenthic sublittoral species assemblages from Central Chile along a global scale of perturbation. *Gayana Oceanología,* 1997 5: 107-113.

ten Hove, HA; Weerdenburg, CA. A generic revision of the brackish-water serpulid *Ficopomatus* Southern 1921 (Polychaeta: Serpulidae), including *Mercierella* Fauvel 1923, *Sphaeropomatus* Treadwell 1934, *Mercierellopsis* Rioja 1954, and *Neopomatus* Pillai 1960. *Biological Bulletin,* 1978 154: 96-120.

Tenore, KR. Macrobenthos of the Pamlico River Estuary, North Carolina. *Ecological Monographs,* 1972 42: 51-69.

Thouzeau, G; Robert, G; Ugarte, R. Faunal assemblages of benthic megainvertebrates inhabiting sea scallop grounds from eastern Georges Bank in relation to environmental factors. *Marine Ecology Progress Series,* 1991 74: 61-82.

Thrush, SF; Hewitt, JE; Norkko, A; Nicholls, PE; Funnell, GA; Ellis, JI. Habitat change in estuaries: predicting broad-scale responses of intertidal macrofauna to sediment mud content. *Marine Ecology Progress Series,* 2003 263: 101-112.

Tundisi, JG; Matsumura-Tundisi, T ; Rocha, O. Ecossistemas de águas interiores. p. 153-194. In: Águas doces no Brasil. Capital ecológico, uso e conservação. Rebouças, A; Braga, B; Tundisi, JG (eds.), São Paulo. Escritura Editora, 1999.

Turner, SJ; Trush, SF; Pridmore, RD; Hewitt, JE; Cummings, VJ; Maskery, M. Are soft-sediment communities stable? An example from a windy harbour. *Marine Ecology Progress Series,* 1995 120: 219-230.

Van Dolah, RF; Hyland, JL; Holland, AF; Rosen, JS; Snoots, TR. A benthic index of biological integrity for assessing habitat quality in estuaries of the southeastern USA. *Marine Environmental Research*, 1999 48: 269-283.

Venturini, N; Muniz, P.Status of Sediment Pollution and Macrobenthic Subtidal Communities of Montevideo Coastal Zone, Uruguay. In: IX CONGRESO LATINOAMERICANO SOBRE CIENCIAS DEL MAR, 2001, San Andrés. Anales del IX Congreso Latinoamericano sobre Ciencias del Mar, 2001.

Venturini, N; Muniz, P; Rodríguez, M. Macrobenthic subtidal communities in relation to sediment pollution: the phylum-level meta-analysis approach in a south-eastern coastal region of South America. *Marine Biology,* 2004 144: 119-126.

Viana, F; Huertas, R; Danulta, E. Heavy Metal Levels in Fish from Coastal Waters of Uruguay. *Archives of Environmental Contamination and Toxicology,* 2005 48: 530-537.

Villa, N. Spatial distribution of heavy metals in seawater and sediments from the coastal areas of the southern Buenos Aires Province, Argentina. In: Seeliger, U.; Lacerda. L.D.; Patchineelam, S.R. (eds.) Metals in coastal environments of Latin America. Springer Verlag, Berlin, 1988.

Walton, W R. Techniques for recognition of living foraminifera. *Contr. Cushman. Fnd. form. Res* 1952 3:56- 60.

Warwick, RM. A new method for detecting pollution effects on marine macrobenthic communities. *Marine Biology,* 1986 92: 557-562.

Warwick, RM; Clarke, KR. Comparing the severity of disturbance: a meta-analysis of marine macrobenthic community data. *Marine Ecology Progress Series*, 1993 92: 221-231.

Weisberg, SB; Ranasinghe, JA; Dauer, DM; Schaffner, LC; Diaz, RJ; Frithsen, JB. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake bay. *Estuaries*, 1997 20: 149-158.

Willig, MR; Selcer, KW. Bat species density gradients in the New World: a statistical assessment. *Journal of Biogeography,* 1989 16: 189:195.

Wilson, JG. The role of bioindicators in estuarine management. *Estuaries*, 1994 17, 1A: 94-101.