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PREFACE

From a human history perspective, the intrinsic characteristics of estuaries have made them preferable sites of occupation and, consequently, intense areas of development. A direct consequence of human occupation of these coastal areas is that estuaries rank among the environments most affected by human presence and activities. The fast expansion of socio-economic activities on coastal and estuarine areas over the last decades, such as tourism, nature conservation, coastal fisheries and industrial and urban development has expanded and complicated the management tasks. In recent years, there has been a growing concern to maintain a steady growth in economical activities and social development in estuarine areas, while preserving their natural features and ecological services.

Given the acceptance by governments of the goal of sustainable development, a more sustainable coastal management strategy requires a more interdisciplinary and integrated management process. There are no easy answers to the question of what is best for a particular system from a resource's management point of view. It is the task of scientists from different disciplines to present as complete a picture as possible to those who make decisions. The ECOMANAGE (Integrated Ecological Coastal Zone Management System) project described here, funded by the *European Commission's* Sixth Framework Programme (Contract nº INCO-CT-2004-003715), aims to provide coastal authorities with the knowledge and tools for such an integrated management approach. The common goal was to work towards a social and environmental sustainable estuarine system management in three distinct transitional waters systems in South America: Santos Estuary in Brazil, Bahía Blanca Estuary in Argentina and Fjord Aysén in Chile. Besides their geographical location, these coastal systems cover a wide spectrum of management challenges because they vary significantly in their ecological state and human pressures, in a gradient that goes from a more pristine state of Fjord Aysén, to the heavily occupied and degraded system in Santos Estuary.

The South American continent is endowed with a unique and valuable marine heritage, which enclose several of the world's largest and most productive estuaries. The accelerated development in most Latin American countries is posing demanding challenges in the management of natural resources, especially in coastal areas. Integrated coastal management approaches are required, combining all aspects of the human, physical and biological aspects of the coastal zone within a single management framework. Integrated coastal management is presented here as a broad, multi-purpose endeavor aimed at improving the quality of life of communities dependent on estuarine resources and helping local decision maker attaining sustainable development of estuarine areas, from the headwaters of coastal watersheds to the outer marine areas. The work presented in this volume is a step in that direction. Hopefully, the knowledge, experience, tools and results presented here will be used in other places with similar conflicting uses of natural resources.

Interdisciplinary and integration was the major thrust of ECOMANAGE. The project was originally assembled from scientifically promising and socially relevant research fields, with physical modelling and eutrophication as the core. The social sciences, human ecology and management oriented subjects were included to provide the project with the integrative principle. The work developed during the project formed the knowledge pool for this book. The volume is a collection of writings selected on the basis of novelty, relevance in a water resource management framework, and insightfulness. Contributions have also been included in order to survey the strengths and limitations of a range of existing coastal zone management practices operating in different local environmental and socio-economic contexts. The core message that is highlighted is that the management challenges posed are complex and multifaceted, encompassing physical forcing, natural hazard and variability and vulnerability, together with socio-cultural vulnerability problems.

Being the result of a multidisciplinary scientific endeavor, the book will have an audience that range across a wide spectrum of environmental and social disciplines. The book should be of interest for anyone working in the field of ICZM (Integrated Coastal Zone Management), from scientists to decision makers. Dealing with examples from South America, the book has a strong local interest. However, the kind of approach developed in the project and portrayed in the book enables this work to be used as a benchmark for scientists working worldwide in related areas or facing the same challenges.

The book addresses costal zone management in an integrative way, with particular focus on water resources. As such, we hope it will be of interest for scientists working in fields such as aquatic ecology, ecohydrology, ground water, marine sciences in general, water quality, coastal zone management, etc. In addition, the strong component of the modelling approach will target the modelling community, from ecosystem to ground water modelers.

The Editors
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PART A

INTRODUCTION

BASIC CONCEPTS OF ESTUARINE ECOLOGY

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1 ESTUARINE SYSTEMS: THE LAND-OCEAN LINK

Estuaries are highly dynamic environments with their physical, chemical and biological structure characterized by high spatial and temporal variability. The temporal fluctuations and spatial gradients in these systems induce large variability in chemical and biological properties of the water and sediment. Estuaries are subject to continuous variations in wind, irradiance, rainfall, water level and freshwater runoff. Moreover, estuaries are very often heavily utilised and impacted by mankind, being used as (natural) harbours, for fish farming, recreation, as waste water recipient, etc. There are many ways to define what an estuary is. Probably the simplest definition is that an estuary is a partially enclosed coastal embayment where fresh water and sea water meet and mix. The estuary can have the simple morphology of a river entering the sea or a complex and lengthy one, like in fjords. Estuaries are among the most productive environments on earth and they are important ecotones, i.e., transition zones between different ecosystems. Ecotones are boundaries between resource patches in the landscape, regulating energy, nutrient and mineral sediment flow between adjacent patches (Naiman et al. 1995, Schiemer et al. 1995). Estuaries and their frequently associated fringes of tidal flats, salt marshes and mangrove forests are the transition zones between one environment and another - tidal flats, salt marshes and mangrove forests are the transition between land and sea, and estuaries the transition between fresh and sea water. Being the transitions between very different environments, all estuaries share significant physical, chemical and biological features. Thus, we can state that an estuary is a transition system governed by complex interacting elements which vary in space and time.

Usually, estuaries have more similarities with the marine than with the freshwater environment, but in all aquatic systems the throphodynamic structure and functions are very similar, with the exception of gelatinous plankton, which does not occur in freshwater systems. Nevertheless, in each and every aquatic system, the local mix of interactions between the abiotic and biotic environment results in different system behaviour and a different response to anthropogenic pressures, making it impossible to use simple rules of thumb to predict ecosystem responses to such pressures.

2 PHYSICAL AND CHEMICAL CHARACTERISTICS OF ESTUARIES

Estuaries and the adjacent coastal areas have a specific size, shape and bathymetry, a specific tidal influence, fresh water inflow, turbidity and residence times, sediment properties, carbon-to-nutrients ratios, water-column turbidity, etc. Together, all these characteristics, together with human-influenced environment make each estuary unique. Estuaries possess a unique combination of characteristics, frequently expressed in steep physical, chemical and biological gradients. Because it is where fresh and salt water meets, estuaries are influenced by processes affecting both these types of water, such as tides, coastal hydrodynamics, variations in the river, etc. These factors not only govern much of the physical and chemical characteristics of the estuaries but also its ecological dynamics. Extreme salinity and temperature fluctuations, muddy substrates, and other physical factors like light availability and residence time help make estuaries challenging ecosystems for aquatic organisms.

The shape of the estuary and its sediment, the wind, evaporation of water from the surface and river flow influences temperature and salinity in estuaries. The water temperature in estuaries varies markedly because of their shallow water and large surface area. Fjords are the exception because of their greater water depth. Water temperature affects the dynamics of a system because it regulates all biological rates. Therefore, a clear seasonality in biological activity is seen in estuaries at mid- to high latitudes. Generally, salinity decreases moving upstream but it fluctuates dramatically both from place to place and time to time. Salinity may also vary with depth in the estuary, as well as across the estuary due to the Coriolis effect.

2.1 Water circulation and stratification

Water circulation inside an estuary can change the conditions of the ecosystem over a much smaller temporal scale, when compared with neritic or oceanic areas. The hydrodynamics inside an estuary are driven by a complex interplay of mechanisms, all with a strong influence on biological processes. Water circulation is conditioned by tidal currents, river discharges, wind and local topography. The resulting circulation patterns may have a large effect on the abundance and production of the microbial community by controlling the supply of allochthonous organic matter, concentrating and retaining locally produced organic matter inside the system. They also produce conditions for long-term coupling of bacterial production and autochthons sources of organic matter. Because an estuary is not a closed system, tidal currents act as an oscillating conveyor belt with the coastal zone, moving plankton, organic and inorganic materials, and sediments back and forth, creating complex distribution patterns.

Estuaries can be classified according to their mean tidal range as microtidal (mean tidal range < 2m), mesotidal (mean tidal range between 2 and 4 m), macrotidal (mean tidal range between 4 and 6 m), and hypertidal (mean tidal range > 6m) (Dyer 1997). The difference in the tidal range confers distinct characteristics to the estuarine dynamics. As an example, macrotidal estuaries, which are characterized by high tidal energy, generally exhibit lower levels of chlorophyll a (Chla) than systems with lower tidal energy. They also exhibit a tolerance to high nutrient loadings from freshwater outflows (Monbet 1992). Estuaries are usually divided in two classes defined by their vertical density profile. When the currents of riverine fresh water inflow and tide are similar, turbulence is the major mixing agent. This process is induced by the periodicity of tidal action. In this case the vertical salinity profile is less variable because most of the energy dissipates in the vertical mixing, producing a rather complex set of layers and water masses. Under these conditions, estuaries are considered partially mixed or moderately stratified. In completely mixed and vertically homogeneous estuaries, however, the tidal action is strongly dominant and the water column is well mixed. Together with the shallowness in this type of estuaries, the balance between fresh and sea water, controlled daily by strong tidal currents and modulated seasonally by the river flow, contributes to the absence of a vertical stratification. Major salinity and temperature changes are more frequently observed horizontally rather than vertically and this spatial heterogeneity is thought to affect nearly every aspect of population dynamics, species interactions, and community structure.

2.2 Residence time

With the influence of tidal currents and river flow, the entire estuary experiences fluxes that interfere with the transport and expression of biological activities and with the distribution of biomass in the water column. The residence (or flushing) time of water depends strongly on tides, freshwater runoff and morphological size, especially length. Other processes can modify the residence time in an individual estuary, such as currents driven by a difference in density between fresh and salt water - this is particularly important in deep fjords, bays and semi-enclosed seas where the bottom waters can be nearly stagnant and where water quality can be degraded severely. Another key process is water storage and buffering by intertidal wetlands, mainly salt marsh or mangrove vegetation that flank the main estuary and results in drag to the flow and temporary storage of waters. The residence time reflects the rate at which dissolved and planktonic components in the water are flushed out to the sea. As such, it controls many of the elements that provide information on the health of the estuary. As such, the residence time of an estuary is an important parameter because it expresses its robustness and ability to cope with human-induced stress; Well-flushed estuaries are intrinsically more robust than poorly flushed systems. Environmental degradation is usually intensified during periods of reduced freshwater inflows, for example, during drought or when human activities in the catchment cause significant reductions in dissolved oxygen, for example through eutrophication. The residence time is usually more critical in areas where contaminant accumulation and increased turbidity from human influences are most likely to occur. This is usually the case in the upper reaches of the estuary and in confined areas in estuaries with intricate morphologies.

2.3 Nutrient availability

Nutrients can be present in two major forms: inorganic (or mineral) and organic (both living and detrital). Nitrogen and phosphorus are the most significant nutrients, and their main species include dissolved (nitrate, nitrite, ammonium, organic N, phosphate, organic P) and particulate (organic N, organic P) components. Particulate species tend to be dominant in the river loads reaching the estuaries, but nitrate and phosphate become more important in populous regions. The dynamics of nutrients depend on a number of physical, biological and chemical processes and their fate after entering the estuary varies as a function of turbidity, water flow and biota. Physical processes include mixing, flushing and sedimentation. Chemical processes include absorption and desorption. Biological processes include fixation of dissolved and particulate nutrients, primarily by bacteria and phytoplankton, and release of inorganic nutrients through mineralization, mostly by bacterioplankton (decomposers). Biological and chemical transformation processes increase in importance with increasing residence time because, when the residence time is large, there is more time for these processes to occur.

Systems with very long residence times can export much less of these nutrients to the coastal zone than systems with very short residence times. When the residence time is high the nutrients can end up being consumed while still in the estuary or lost by chemical processes like denitrification (i.e., loss of nitrogen to the atmosphere). There are no general rules to predict what nutrients limit estuarine primary producers, and when. Instead, the norm is that estuarine seasonal cycles depend on the temporal occurrence of deliveries of nutrients, the relative magnitudes of the sources of nutrients, and the biological demands. Each estuary may have its own combination of these three types of conditions, resulting in a seasonal cycle with reasonably well-understood control mechanisms (Valiela 1995). In some estuaries tidal mixing may be the major mechanism providing nutrients. In fjords for example, nutrients are regenerated by the benthos and the advection induced by tidal movements, together with turbulence, supplies phytoplankton with nutrients. River and estuarine waters are often enriched with phosphate from urban and industrial wastewater and from land runoff and they receive silicate from tributary river inflows via rock weathering and soil leaching. In pristine environments, the transport of nutrients from the drainage basin to its watercourses is dependent on the chemical and mechanical weathering of soil minerals, whereas in cultivated environments agriculture is considered to be the largest contributor to river nutrient loads (Tappin 2002).

2.4 Oxygen concentrations

In estuaries, salt marshes and mangrove forests, oxygen concentration is highly variable and often reaches extreme levels. Dissolved oxygen is an important chemical variable because of the metabolic requirement of aerobic organisms. Decomposition of the large quantities of organic matter produced in these environments or introduced as sewage or waste inputs may deplete dissolved oxygen to hypoxic and anoxic levels. Nitrogenous compounds may create a significant extra oxygen demand in estuaries through microbially mediated nitrogen transformations. Ammonia utilizes dissolved oxygen during nitrification to produce nitrate, via nitrite. Ammonia is usually a by-product of most biological processes and additional ammonia is input to estuaries via tributary rivers and wastewater discharges.

At the same time, high rates of photosynthesis may increase dissolved oxygen concentration to super-saturated levels. High nutrient inputs to estuaries and the associated eutrophication can lead to algal blooms and this, in turn, can result in the consumption of dissolved oxygen by decaying algae once the nutrients become depleted. Dead phytoplankton is further decomposed by bacteria, thus enhancing the oxygen demand. The residence time plays a major role in this process because it determines whether excessive nutrient inputs are likely to lead to algal blooms and oxygen sags. Low dissolved oxygen levels in estuarine waters are generally attributed to direct effluent discharges, sewage treatment plants and industrial pollution.

However, high oxygen demand and anoxia can also be associated with natural processes, especially with the increase of organic material in the estuary turbidity maximum. Considering the influence of varying residence times and other environmental factors, such as water temperature and wind intensity, there is usually no simple relationship between the oxygen demand of waste effluents and reductions in oxygen concentrations. The oxygen deficit depends on water flow, turbidity, and oxygen supply and demand, and this varies among and within estuaries (Owens et al. 1997).

2.5 Underwater light climate

Many estuaries are relatively shallow and one would expect an optimum underwater light climate for primary production, both in the water column and on the sediment. However, high concentrations of suspended sediment are common, which greatly reduces water clarity. This permits very little light to penetrate through the water column. The resuspension of fine sediments induced by tidal currents determines the underwater light climate. Tidally driven resuspension, and riverine sources of sediments influence suspended matter concentration, determining the photic depth in the water column. Mean annual chlorophyll a levels are significantly lower in strongly tidal than in weakly tidal estuaries with similar nutrient levels (Monbet 1992). Larger and more energetic tides ensure that accumulated sediment is systematically suspended, leading to high turbidity and low light levels with less potential for bloom conditions, regardless of nutrient levels. The result is that in many estuarine systems, light is a key limiting factor for pelagic primary production (Cloern 1999, 2001). Estuaries with marked tides generally exhibit a tolerance to eutrophication, being insensitive to some degree to the nutrient loading in their inflowing rivers.

3 TYPES OF ESTUARINE COMMUNITIES

Estuarine ecosystems include several distinct communities, each with their own characteristic assemblage of plants and animals. Some of these communities are permanent parts of the system, while others like plankton and nekton come in and leave with the tide. To better understand the role of the different estuarine communities, it is important to have a closer look at the main compartments of these systems.

3.1 Water column or pelagic communities

Typical features of oceanic pelagic systems are the dominance of locally-produced (autochthonous) organic material and the oligotrophic conditions with characteristically small phytoplankton cells. A rather different situation is found in estuarine ecosystems, where a high content of allochthonous material is present, as well as high levels of nutrients (indicating mesotrophic, eutrophic and even hypertrophic conditions), larger phytoplankton cells like centric diatoms, and intense bacterial activity. The type and density of plankton inhabiting estuaries varies immensely with the currents, salinity, and temperature. Most of the phytoplankton and zooplankton in small estuaries are marine species flushed in and out by the tides, while larger estuaries with longer residence times may also have their own, strictly estuarine species. The major distinction between estuaries and lakes, apart from salinity, is the tidal energy in estuaries, which ranges from small (the Baltic) to enormous (Bay of Fundy, Nova Scotia) in direct proportion to the local tidal range. The tide generates tidal currents, which in turn generate turbulent mixing, which leads to resuspension of sediments and hence to turbid water where the sunlight cannot penetrate very deeply, which reduces the thickness of the euphotic zone, strongly reducing the growth potential for phytoplankton. In a more general way the controlling mechanisms on the production of estuarine and coastal systems are usually summarized in five major conditions: ambient light, nutrient availability, temperature, grazing, and transport.

River inflow, reflecting climate variability, affects biomass through fluctuations in flushing, but also induces changes in the growth rates through fluctuations in total suspended solids. In well mixed estuaries, phytoplankton populations may have to adapt to continuously changing irradiance conditions ranging from complete darkness to saturating light. The result is that there may be several regulatory mechanisms acting at the same time or with particular spatial/temporal relevance. The specific mechanisms and timing by which light, nutrients, grazing and predation interact may differ, but the major variables are near-universal. Although estuaries may appear very distinctive environments at first sight, the seasonal cycle is determined by the same limiting factors that are prominent elsewhere in the sea, but modified to an extent by the seasonal input of fresh water (Valiela 1995).

The formation of blooms in the estuary is controlled by local conditions and transport-related mechanisms that govern biomass distributions (Lucas et al. 1999a, Lucas et al. 1999b). Local phytoplankton population growth rates may vary significantly in the horizontal due to variations in water column height, as well as differences in turbidity, nutrient availability, grazing pressure, and time scales for vertical transport through the water column. Biomass abundance at any particular place and time is a function of: (1) spatial variability of population dynamics, and (2) spatially variable transport of water. Several processes will determine if local high phytoplankton growth rates are the same as the bloom formation areas (biomass accumulation). The first control can be defined by the local combinations of both biotic and abiotic parameters responsible for the balance between production and loss (turbidity, nutrients, grazing pressure, etc.). Therefore, local conditions control net population growth at a particular location. The second major control - transport - determines biomass concentration and distribution, thus controlling if and where a bloom actually occurs (favorable conditions for patchiness vs. dispersion of mass through the domain, etc.). The transport inside the estuary determines the residence time of the water in different parts of the system, determining whether phytoplankton remain for the time necessary to generate a bloom, but also conditioning the exchanges between sediment and water column. Also with respect to bloom development, well-mixed shallow subtidal areas are much more dynamic environments than deep channel regions, exhibiting a broader range of effective growth rates over tidal time scales and potentially acting as a significant source and sink for phytoplankton biomass (Lucas et al. 1999a).

The relation between freshwater flow and accumulation of phytoplankton biomass in estuaries is complex. In estuaries where the processes of material transport are mostly tidally driven, tidal variability dominates seasonal effects. River discharges are particularly relevant in winter months characterized by high flow values. While high freshwater inputs can stimulate primary production by importing nutrients into the system, the development of blooms is only possible when the net rate of biomass accumulation exceeds the losses (either by biotic or abiotic means). Therefore, also low river inputs causing longer residence times may allow the accumulation of phytoplankton and may trigger a bloom.

3.2 Benthic communities in tidal flats

Sediment areas exposed at low tide are called tidal flats and, if they have a clay content of more than 10%, mud flats. Mud flats are particularly extensive in estuaries with a large tidal range. Estuaries with a high tidal range usually have large tidal flat areas, and sizable natural microphytobenthic communities, which play an important role in carbon fixation and nutrient removal in shallow waters (Gao and Mckinley 1994, Simas et al. 2001). Benthic primary productivity in shallow waters is strongly dependent on the regulation of underwater light climate by suspended particulate matter (Schild and Prochnow 2001). If an excess of nutrients exists, light availability will be the key limitation. In intertidal areas, the combination of shallow waters and strong tidal currents creates a complex pattern of SPM transport, deposition, and resuspension dynamics. Sub-tidal benthic primary production will probably be low due to natural turbidity.

In temperate climates the mudflats are often fringed by salt marshes that are inundated at spring tides or, in the tropics, by mangroves. These vegetated mudflats play a critical role in determining the robustness of the estuary, by trapping fine sediments, sequestering nutrients and pollutants, influencing the water residence time, and converting nutrients in the water column into plant biomass. Mudflats are home to a wide range of organisms that tolerate the changing conditions induced by the tidal movements. Almost always large numbers of benthic diatoms grow on the mud and frequently produce extensive blooms. Bacteria are also extremely abundant in the tidal flats where they decompose the organic matter brought in by rivers and tides.

3.3 Salt marshes

Salt marshes are buffer areas that link land and sea. Salt marshes generally start at the level of the average neap tide and extend upward to and beyond the height of the highest tides. They are one of the few examples of a community of higher plants that can tolerate saltwater and survive in the marine environment. In total there are about 500 species of plants belonging to 18 families of angiosperms found in salt marshes worldwide (e.g. *Spartina*). Many species are perennial grasses. The salt marshes are dominated by grasses such as *Spartina* spp., and by rushes, *Juncus* spp. The duration of exposure and inundation during the tidal cycle determines the species zonation. Even though salt marsh plants tolerate full strength seawater, they grow faster in low salinities because salts of seawater are an osmotic stress, with a metabolic cost imposed on plants.

Salt marshes occur in the alluvial plains associated with an estuary and generally include channels, called tidal creeks that fill and empty with the motion of the tides. These meandering creeks usually form an intricate network of drainage channels across a salt marsh. Besides having drainage creeks, salt marshes also have mud flat areas (called pans) and tidal flats. Salt pans are circular to elliptical depressions, which are flooded at high tide and remain filled with salt water at low tide. Salt marshes stabilize the sediments, thus promoting their own growth. The roots and stems tend to capture the suspended sediments carried by currents and waves. Few animals feed on the salt marsh directly, most of the energy captured by the marsh in photosynthesis being slowly released to the adjacent water and sediments as the vegetation decays. Terrestrial animals including insects, birds and mammals, account for about 50% of the fauna found in salt marshes. Marine animals, mostly invertebrates, include bivalves, gastropod snails and crabs. The salt marsh is a detrital system where grazing herbivores play a minor role. Most of the detrital material from *Spartina* of the low marsh is washed out by the tide; that of the high marsh is decomposed in place by bacteria.

3.3.1 Nutrient dynamics

The growth and development of salt marsh communities is influenced by the concentrations of nutrients and these, in turn, by groundwater flows. As such, salt marsh nutrient fluxes can be affected by the hydrological conditions, particularly the magnitude and status of groundwater flows (Sutula et al. 2001). The concentration of nutrients in salt marsh creeks depends on the balance between the supply (from inside and outside) and the rate of uptake by the growth of salt marsh vegetation. Where adequate levels of both phosphorus and nitrogen occur, other elements, such as silicon, can become limiting (Jacobsen et al. 1995). The importance of salt marshes as a nutrient source and sink for the estuary is an open question. The amount of nitrogen cycled depends on tidal input, physical and chemical exchanges with air and water, and biological fluxes. Salt marshes are characterized by their large nutrient storage capacities that under certain circumstances can become ’leaky’ with subsequent nutrient releases (Turner 1993). The release of nitrogen and phosphorus from the salt marsh occurs generally during the process of the decomposition of organic matter, but direct losses by the leaching of nitrogen, phosphorus and also carbon, from live plant tissues can also take place. The flux of released nutrients can be high enough to account for significant increases in the activity of the estuarine phytoplankton community and, consequently, of potential significance for many other estuarine communities.

3.3.2 Fluxes of organic matter

Primary production and decomposition rates are high within the salt marsh, usually comparable to those of tropical rain forests. The behaviour of DOM and POM is essentially similar to suspended sediment and is based on the flux of the tidal water flow. The fate of excess carbon production within these systems is not well understood. Some salt marshes are dependent on tidal exchanges and import more than they export, whereas others export more than they import. Excess production may end up in the sediments, being transformed by microbes in water in the marsh and tidal creeks, or exported to the estuary physically as detritus, as bacteria, or as fish, crabs, and intertidal organisms in the food web. The possibilities for the export of organic matter to adjoining marine ecosystems have also been widely recognized. The basic model of salt-marsh estuaries as exporting systems usually referred to as the “Outwelling Hypothesis” (Dame et al. 1986), developed from the notion that marsh productivity may be “outwelled ”as organisms rather than as organic matter and nutrients (Odum 2000).

3.4 Mangroves

Mangrove forests are important coastal ecosystems that provide a variety of ecological and societal services. At tropical and subtropical latitudes the herb-dominated salt marsh is replaced by mangrove forests. These are concentrated along low lying coasts with sandy shores and in estuaries. They stand as a transition between two environments (land and sea), usually in estuaries where they act as an interface between river and sea. Mangroves are associated with the terrestrial climates of the tropical rain forest, tropical dry forest, savanna and desert, due mainly to the sensitivity of mangrove to frost.

Mangroves are forests of trees and shrubs that are rooted in soft sediment in the upper intertidal zone where wave action is absent, sediments accumulate and the mud is anoxic. They extend landward to the spring tide high water line, where they are only rarely flooded. The term ”mangrove” refers to a variety of trees and shrubs belonging to some 12 genera and up to 80 species of flowering terrestrial plants (angiosperms) found world-wide. Mangrove trees of different species are usually distributed relative to elevation within the intertidal zone. The most frequent are the Genus *Rhizophora* near the water (intertidal zone, inundation by average high tides), *Avicennia* (flooded by average spring tides) and Laguncularia (only reached by the highest tides). One of the most widely distributed is the red mangrove *Rhizophora*. The dominant genera (*Rhizophora*, *Avicennia*) share some common features: they are salt-tolerant and ecologically restricted to tidal swamps, and possess both aerial and shallow roots that interlink and spread widely over muddy substrate.

These forests are a unique marine system, having aerial storage of plant biomass, harboring both marine and terrestrial species. The forest comprises euryhaline plants, tolerant to a wide range of salinities, found in fully saline waters and well up into estuaries. Immersion of roots in seawater up to 1m in depth is common. The roots of mangroves are morphologically specialized for anchoring and nutrient transport. Mangroves and salt marshes have many similarities in physical and biological processes. These include their role in trapping sediment and pollution, converting nutrients to plant biomass and serving as a habitat for numerous organisms like fish and crustaceans.

Mangrove trees have special physiological adaptations that exclude salt from entering their tissue, or that allow the excretion of salt in excess. Many species are viviparous, producing seeds that germinate on the tree. Mangroves harbor a rich fauna where birds, monkeys, snakes, frogs and insects are common inhabitants. Barnacles, snails, fiddler crabs and land crabs are also found around mangroves.

3.4.1 Mangrove forest components and abiotic conditions

Ecologically, a mangrove community can be divided into: (1) Above-water forest. A study of Florida mangroves showed that about 5% of the total leaf production was consumed by terrestrial grazers, the rest entering the aquatic system as debris and becoming available for marine detritivores, either fish or invertebrates (Twilley et al. 1986); (2) Intertidal swamp. Leaf litter is a major source of nutrients and energy in the mangrove swamp, and many residents are detritivores; (3) Submerged subtidal habitat. High organic content in the fine-grained mud; burrowing animals (crabs, shrimps, worms, etc.) are common, and their burrows facilitate oxygen penetration into the mud and thus ameliorate anoxic conditions.

Mangrove systems occupy the full tidal range, and as a consequence, the organisms in these environments are exposed to highly variable light conditions, ranging from full sunlight at low tide to very little light at high tide. Penetration of light and water movement varies over short distances and in the course of a day. This physical variability is reflected in highly variable chemical conditions. Complex tidal currents flow in mangrove forests, where they are involved in ecological processes. In addition, these currents also fragment and transport the litter produced by mangrove vegetation. Temperature is also highly variable: because it is a shallow water system (particularly at low tide), water temperature varies with air temperature, seawater and river water temperature may be different inside the estuary, so temperature may change with each tidal cycle (shallow areas in these environments can heat up to 40 ℃). Oxygen concentration is highly variable and often reaches extreme levels. While decomposition of large quantities of organic matter can deplete dissolved oxygen, high rates of photosynthesis can increase its concentrations to super-saturated levels.

3.4.2 Production

Mangrove ecosystems rank amongst the most productive communities in the world, with their net primary production estimated at $1.1 \times 10^{15}\,\mathrm{g\,yr^{-1}}$ worldwide (Duarte and Cebrián 1996). Most of the plant material is not eaten directly, but decays and enriches the adjacent waters through detritus food chains. Mangrove forests export a considerable portion of their production to the surrounding waters, largely as leaf fall and other detrital material. Concentration of dissolved inorganic P in mangroves is generally low. A close microbe-nutrient plant connection may serve as a path to conserve scarce nutrients necessary for the existence of these forests (Alongi et al. 1993). Numerous studies have shown that the influence of mangrove forests on the adjacent lagoonal and near-coastal ecosystems is variable in terms of matter transfer balances. Whether mangroves act as a source or sink of organic matter depends on factors such as topography, forest types, and tidal regime.

Tidal inundation generates a nutrient exchange between sediment and estuarine waters. Water exchange transports nutrients into mangrove areas, and exports organic material out. But mangroves are rich in recycled nutrients because the roots trap detritus which are mineralised in the sediments. The recycled nutrients then become available for uptake by the roots of the mangroves. As such, the mangroves are not solely dependent on dissolved nutrients in the surrounding (oligotrophic) seawater. Other typical features of mangrove sediments are relatively low concentrations of dissolved inorganic nutrients, for example, nitrate, ammonium and phosphate in porewater, and the presence of tannins derived from leaching and decomposing roots and litter. Ammonium is the main form of inorganic N in mangrove sediments because nitrification is prevented due to the lack of oxygen in the sediment.

3.4.3 Interaction with sediments

Mangrove forests tend to accumulate sediment by creating conditions for the fine particles trapped in the root system to become permanently deposited. This sediment trapping capacity of mangroves is essential for the ecosystem. Mangroves form protective barriers against wind damage and erosion in regions that are subjected to severe tropical storms. In some areas they may facilitate the conversion of intertidal regions into semi-terrestrial habitats by trapping and accumulating sediment. The intertwined roots further reduce water velocities, trapping suspended sediments and organic material (particularly leaves).

3.5 Sea Grass Meadows

Other communities that thrive in the shallow and well lighted areas in some estuaries, coastal lagoons and coastal areas are the sea grass meadows. These are among the few higher plants that are totally adapted to the marine environment, with about 50 species that can live totally submerged in seawater. In temperate waters, the most common genus is *Zostera* (eelgrass), while in tropical waters it is *Thalassia* (turtle grass). These plants absorb nutrients directly from the water across the leaf surface and from the sediment by their roots. Sea grass meadows are rich biological communities with high rates of primary production. Few animals eat sea grass directly: manatee, green turtles, parrot fish and surgeon fish are the principal vertebrate herbivores in the tropics. Sea urchins are the only invertebrates feeding on these plants. Sea grass meadows serve as host to epiphytes including micro- and macroalgae such as benthic diatoms and filamentous red algae. Between 25-30% of total photosynthesis may be due to epiphytic algae. Invertebrate species feeding within the sea grass meadows on epiphytic algae include gastropods, nudibranchs, isopods, amphipods and shrimp. A considerable fraction of the leaves are sloughed off and may float considerable distances, breaking down, sinking and becoming part of the sediment, eventually entering the detritus food chain. During this breakdown process, the leaves become floating bacterial cultures. These in turn are used as a food source by filter and deposit feeders. Sea grass meadows stabilize the sediments in which they grow because the leaves deflect and reduce the water movements from waves and currents. Suspended material tends to settle in the quiet waters in the meadow and is bound by the network of rhizomes and roots.

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THE CONTINUOUS CHALLENGE OF MANAGING ESTUARINE ECOSYSTEMS

M. MATEUS, J.W. BARETTA AND R. NEVES

1 THE SIGNIFICANCE OF ESTUARIES IN HUMAN AFFAIRS

In the course of human history, the coastal plains and river valleys have usually been the most populated areas over the world. Proximity to water bodies has been an incentive for the location of human settlements for millennia. Presently, about 60% of the world's population lives along the estuaries and the coast (Lindeboom 2002). Located at the interface between land and the sea, estuaries are sites of significant biotic diversity and human development. Estuaries provide many goods and services including coastal protection, tourism, water purification, breeding and nursing grounds for commercial fish species, etc. The biological productivity sustaining a high level of food production in these areas has been a major attraction for human settlement, as well as the use of the rivers and estuaries as transport routes, fundamental for economic and social development.

From a human history perspective the function of estuaries as natural harbors and provider of abundant natural resources made them the location of some of the world's greatest cities. A direct consequence of human occupation of these coastal areas is that estuaries rank among the environments most impacted by human activities. In many cases the consequence of human intrusion has been disastrous. Human actions also have resulted in worldwide manipulation of the hydrological, chemical, and biological factors that regulate estuaries ecological dynamics (Strayer et al. 1999, Council 2000, Cloern 2001).

Human modification of marine environments, especially coastlines, estuaries and wetlands has gone hand in hand with social and economic development. As such, any analysis of the water resources and their conflictive management policies of these areas must be based on awareness of environmental and economical fundamentals (Allan 2005). Management efforts of marine living resources are increasingly shifting towards ecosystem-based management (Pikitch et al. 2004), and estuaries are no exception. Ecosystems are complex adaptive systems that require a flexible governance with the capacity to respond to environmental feedback (Levin 1998, Dietz et al. 2003). There is a need to deal with scientific insights, economic and social factors in making natural resource management decisions. These decisions, in turn, have ecological, economic, social and political ramifications. The inevitable result is that it becomes difficult sometimes to isolate the key elements that affect decisions about environmental impacts and the management of such resources. In addition, changing human values and social priorities also form part of the context for resource management. This facet of societal change, together with environmental stochasticity, makes management a dynamic endeavor.

2 CONFLICTING INTERESTS

The oceans cover 75% of the earth's surface, accounting for 90% of the planet's water supply. Marine ecosystems provide a wide variety of goods and services, including vital food resources for millions of people (Holmlund and Hammer 1999). Man's utilization of coastal areas sea can basically be reduced to three aspects: (1) exploitation of marine organisms for food and other purposes, (2) use of the sea as a dumping ground, and (3) land reclamation. Together with environmental functions or services provided by estuaries, such as food production, mineralization of organic wastes, and aesthetic value, there are other services and amenities that are crucial for human activity like transport function, recreational activities, tourism, etc (Figure 1). This explains why estuaries, when compared to other marine areas, have the highest mean financial value per hectare per year (Figure 1). However, the rapid degradation of estuarine systems reveals the conflicting nature of human interest in these coastal areas. Maintenance or expansion of a regional economy is a major, usually even the primary, objective. While exploiting its resources, human activities also contribute to the destruction of other resources. Sometimes this apparent paradox denotes unsustainable practices and management shortcomings. It also can be the symptom of conflicting interests between development and conservation. But a conflict of interests can also arise in the conflict between human needs. As an example, the changes in river flows due to irrigation, damming and water diversion can modify the entire food web up to the level of fisheries, with significant negative consequences (Wolanski et al. 2006). Aquaculture is also an example of an enterprise with social and economical impact, but at the same time with the potential to degrade the environment.

A large and still increasing proportion of the human population lives close to the coast; thus the loss of services such as flood control and waste detoxification can have disastrous consequences (Adger et al. 2005). Changes in marine biodiversity are directly caused by overexploitation, pollution, and habitat destruction, or indirectly through climate change and related perturbations of ocean biogeochemistry. Among several irremediable problems, regional ecosystems such as estuaries (Lotze et al. 2006), and coastal communities (Jackson et al. 2001) are rapidly losing populations, species, or entire functional groups. Human-dominated marine ecosystems are experiencing accelerating loss of populations and species, with largely unknown consequences. Marine biodiversity loss is increasingly impairing the ocean's capacity to provide food, maintain water quality, and recover from perturbations (Worm et al. 2006). Ultimately, since flows from natural systems are limited, a conflict between human objectives and conservation of resources is inevitable, unless the rate at which humans extract resources from the marine environment are also limited (Ludwig 1993, Ludwig et al. 1993).

3 HUMAN INFLUENCE: ESTUARINE DEGRADATION THROUGH TRANSFORMATION

Coastal ecosystems have suffered multiple pressures, sometimes undergoing degradation in small, incremental steps that are difficult to recognize, while other in fast and huge steps. The increase of human populations in the river basins, by natural growth and internal migration, has resulted in a doubling of the population along many coasts over the last 20 years. Human activity affects ecosystem structure and functions through the disruption of the pattern and rate of matter as well as energy flow through ecosystems (Ohl et al. 2007). Transport and transformation and removal of resources driven by societal and economic pressures change the landscape and influence biodiversity, redefine the ecological state of ecosystems and the rate of delivering goods and services.

General analyses and reviews over the past two decades have identified a range of pressures that cause undesirable change in coastal ecosystems (Burke et al. 2001, Redman et al. 2004). Physical alteration, habitat degradation and destruction, pollution, water withdrawal, overexploitation, and the introduction of non-native species are the leading causes of ecosystem degradation. When compared with other marine systems, estuaries are under more intense degradation (Figure 2). Direct human impacts on estuaries have different origins, ranging from the consequences of engineering works (e.g., harbor construction, land claim, etc.) on the estuarine water residence time and change in sediment patterns, to the effects of wastewater discharges on the health of the biota. Liquid discharges and solid dumping with anthropogenic origins include an array of chemical contaminants, such as metals, organometals, petroleum hydrocarbons, organic compounds from pesticides, industrial wastes products and nutrients.

3.1 Modification and destruction of habitats

Most of the human activities related with the occupation and development of estuarine areas causes problems by habitat modification or destruction. Estuaries are dredged or filled and transformed into marinas, seaports, industrial parks, cities, and garbage dumps. Historically, most reclamation has been mainly related with flood protection and the production of agriculturally valuable land. A more recent trend is the reclamation to create land for residential housing, industry (petrochemical and oil refineries installations, etc.), and port, dock and airport facilities. Numerous estuaries have been transformed and even completely destructed through landfill to build urban or industrial infrastructures. A warning example comes from the United States where about one third of the estuaries have disappeared altogether.

Dredging and maintenance dredging of navigation channels may have several impacts on the marine environments. Estuaries are particularly sensitive areas, and any artificial deepening may result in permanent modifications in the ecological functioning of the system, even with permanent loss of environmental services. As an example, the land-building function of mangrove vegetation has very important implications in coastal management because it works as a natural barrier to protect adjacent land by enhancing sedimentation and reducing erosion by wave action, tides, and river flow. This is important for shallow estuaries that are prone to flooding, especially where the land is below sea level.

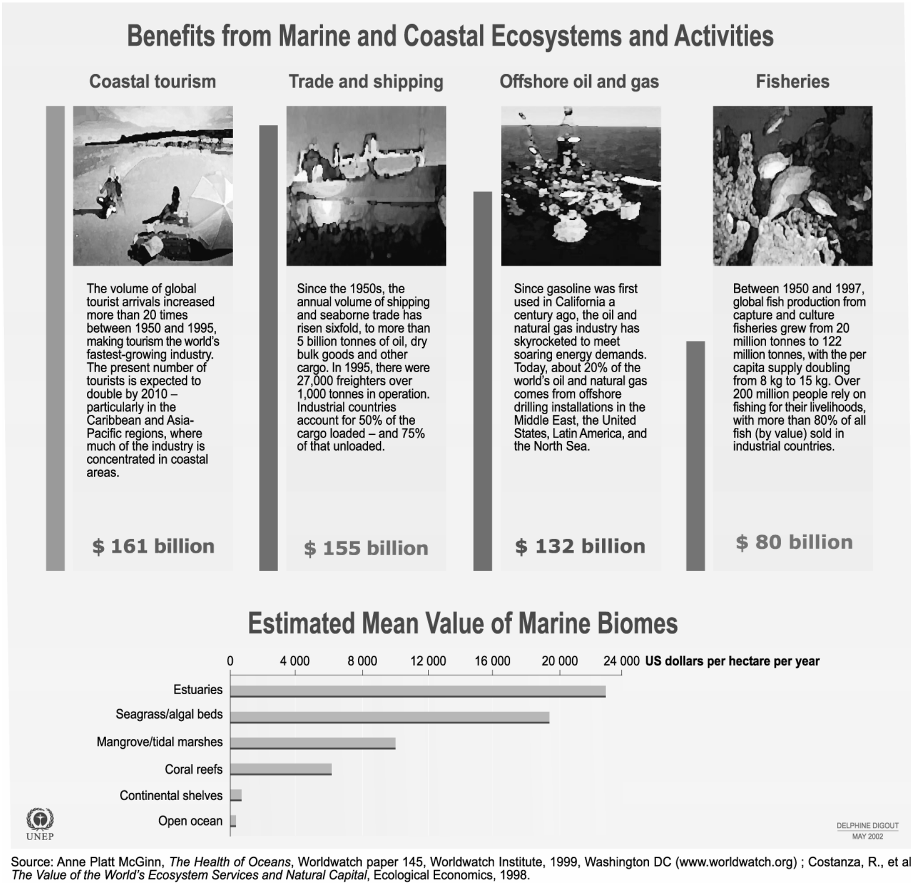


FIGURE 1: *Several activities generate significant revenues in coastal and marine areas. This graphic illustrates the economic benefits of coastal tourism, trade and shipping, offshore oil and gas, and fisheries. It also illustrates an estimation of the financial value of selected marine biomes (UNEP/GRID-Arendal 2002a).*

3.2 Port activities

Port activities are among the main driving forces in many estuaries around the world. They account for a number of known environmental pressures and have been responsible for changes in the state of the systems. There has been a continuous process of change in international transport management over the last 10 years, from a segmented modal approach towards a much more integrated transport concept tailored to better meet the pressing needs of customer industries. This, in turn, is resulting in an increasing pressure on ports to adapt their role and function to this more demanding operational environment (Juhel 2001). The interaction between the port and the city often surrounding it, in terms of transport network requirements, environmental protection, and overall safety, is a prerequisite for effective delivery of integrated logistics services. This means that the expansion of port activities, either by physical expansion or intensification of its use, or both, is always made in conjunction with the development of other supporting infrastructure. The overall consideration is that port development does not necessarily means new and better equipment; it means new and/or wider roads, larger areas for container storage and transport, traffic intensification, both in water and on land, etc.

A major problem associated with port activity in estuaries (which are shallow by nature) is the amount of dredging, both to expand the port or simply to maintain and improve navigability, as most estuarine channels require almost continuous dredging to keep them at the required depth. The disposal of thousands, if not millions of tons of sediment is a common problem which results directly from the maintenance-dredging activity. This is particularly problematic when sediments are dredged from polluted areas. Heavy metals and other pollutants that are adsorbed to the sediment and buried can still be remobilized, by a number of processes such as bioturbation, natural erosion and dredging. The disposal of dredged sediments requires an integrated management approach because they can contain potentially hazardous substances. If a system is degraded because of multiple anthropogenic pressures, as many estuaries in developing countries,as well as most estuaries in the developed world, effective management programs must be implemented to achieve sustainability. Consequently, the development of a Coastal Management Program as a management tool for port authorities should be a priority in order to prevent or minimize negative impacts of dredging activities.

3.3 Eutrophication

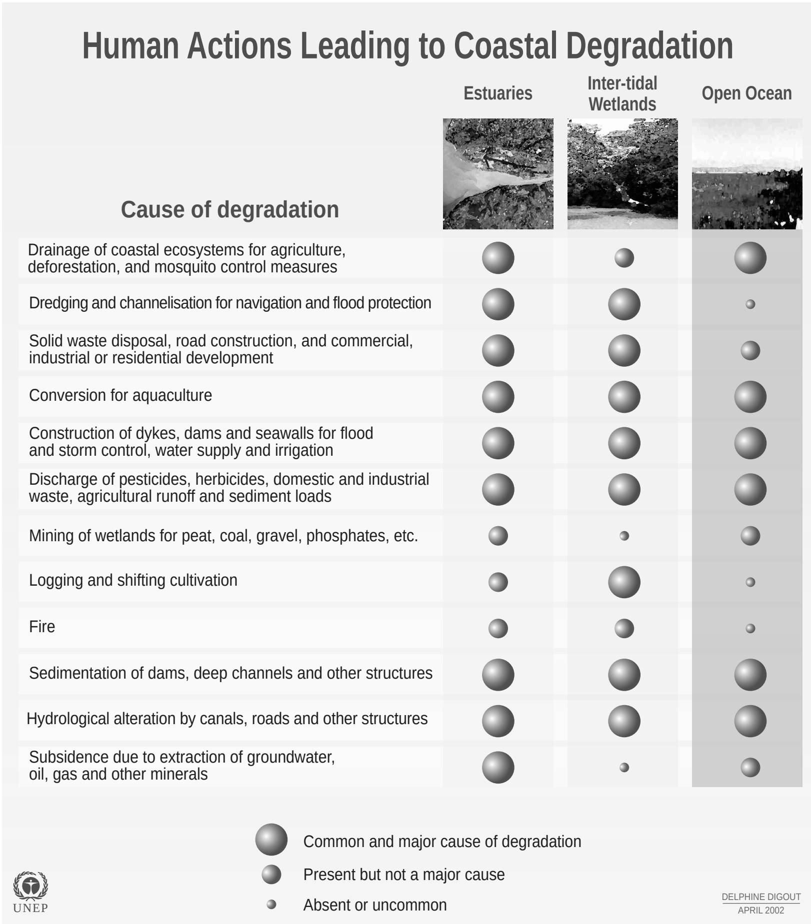
The increase in human occupation of both the estuarine and watershed is seen as the major cause of nutrient enrichment of estuarine and coastal areas. Fluxes of mineral nutrients, such as phosphate and nitrate, into the sea have world-wide more than doubled in the last decades (Meybeck 1998). The industrial and urban expansion in most developed countries has been pushing the natural state of estuarine systems to a state in which there is an artificial acceleration of nutrient-enrichment processes. Nutrient loads in many rivers have increased markedly over the last decades and this increase is thought to be at least partly responsible for the changed eutrophic status of a number of estuaries and coastal seas. This has become a significant problem in many estuaries and coastal zones, manifesting itself in symptoms such as high levels of chlorophyll a, excessive occurrence of macroalgae and epiphyte blooms, occurrence of anoxia and hypoxia, and harmful and toxic algal blooms (Bricker et al. 2003). In some ecosystems, the amount of extra nutrients is small enough that it may generate an increase of biological productivity without dramatic modification of biodiversity (Zalewski 2002). More commonly, however, the load of nutrients is so high that it degrades water quality, compromising ecological services, biodiversity, and productivity.

Estuarine ecosystems usually have a high content of allochthonous material and high concentrations of nutrients (comprising mesotrophic and eutrophic conditions), supporting high rates of phytoplankton and bacterial production. The increase of both organic material and nutrients in the system above background levels, as a result of eutrophication, poses serious threats. Physical characteristics of the estuary such as turbidity and the residence time control availability of nutrients and light in the system (Monbet 1992). To a certain extent, the residence time of an estuary determines the risk of degradation or eutrophication in the adjacent coastal areas. This depends on whether estuaries and coastal wetlands have sufficient time to deplete the nutrient reservoir, or whether nutrients make their way to the shelf without significant loss.

Cultural eutrophication reflects the enrichment of catchments areas like estuaries induced by human activities with nutrients like N and P, but not with silica. For some time now this unbalanced nutrient enrichment has been hypothesized as the cause of the shift from diatom dominance to non diatom dominance in the phytoplankton composition (Officer and Ryther 1980). Eutrophication conditions, with an increase in nitrogen and phosphorus and not in silica, forces a change in N:Si and P:Si ratios that is favorable to flagellate blooms and unfavorable to diatoms. The transition from diatom-based to non-diatom based phytoplankton communities has been associated with a degradation of the water quality (Turner and Rabalais 1994). Because Harmful Algal Blooms (HAB) occurrences are frequent near shore, local nutrient inputs are usually thought to be a causative factor. Nutrient enrichment of coastal areas may have other far-reaching consequences, such as loss or degradation of sea grass beds, fish-kills, interdiction of shellfish and other types of aquaculture and smothering of bivalves and other benthic organisms. Irrespective of the type and magnitude of the impacts, the modifications in the system induced by nutrient enrichment usually have significant economic and social costs, some of which may be readily identified (e.g. direct costs such as productivity losses), whilst others (e.g. indirect and non-use values) are more difficult to determine and tend to be ignored (Turner et al. 1999).

3.4 Oxygen depletion

Oxygen depletion is among the most serious threats that coastal systems such as estuaries can face (NRC 2000), and environmental degradation problems associated with the occurrence of low oxygen are increasing on a global scale. There is no other environmental variable of such ecological importance to coastal marine ecosystems that has changed so drastically due to human influences in recent decades (Diaz and Rosenberg 1995). Anoxia is most often associated with inputs of sewage and other organic materials. In estuaries and coastal areas near major population centers, the low dissolved oxygen levels are usually attributed to industrial and direct effluent discharges, especially from sewage treatment plants. The discharge of organic waste depletes oxygen directly as it decomposes, and the addition of nutrients can lead to oxygen depletion by stimulating primary production. There is a link between eutrophication and problematic oxygen levels because the boost in primary production promoted by nutrient enrichment leads to an increase in organic matter to be degraded by bacteria later on. The anthropogenic nutrient loading has increased the frequency and severity of hypoxia in estuaries and semi-enclosed seas (Rabalais and Turner 2001).



Source: United Nations Environment Programme (UNEP).

FIGURE 2: *Physical alteration and destruction of habitats rank among the most serious threats to coastal areas. Half of the world's wetlands, and even more of its mangrove forests, have been lost over the past century due to physical alterations, with accelerating social and economic development and poor planning being major causes. All causes of degradation are more intense in estuaries (UNEP/GRID-Arendal 2002b).*

4 HUMAN AND COASTAL SYSTEM HEALTH

Threats to human health resulting from human interaction with aquatic ecosystems involve multiple factors, which might be broadly grouped into three categories that are usually inter-linked: (1)Effects of water pollution (chemical, microbiological, radioactive, thermal) on humans and on the physiology of individual organisms; (2) The result of management of aquatic resources (e.g., wetland drainage, land reclamation, dredging, aquaculture, etc.); (3) Effects of global change affecting climate and the hydrological cycle (e.g., habitat degradation, warming, increased rainfall, storms). It is obvious that the role of humans in decreasing the quality of life in the marine environment has been enormous. The detrimental effects of pollution can directly or indirectly affect all forms of life in estuaries and this, in turn, can similarly present a hazard to human health when marine organisms are used as food or by direct contact with the water and sediments. A significant number of human activities contribute to the release of pollutants to the estuary by rivers and tributaries, run-off, groundwater and the atmosphere. Considerable amounts of these contaminants remain inside the estuary or adjacent coastal zone, particularly in poorly flushed areas.

5 MANAGING COASTAL WATER RESOURCES: A SHARED RESPONSIBILITY

In simple terms, the great challenge of this century is to find the means to develop human capital (socio-economically, culturally and equitably), while at the same time preserving and protecting natural capital. The socio-economic dimension, with its focus on human concerns, is a crucial component of the approach, taking full account of:

1. Stakeholders having input in the planning and management of the resource, ensuring especially that the interests of all quadrants of society, as well as natural interests, are fully represented;
2. The multiple uses of the resources and the range of people's needs;
3. Integrating water plans and strategies into the national planning process and environmental concerns into all government policies and priorities, as well as considering the implications of proposed and adopted actions;
4. The essential needs of the ecosystems so that they are properly protected.

Summarizing, coastal management embraces the principles of participation and transparency to attain social equality and good governance. This reaffirms that governments are the stewards of valuable assets such as their coastal waters and ecosystems, and with effective cooperative management, these common resources can contribute to sustainable economic growth.

6 THE ECONOMICAL AND ECOLOGICAL CONTEXT OF NATURAL RESOURCE MANAGEMENT

Management decisions based purely on economical considerations always compare the current market value of the natural resources against an ill-defined ecological value. However, in making decisions, relevant factors must include such things as the relative value of ecological processes, conservation priorities, and alternative land use practices. Because of the uncertainty in forecasting the economic value of alternatives, and the difficulties inherent in defining very different values in the same currency, decisions end up being taken mostly by political and social forces on short term economical considerations The concept of intergenerational equity is usually part of definitions of sustainability but this concept is intractable from a manager's perspective. Also, management decisions usually rely on dealing with the resource as it is now (e.g., available space, fish populations, etc.), instead of on the basis of their sustainability. From the perspective of developing procedures for sustainable resource use practices, there is more at stake than simply the particular interests of the resource owners or managers. A number of general principles have been suggested for the discussion of sustainability and resource use (Ludwig et al. 1993):

- Human motivations and responses should be included as part of the system to be studied and managed;
- Past examples show that resource exploitation has seldom been sustainable, and frequently scientific advice is ignored;
- Resources should be managed explicitly for uncertainty by considering a variety of different strategies, favoring actions that are informative, reversible and that are robust to uncertainty;
- Management strategies should be adaptive, considering uncertainty and surprise as an integral part of anticipated response;
- Policies and actions are required that involve not only social objectives, but that continue to improve understanding and provide for flexibility in the event of unexpected events;
- Trial-and-error must be seen as an integral part of adaptive management;
- Such an approach should be interdisciplinary and combine historical, comparative and experimental approaches to resource use.

7 ECOSYSTEM-BASED MANAGEMENT: THE SUSTAINABLE USE OF ESTUARINE RESOURCES

Sustainability is both an ecological and an economic concept. Achieving adequate sustainability practices must be a priority to maintain or restore these environments and the ecological services they provide. This, in turn, will be translated in economic benefits. Sustainable use of estuarine systems and the goods and services they support depends on: (1) efficient coupling between advances in the environmental sciences and their application for the public good, and (2) our understanding of the interdependency of ecological and socio-economic systems. Today, there are unacceptable disconnects between these processes on both counts (Bowen and Riley 2003). Ecosystem-based strategies consider the effects of human activities in the context of natural variability and change. Ecosystem-based management is emerging as a unifying approach to secure and efficient marine operations, environmental protection, resource management, land-use planning and environmental engineering (Sherman and Duda 1999, Cicin-Sain et al. 2000). This is especially significant in estuaries where the combined effects of habitat alterations, land sources of pollution, over fishing, harmful algal blooms, and invasive species are most severe (Botsford et al. 1997). Implementing a strategy of ecosystem-based management requires the capability to engage in adaptive management and a decision making process that depends on routine and rapid detection of changes in the state of the system. Although the challenges are many, the coordinated development of a multidisciplinary scientific effort may provide an important means to bridge the gap between science and management by the routine and repeated provision of scientifically credible, quantitative assessments of the status of estuarine ecosystems across the land-sea interface.

8 MEETING SOCIAL NEEDS

Estuaries and other coastal systems are experiencing unprecedented changes that make them more susceptible to natural hazards and less able to support living resources. A broad spectrum of phenomena from global warming and sea level rise to harmful algal blooms (HABs) and losses of biodiversity are exhibiting troubling trends in their magnitude or frequency. These trends are related to both natural processes and increasing human demands on coastal ecosystems to support commerce, living resources, recreation, and living space.

One of the major challenges in coastal zone management, and particularly in estuarine management, is to balance the environmental constraints with social needs. This is a rather demanding task because it starts with a fairly consensual identification of major social goals followed sometimes by a not so obvious definition of particular social needs for any specific system. Because of their nature, societal goals can only be achieved with the development of an integrated and holistic approach. They can be broadly summarized in the following list (UNESCO 2005): (1) Improve the safety and efficiency of marine operations; (2) Mitigate and more effectively control the effects of natural hazards; (3) Minimize public health risks; (4) Protect and restore healthy ecosystems in a more effective way; (5) Improve the capacity to detect and predict the effects of global climate change on coastal ecosystems; (6) More effectively restore and sustain living marine resources.

To achieve these goals, an informed management for sustainable use of estuarine services requires the capability to routinely and rapidly assess their state and health, detect changes on a broad spectrum of time and space scales, and provide timely predictions of likely future states. Relevant and informed decisions, whether they are concerned with ship routing, beach closures, fisheries management, dredging disposal, or mitigating the effects of an oil spill caused by an accident, require the provision of useful marine data and information at rates tuned to the time scales at which decisions must or should be made. This begs for an integral approach to achieve an appropriate set of priorities in the estuarine zone management that equilibrates ecological and social and economical needs. For the time being this capacity is still in an embryonic stage in most countries.

9 INTEGRATED APPROACHES TO ESTUARINE ECOSYSTEM MANAGEMENT

Estuarine areas, with their overlapping economic interests competing for the same common property resources, are where integrated approaches are most urgently needed. Integrated coastal management usually focuses on three major goals: (1) overcoming the conflicts associated with the sectoral management, (2) preserving the productivity and biological diversity of coastal systems, and (3) promoting and equitable and sustainable allocation of coastal resources (Post and Lundin 1996). The forms of integration required by coastal management have many dimensions. An example is the combination of good science with governance. Being complex systems under significant human pressures, estuaries cannot be managed in the absence of the best available information of both biophysical and social sciences. Marine sciences help characterize problems over time, distinguishing natural and human-related causes of environmental change. When combined with the results of economic and social research, these efforts contribute to innovative management solutions. Another example is the integration among sectors and disciplines. The complex overlay of processes and institutions in estuarine areas makes it impossible for a single agency or entity to meet the challenges of management alone. Success lies in forging partnership among institutions meaning, among other things, to develop an interdisciplinary dialog to achieve a holistic approach to the management of the system.

9.1 The interdisciplinary and cross-sectoral dialog

Decisions concerning water related resources management are too often taken without sufficient scientific and empirical background, addressing only short term and single goals, and ignoring the complexity of processes in aquatic ecosystems (Naiman et al. 1995). Management of socioeconomic interests and associated environmental issues requires practical assessment techniques that should be based on an interdisciplinary approach. An effective management tool should always consider the development of protection policies to reduce impacts on the environment. Developing a fully integrated, multi-disciplinary system for estuarine management has been a particularly challenging task because: (1) it requires systematic monitoring and research activities and these are usually non-existent or primitive at best, and (2) the operational capacity for detecting, assessing and predicting changes in ecosystem health, the sustainability of natural resource and public health risks, is poorly developed.

9.2 A holistic approach to the study of estuarine systems

The recent cooperation between ecologists and water managers has led to attempts to integrate an ecosystem approach into Integrated Water Resources Management (IWRM). The rationale in this approach has been to conceptualize a catchment- or basin-based holistic approach, which takes into consideration the multiple roles of water both in ecosystems and in human socio-economic systems. This involves consideration of terrestrial and aquatic ecosystems and the water links between them, requiring from water managers an understanding of the linkages between water circulation and ecosystems.

Fundamentally, this is a response to the much-criticized fragmented sector-by-sector approach to water-related resource management (aquaculture, sewage disposal, fishing, recreation, etc.). The new paradigm highlights instead the benefits that an integrated, overall approach to water management, on a catchment or basin basis, can deliver. IWRM promotes not just the cooperation across sectors, but also the coordinated management and development of land and water (both surface water and groundwater), so as to maximize the resulting social and economic benefits in an equitable manner, without compromising ecosystem sustainability. It is imperative to address the ecological and socio-economic links in the management of dynamic systems such as estuaries and coastal areas. The Driver-Pressure-State-Impact-Response (DPSIR) model (Bowen and Riley 2003) provides a framework for achieving this. This methodology will be addressed in the next chapter.

10 FINAL REMARKS

Estuarine areas have been and remain in close association with humans. This relation of human dependence on the natural characteristics of estuarine systems implies that these resources have to be managed in ways that guarantee their sustainability, but also in such a way that social and economical structures have viable development targets. Considering the scale of the multiple resource demands imposed on estuaries they, rank among the ecosystems under heaviest (and increasing) pressure. The multiple usage of the goods and services provided by estuarine systems reflects the variety of stakeholder interests and perceptions. To attain a sustainable utilization of estuarine areas and resources requires a far more complicated approach of social, economic and environmental issues, than is the case for purely marine or purely terrestrial environments. The effective use of available information in development planning and management for estuaries depends on an Integrated Water Resources Management strategy. Resource sustainability cannot be detached from the sustainability of human economies, natural communities and ecosystems. Sustainability is a moving target because ecosystems change over time and so do the economic, social and political climates in which decisions are made change. The rate of development of estuarine areas and the resulting environmental impact may largely be determined by the efforts that are devoted to the early detection of environmental problems. Hence, the successful management of estuaries and coastal waters requires a basin-wide management, which considers the river basin as the fundamental unit of territorial management (Zalewski 2002). To pursue this goal means that present practices by official institutions based on municipalities or counties as an administrative unit, or based on managing specific activities must be changed or abandoned altogether. Without these changes and a holistic approach towards the management of these systems, estuaries and coastal waters will continue to be degraded.

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THE DPSIR FRAMEWORK APPLIED TO THE INTEGRATED MANAGEMENT OF COASTAL AREAS

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1 INTRODUCTION

Over the last decades numerous and diverse problems with ecological implications have challenged both environmental scientists and decision-makers. These problems have ranged in scale and magnitude: global climate change, loss of habitat and biodiversity, habitat destruction, and effects of multiple anthropogenic chemicals on ecological systems. Extant and emerging problems have highlighted the need for flexible approaches to deal efficiently with the problems by establishing a link between ecological data with the needs of decision-making environmental managers. Several methodologies and tools have been developed and used to face the multifaceted challenges posed by the management of resources where apparent conflicts of interest exist. Ecosystem function is affected by human activities through the disturbance of energy and matter flow (Ohl et al. 2007). These changes in ecosystem processes influence biodiversity, change the ecological state of ecosystems and impact both on society and the economy. Thus the inclusion of socio-economic dimensions into standard ecological research has been identified as a challenge in the new paradigm of sustainable development and management of natural resources. Efforts to expand the understanding of these interdependencies have led to improvements over the last decade Bowen and Riley 2003), mainly by using socio-economic indicators that link the changes in environment to social and economic drivers, and political responses. The challenge has been to understand the relationships between social/economic interests and associated environmental issues, which require practical evaluation techniques based on an interdisciplinary approach.

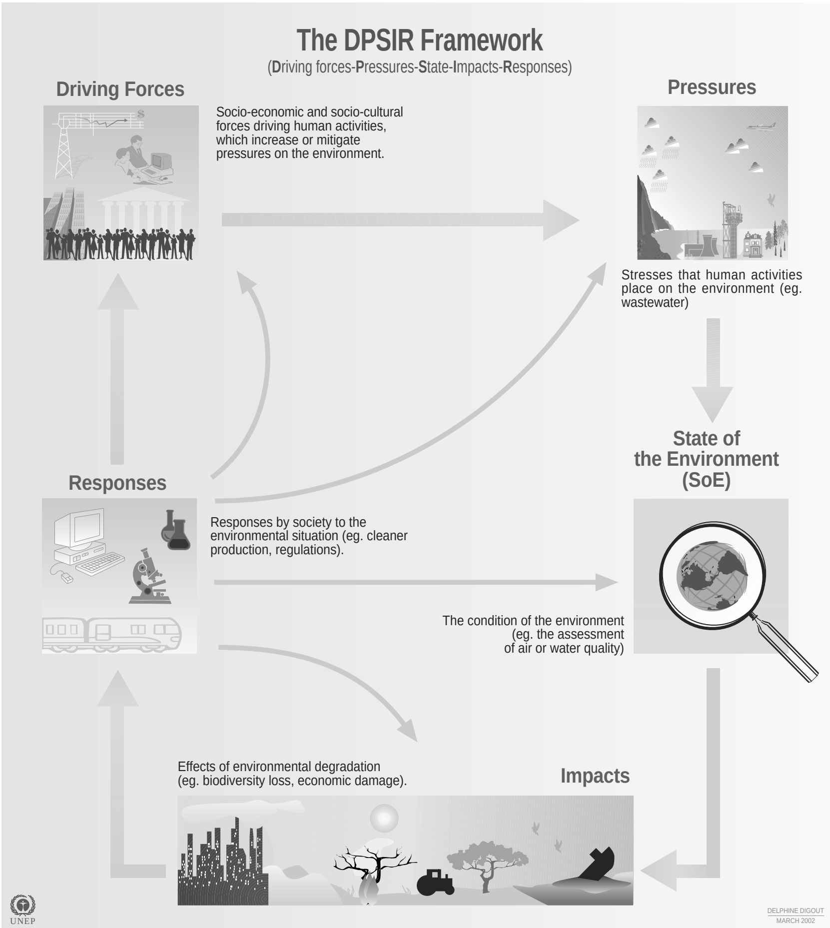
Together with the multidisciplinary approach required by the new demands on the management of resources with a holistic perspective, a multi-sectoral approach must also be considered. The parties involved in the process: scientists, civil servants and stakeholders all speak different languages, function in response to different reward systems, and work on different time scales. The realization of the magnitude of these problems led to the development of integrative approaches able to deal with these diverse requirements and still provide realistic solutions. The DPSIR (Drivers-Pressures-State-Impact-Responses) framework is such a tool (Figure 1), allowing the description of environmental problems by defining the relationships between anthropogenic activities and the environment. The framework provides a better context in which to integrate different types of indicators, opening the possibility of taking into account not only the environmental but also the socio-economic impacts that result from changes in the state of coastal systems. Also, it places side-by-side environmental and socio-economic interests. The DPSIR framework helps to allow sustained and routine provision of quality environmental data and information and the availability of sound scientific advice to enable responsive government decisions and to enhance the effectiveness of management actions.

2 THE BACKGROUND OF THE DPSIR FRAMEWORK

The origins of the DPSIR framework go back to the Stress-Response framework developed by Statistics Canada in the late 1970s (Rapport and Friend 1979). This first framework was later extended in the 1990s by, among others, the Organization for Economic Co-operation and Development (OECD 1991, 1993) and the United Nations (UN 1996, 2001), resulting in the PSR (Pressures, States and Responses) framework. Also during the 1990s, this paradigm was further extended to its present form of the DPSIR framework, originally in two studies by the European Environmental Agency (EEA 1995, Holten-Andersen et al. 1995). The objective of these frameworks has been to clarify multi-sectoral relationships and to highlight the dynamic characteristics of the ecosystems and socioeconomic changes (Elliott 2002). All these frameworks share the distinction between (i) forces that act on the environment, (ii) changes that, as a consequence, take place in the environment and (iii) the societal reaction to those changes. The DPSIR framework follows the same general model as previous frameworks but diverges in the sense that it distinguishes more steps in the process (Niemeijer and de Groot 2008). So while there are some differences between these frameworks in terms of terminology and the degree of detail, they are all based on the causal chain concept.

The DPSIR Framework is an instrument for analyzing environmental problems, with regards to sustainable development (Borja et al. 2006). The basic aims of its approach are: (i) to be able provide relevant information on the different elements of the DPSIR sequence, (ii) to clarify the ways in which they are connected and related to each other and (iii) to estimate the effectiveness of responses. The DPSIR framework provides helpful insights on the relationships between the origins and consequences of environmental problems and, at the same time, helps to understand their dynamics by addressing the links between DPSIR elements. This integrative approach presupposes substantial understanding of the underlying causal relationships between human activities and the resultant impacts on ecosystems, coastal economies and communities, and human response mechanisms. Nevertheless, the integrative nature of the framework leads to its wide use, especially by the European Environmental Agency, in selecting indicators for evaluating the implementation of EU environmental policies.

The DPSIR framework has rapidly become popular among researchers and policy makers alike as a conceptual framework for structuring and communicating relevant environmental policy research (Svarstad et al. 2008). For this reason it has been successfully implemented in different kinds of coastal management issues, and its contribution to highlight the dynamic characteristics of ecosystem and socio-economical changes has been validated (Turner et al. 1998). A presumed strength of the DPSIR framework lies in its simplicity to capture key relationships between factors in society and the environment. While simple in concept, the framework is flexible enough to be conceptually valid over a range of spatial scales (von Bodungen and Turner 2001). Consequently, it can be used as a communication tool between scientists from different disciplines as well as between researchers, on the one hand, and policy makers and stakeholders on the other.



Source : Global International Water Assessment (GIWA), 2001; European Environment Agency (EEA), Copenhagen.

FIGURE 1: *DPSIR framework for State of Environment Reporting (UNEP/GRID-Arendal 2002).*

3 THE HEURISTIC DPSIR-CONCEPT

The Driving forces-Pressure-State-Impact-Response concept (DPSIR) provides a heuristic framework for the analysis of cause-effect relationships in complex systems which are subject to human action (Brandt 2000). The general idea behind the DPSIR concept is that human activities, i.e. the drivers, exert a certain pressure on a particular part of the natural environment causing a change in its components and/or in its overall state. The outcome of this process is an environmental impact, which usually results in certain responses by society. The response can run across different segments of society, from the political arena, to socio-economic and purely economic sectors. Eventually, responses can modify the nature of the driving forces (thus mitigating or even enhancing the actual pressure) and/or compensate for the impact. Finally, the driving forces may also be altered directly by the impact.

A clear example in many estuaries relates to sewage discharges in the system. The increased demand for housing (Driving force) can lead to the intensification of direct discharges of untreated sewage in the waters, resulting in the increase of nutrient loads and faecal contamination of nearby streams (Pressure), leading to the eutrophication of water bodies (State) and subsequent changes in aquatic life and biodiversity and contamination of food resources (Impact). One way to address this situation (Response) would be to improve the sanitary system; another would be to require changes in occupation practices and even to ban the consumption of contaminated marine organisms such as shellfish.

3.1 Drivers of (environmental) change

The first step in the DPSIR framework is the definition of the driving-forces that lead to environmental pressures. For this first step it is important to identify the major stakeholders, their values and interests, and also the potential conflicts between them. A driving force, also termed a driver, is an established social need that represents a factor and social force that may induce changes in the state of the environment. This social need usually arises from the economical sphere, which means that drivers are frequently linked to the financial system. As such, drivers are usually considered to be economic and social goals of those involved in the industry, as well as economic and social policies of governments. In coastal areas, shipping, fisheries, tourism and aquaculture are among the most commonly mentioned drivers of DPSIR models.

3.2 Pressures (on the environment)

Pressures can broadly be described as the means through which drivers are actually expressed, i.e, in the way they interfere and perturb the system. Inside the framework, pressures are the link between socioeconomic activities and the natural system. In a sense, all human activities end up by generating pressures on the environment, to a lesser or greater degree. The existing pressures on estuarine and coastal areas can be divided into four groups: (i) pollution, comprising urban, industrial, agricultural and aquaculture discharges; (ii) alteration of the hydrological regime, including water abstraction, flow regulation and restoration activities; (iii) changes in the morphology, including land reclamation and infrastructures; and (iv) biology and its uses, including all kind of resource exploitation, changes in biodiversity and recreation (Borja et al. 2006). As such, pressures fall into three general categories that range from simple interference to inducing changes in the natural functioning of ecosystems: (a) fluxes into water bodies, (b) excessive usage of natural resources, and (c) changes in the food web.

3.3 The state of the environment

The combination of physical, chemical and biological conditions defines the state of the environment in a given area. This state is affected by the pressures and eventually modified in its environmental conditions. The result of this induced change may be expressed as a loss of ecosystem services. So, if the state is changed, human dependence on the system may also be compromised (e.g., loss of fish stocks, bathing areas, etc.). Although there is a link between pressure and state, the relationship between them in estuaries is strongly influenced by geomorphology and hydrodynamics: estuaries subject to similar nutrient-related pressure often exhibit totally different eutrophication symptoms, and in some cases no symptoms at all. Factors such as flushing time, tidal range, and turbidity play a major role in determining the nature and magnitude of symptom expression.

3.4 Environmental and societal impacts

The state of the system needs to be assessed in terms of its physical, chemical and biological conditions, and this leads to the definition of impact on each component. Thus, impacts correspond to the effects resulting from the change in the state of the ecosystem. Usually these effects are studied by identifying changes in bio-physical-chemical conditions that lead to changes in the components of the environment (e.g. water quality, biodiversity etc.). However, this has also impacts on society. Hence, environmental impacts are related to the health of the ecosystem, while social impacts are linked to effects on human health and to the effects and resources that society identifies as valuable. An assessment of the impacts requires monitoring procedures and the definition and use of indicators of change.

3.5 Societal responses

The DPSIR model assumes that all pressures degrade the ecosystem, and that this negative impact can only be reverted through subsequent responses. This means that the magnitude of the impacts may lead to a re-evaluation of current management policies and may eventually lead to the realization of the need of different management responses. In this sense, a response is a societal action related to an actual environmental problem or perceived risk. This action, often moved by public policies of governmental actors, can also be stimulated by other sectors of civil society such as NGOs, universities, etc. A response can be described as a reaction to the negative effects of impacts. The responses vary according to the scale of the impacts, becoming an attempt to mitigate the impacts or reverse them in an attempt to reestablish the ”normal” state of the system, if possible. If preventive measures are taken to eradicate or ameliorate the impacts of pressures in the system, then it will change the original drivers.

The human or societal response to the changes resulting from our activities has to be established to meet what we may call *six tenets for environmental management* (Elliott 2002). Some of these tenets are well-known in national and international strategies (the first three), while others need to be considered to guarantee that solutions to environmental change sit within our developed systems. Accordingly, our actions (Responses) have to be: (1) Environmentally sustainable (i.e. nature-friendly in the present and in the future); (2) Technologically feasible (i.e. with adequate methods and equipment); (3) Economically viable (i.e. at a reasonable and supportable cost); (4) Socially desirable (i.e. wanted by our societies); (5) Legally permissible (i.e. in compliance with national and international legislation); (6) Administratively achievable (i.e. carried out and enforced by our system of departments, agencies and governments).

4 METHODOLOGIES USED IN THE DPSIR ANALYSES

The integrative nature of the DPSIR framework in the study of human-ecosystem interactions means that a significant range of techniques, methodologies and tools must be used to achieve that end. The temporal scope of the framework, with processes spanning across different time scales, and addressing present and future states, requires these methodologies and tools to diagnose and predict. Some of the procedures are of a descriptive and static nature, i.e. they give a snapshot of the actual state of the system (e.g., environmental indicators, conceptual models), while others are dynamic, meaning that they can describe the temporal evolution of the system to some degree (e.g. mathematical models).

4.1 Environmental indicators

The use of indicators is fundamental in the DPSIR framework because they provide an objective system of information and evaluation. An indicator can be described as something that provides a clue to a matter of larger significance or makes perceptible a trend or phenomenon that is not immediately detectable (Hammond et al. 1995). In the DPSIR context, the European Environment Agency (EEA Glossary, 2007) describes an environmental indicator as "*a parameter or a value derived from parameters that describe the state of the environment and its impact on human beings, ecosystems and materials, the pressures on the environment, the driving forces and the responses steering that system.*" As such, an environmental indicator is a qualitative or quantitative parameter characterizing the current condition of an element of the environment or its change over time. Such environmental indicators have three basic functions (Aubry and Elliott 2006):

- To simplify, considering that only a few indicators are selected according to their perceived relevance for characterizing the overall state of the ecosystem.
- To quantify, because the value of an indicator is compared with reference values considered to be characteristic of the state of the ecosystems, thus quantifying the shifts from reference or expected conditions.
- To communicate, by facilitating the transmission of meaningful information on environmental issues to stakeholders and policy makers, by promoting information exchange and comparison of spatial and temporal patterns.

Indicators are increasingly being developed and used as management tools to address environmental issues. Over the last years, environmental indicators have taken on such importance because they provide a signal that communicates a complex message in a simplified and useful manner (Jackson et al. 2000). Also, environmental indicators provide an important source of information for policy makers and help to guide decision-making as well as monitoring and evaluation, because they can provide valuable information on complex issues in a relatively accessible way.

Environmental indicators have come to play a vital role in environmental reporting as prime assessors of pressures on the environment, of the evolving state of the environment, and of the appropriateness of policy measures (Niemeijer and de Groot 2008). But because they are so important, it is a major challenge to determine which set of parameters and values of ecological systems characterize the entire system and still are simple enough to be effectively and efficiently monitored and modelled (Dale and Beyeler 2001). Indicators, therefore, need to be properly selected and the methodology of their calculation specified if the dynamic parts of a given system are to be understood and appear compelling to the user communities (Bowen and Riley 2003).

4.2 Environmental modelling

Environmental modelling is an explicit treatment of our understanding of the deterministic and stochastic mechanisms that affect our studied object (Akçakaya et al. 1997). Numerical models stand as a way to look at real systems and to translate them into compartments, identifying the connection between them. They are versatile tools that enable an in-depth look at natural systems which cannot be achieved by the simple combination of analytical methods. The use of models makes it possible to explain cause and effect in environmental processes, distinguish between anthropogenic and natural contaminant sources and their respective impact, etc.. Modelling results are also important to complement and interpolate data from traditional observational research methods. Because models have the capability to bridge the gap between small scale and large scale processes, they become an essential tool for understanding complex processes that link different compartments of the system (e.g., benthic and pelagic systems) and run across the land-sea interface by linking catchment and estuarine processes. This is particularly relevant in eutrophication-related studies, where nutrient dynamics can be addressed in the vast context of major biogeochemical cycles (Harrison 1992).

Models are increasingly becoming indispensable tools in environmental studies and management decisions (Neves 2007). In the DPSIR framework, models are commonly used to elucidate each component and the relation between the different components (e.g., the pressures with the state). Combining the DPSIR with numerical models allows the generation of predictions on potential levels of selected impacts, making responses actions ”prior” to the full manifestation of those impacts in the environment. However, it is obvious that no model will ever be able to address all problems and answer all questions. For this reason there are so many types of models. Water quality models, ecological models, hydrodynamic and groundwater models are just a few examples.

Most models address specific disciplines of knowledge but can be coupled to other models to achieve an integrated model approach to the study of natural systems. The use of models in decision making must have the main objective of improving communication and understanding of the nature of the problems. If they achieve this goal, the results they produce will be integrated quite naturally with value judgments and political constraints. This will result in better decisions than would have been made if the models had not been used. To produce this outcome, models must be carefully and thoroughly documented, and limitations, sensitivities and assumptions must be explicitly stated. In addition, modelers must be sensitive to the needs and limitations of the people who intend to use them. It is as important, if not more important, for the ecologist to communicate the uncertainties and assumptions underlying the model, as it is to communicate the set of predictions. Ultimately, the relevance of models for environmental decision-making is in the mind of the policy maker, and not in the expert opinion of the modeler.

4.3 The role of conceptual models

It is questionable whether we will ever have fully validated numerical models that can adequately predict the ecological effects of human activities. Even so, models can be seen as serious attempts, and probably the most adequate, to relate human drivers with ecological states. The first step towards creating such models is to have some knowledge on the physical and biological features of the system and the definition of the problem, hence the production of conceptual models. Considering trends in marine environmental management, it is in fact fundamental to develop conceptual models. For simplicity, these can be regarded as diagrams which bring together and summarise information from many areas.

The schematic approach of conceptual models confer them the simplicity that lengthy and detailed descriptions cannot. As such, they have an educational significance and at the same time provide the basis for communicating the main message to managers and developers. Conceptual models are usually a good starting point for developing quantitative and dynamic numerical models, or to point to the limitation of such models and the available scientific knowledge. They also have the advantage of exposing gaps in knowledge, thus helping to define further field and laboratory studies to fill these gaps. In particular, they allow a problem to be deconstructed as a precursor to each aspect being assessed, prioritized and addressed (Elliott 2002). Under the DPSIR framework it is essential to be aware of the spatial and temporal links in the marine system. This, in turn, has to be coupled with the diverse nature of stressors on the systems which requires conceptual models to be linked together and further developed towards numerical and predictive models.

4.4 Stakeholder's involvement through participation

"*Environmental issues are best handled with participation of all concerned citizens, at the relevant level*". This is the introductory statement of Principle 10 of the Rio Declaration on

Environment and Development (1992). This principle states that individuals must have the opportunity to participate in decision-making processes and that States shall promote and encourage public awareness and participation by making information widely available. Coastal management programs must ensure strong public involvement of stakeholders, because they represent the people who are most affected by the coastal development process. This is best achieved by making public education and consensus-building important components of any initiative. The responsibility of stakeholders must go well beyond the awareness to participate in the decision making process; they must also be held accountable for such task.

A way of incorporating stakeholders' opinions into the decision-making process has been through the methodology of participative experts' model (Failing et al. 2004). In this strategy, the model of the system, irrespective of the complexity, starts by being generated with the participation of some stakeholders or social actors. As a rule, the chosen stakeholders are those that the group of experts identifies as more relevant to the potentially analyzed problem. This degree of involvement confers common sense to the modelling exercise by keeping the aims of scientists at realistic levels, and assures that the model is not socially naive. An open dialogue between scientists and stakeholders is necessary in order to make decisions regarding what can be done and what shall be left either to other scientists or other modelling tools, or both. As addressed before, the most important feature about models is that they must be relevant to decision-makers. This means that if models do not include stakeholders in their development, the study on the availability of significant societal resources might be doomed to failure. If the process of model building is collaborative and iterative, and if it involves representatives of all stakeholders, it has a chance of being realistic, hence useful, i.e., will have the ability to answer the right questions.

A well-structured decision process involving stakeholders can typically be summarized in three key steps Keeney 1992, Clemen and Winkler 1999):

- Setting objectives and indicators for each of them. These indicators (also known as performance measures) become the criteria for evaluating and comparing policy alternatives. Since setting objectives is a deliberative and value-based activity, it demands input from a broad range of stakeholders. Defining indicators is both deliberative and analytical, requiring involvement from both technical specialists and stakeholders.
- Identifying policy alternatives and assessing their impact on the objectives. The impact of the policy alternatives is measured by the indicators. The description of impacts should explicitly characterize the uncertainty associated with the estimate. This is an analytical activity, conducted largely by technical experts, with input from stakeholders in the form of selecting the experts and defining their terms of reference.
- Evaluating and choosing a preferred policy alternative. Choices will most likely involve trade-offs among competing objectives and methods for making choices should allow stakeholders to state their preferences (value-based information) for different outcomes, based on good information (factual or technical information). This again is a deliberative task involving both scientific and stakeholder participation.

5 THE DPSIR FRAMEWORK AND THE INTEGRATED COASTAL ZONE MANAGEMENT

Integrated Coastal Zone Management (ICZM) promotes sustainable management of coastal areas in a dynamic, multidisciplinary and iterative process. It includes all the processes involved in this task, from information collection, planning, decision making, management and monitoring of implementation. It is also a process that involves the informed participation and cooperation of all relevant actors to evaluate the societal goals in a given coastal area, and to take actions towards meeting these objectives. In the long term, ICZM tries to achieve a balance between environmental, social, economic and cultural goals, always keeping within the limits set by natural dynamics. The integrative nature of this approach is in its range of objectives, but also in the integration of the many instruments needed to meet these objectives, as well as the integration of the terrestrial and marine components of the target territory, in both time and space.

In recent times, a few new concepts have come out related to coastal managemen, whose application has been encouraged by institutions (i.e. EU Parliament and council 2002/413/EC), and the DPSIR framework as a tool for the former. The ICZM concept is based on a holistic approach to manage conflicts between different coastal uses and interests (aquaculture, resource extraction, tourism, housing, etc.) and to facilitate the use and dissemination of information, especially between society, managers and scientists. Today, DPSIR is increasingly used as a framework for structuring case studies in relation to issues of human interferences in an effort to manage landscapes and seascapes (Elliott 2002, La Jeunesse et al. 2003, Scheren et al. 2004, Holman et al. 2005). The DPSIR approach has become increasingly accepted and applied to different case studies to solve problems involving a range of coastal marine environments: coastal areas, coastal lagoons, deltaic systems, estuaries, river basins. A summary of applications of the DPSIR framework to marine environments is presented in Table 1. In this sense, the DPSIR framework has received much attention and use in ICZM strategies and programs.

ICZM efforts worldwide face major challenges. This is particularly evident in estuarine management, where the goal is to balance environmental constraints with social needs, while maintaining the habitual fragile balance between ecosystem performance and human-related activities. Because of their nature, societal goals can only be achieved together with environmental goals with the development of an integrated and holistic approach. The DPSIR framework is an effective way to deal with complex issues, such as the management of nutrient fluxes (Smith et al. 1999) and the impact of development in catchment areas (Cave et al. 2003), inside the broader scope of the ICZM programs.

6 FUTURE PERSPECTIVES

There are no straightforward answers to the question of what is best for a particular system when there are potential conflicts between natural and economic interests. It is the task of scientists from different disciplines to present as complete as possible a picture to those who make decisions or have the capacity for lobbying in the decision-making process. To achieve a holistic view of these systems and fully incorporate the needs of policy-makers, frameworks such as the DPSIR framework are essential. In complex ecosystems like estuaries, where the human presence and activity is growing at an alarming pace, there is an urgent need to link science (the knowledge on the system functioning) to the causes of change in its state and to the social, economic and legal responses by Man to that change. This necessity is behind the increasing use of the DPSIR approach (Elliott 2002).

TABLE 1: *A summary of applications of the DPSIR framework in ICZM strategies.*

Study site	Area	Subject	Reference
Guanabara Bay basin (Brazil)	River Basin	Sustainable Development	Bidone and Lacerda (2004)
Thermaikos Gulf (Greece)	River Basin	Hindcasting coastal evolution	Karageorgis et al. (2006)
Po Catchment-Adriatic Sea (Italy)	River Basin & Coastal area	ICZM	Pirrone et al. (2005)
Aixos River catchment and Thermaikos Gulf (Greece)	River Basin & Coastal area	Eutrophication	Karageorgis et al. (2005)
Southern European Coastal Lagoons	Coastal Lagoons	ICZM	Aliaume et al. (2007)
Sacca di Goro (Northern Adriatic Sea, Italy)	Coastal Lagoon	Aquaculture impacts	Marinov et al. (2007)
UK Coast	Coastal area	Offshore wind power	Elliott (2002)
Nestos Delta (Greece)	Coastal area	Environmental status indicators	Karakos et al. (2003)
Italian Coast	Coastal area	Coastal environment assessment	Casazza et al. (2002)
Ria Formosa (Portugal)	Coastal Area	Dredging activities	Pacheco et al. (2007)
Ria Formosa (Portugal)	Coastal Area	Eutrophication	Newton et al. (2003)
Bay of Gdansk (Poland-Russia)	Coastal Area	Eutrophication	Kannen et al. (2004)
German Coast	Coastal Area	Future Planning	Kannen (2004)

DPSIR was projected to explicitly relate environmental changes driven by socio-economic pressures with the required socio-economic measures to mitigate adverse impacts of change caused by human actions. For estuaries and coastal areas in general, the DPSIR analysis has the ability to link large-scale human drivers of change and their impacts on the systems, with management responses (e.g., sewage treatment, preservation of mangrove areas, modifying dredging activities, etc.). A major advantage of the framework lies in its capacity to integrate socio-economic aspects with ecological impacts, addressing not only the consequences of human activities on the system, but also its feedback.

The DPSIR framework works well at simplifying the complexity of natural systems management, such as estuarine areas, at the same time informing policy makers, scientists and the general public on the actions that can cause changes in the status of the system and the nature and consequences of those changes. Several shortcomings have been ascribed to the DPSIR framework as a tool for establishing effective communication between environmental scientists of different disciplines, and between stakeholders and policy makers. One of these shortcomings seems to be the lack of efforts to find a satisfactory way of dealing with the multiple attitudes and definitions of issues by stakeholders and the general public (Svarstad et al. 2008). Nevertheless, this framework has proven to be an effective way to deal with the complex task of managing natural system when real conflicts exist in regard to their use and transformation and, most important, a central methodology for establishing cause-effect relationships in the use and exploitation of natural resources and their status. The DPSIR framework is a practical tool for testing observations and hypotheses. It is being used successfully and increasingly as a research aid to interpret ecological relationships in ongoing evaluations of management alternatives and to develop effective ecological and societal targets for a meaningful, conflict-free, sustained and sustainable development.

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COASTAL ZONE MANAGEMENT IN SOUTH AMERICA WITH A LOOK AT THREE DISTINCT ESTUARINE SYSTEMS

M. MATEUS, F. GIORDANO, V.H. MARÍN AND J. MARCOVECCHIO

1 INTRODUCTION

Estuaries and adjacent coastal areas can be characterized by several variables such as: size, shape and bathymetry, tidal influence, fresh water inflow, turbidity, residence times, sediment properties, and water-column turbidity. Also of great importance are the geographical location (mainly the latitude) and human pressures. In combination, all these characteristics shape the ecology of an estuary, conferring singularity to each system. The ECOMANAGE project focused on three estuarine systems in South America: Santos Estuary (24^o S) and Bahía Blanca (39^o S) in the Atlantic coast, and Aysén Fjord (45° S) in the Pacific coast. These estuaries, in terms of their ecology, differ in features like the role of the benthic system, anthropogenic nutrient inputs, presence/absence of tidal flats. The main purpose of this chapter is to describe in general terms the backdrop of the South American coastal zone management reality, highlighting the major features of these three system. Together, they represent key coastal zones regarding their integrated management. All show conflicting interests between urban, industrial and agricultural development and environmental conservation. Thus, beyond their differences, they share some of the major regional environmental concerns in South America, namely, the transformation of the landscape and seascape with the loss of natural patrimony, increased human waste and industrial disposal (UNEP 1999).

2 THE SOUTH AMERICAN REALITY

The South American region is characterized by a remarkable heterogeneity in climate, ecosystems, human population distribution and economic development. The combination of the prevailing atmospheric and oceanic circulation defines the climate and the land and sea productivity of the region. This partly explains the distribution of human settlements and the availability of basic services (e.g., water supply). According to the medium prospect of the United Nations (Nawata 1999), an increase in population to 838 million is expected for South America by the year 2050. The growth rate of coastal populations in almost every Latin American country is greater than its national growth rate. Nearly 75% of the region's inhabitants live in cities, and 60% of the largest 77 cities are in the coastal zone. As a result, over the last decades South America has become more urban and also much more coastal (Hinrichsen 1997). This concentration along the coast is accompanied by a similarly disproportionate share of the region's infrastructure and economic activity, some of which requires proximity to the waterfront. Land-use changes have become a major force driving ecosystem changes. Up to 19% of the total area of Latin America is used as agricultural lands (excluding pastures). The waters of Peru and Chile support one of the top five commercial fisheries and, until recently, the world's fastest growing fishery thrived off the coasts of Argentina and Uruguay (IDB 1995).

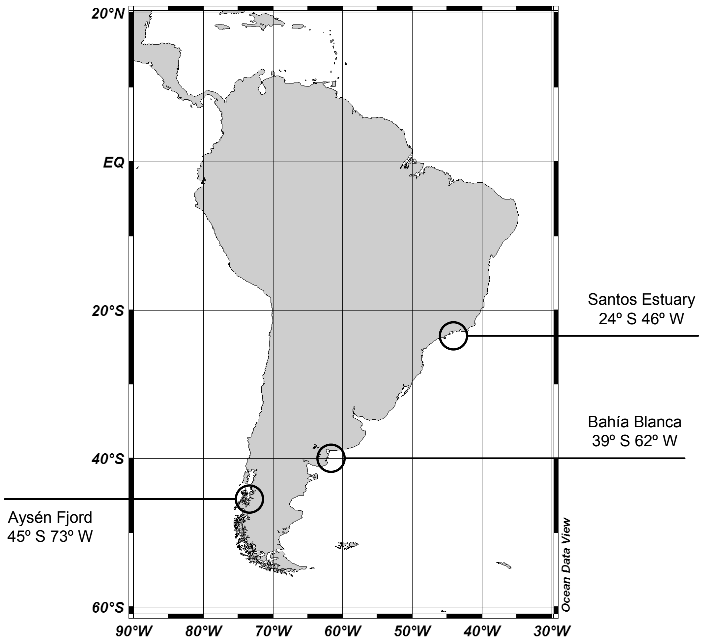

FIGURE 1: *The three ECOMANAGE study sites in South America: Santos Estuary in Brazil, Bahía Blanca in Argentina and Aysén Fjord in Chile.*

2.1 Biodiversity of the Region

South America hosts a significant percentage of the world's biodiversity in terrestrial and marine habitats (Heywood and Watson 1995). For example, there is a large variety of coastal wetlands. The unique location of the region, its extreme climatic variations, tidal patterns, and geological features, make these coastal wetlands rank among the most productive of the world. However, a significant part of them endure the impact of population growth, expansion of the agricultural activity, and land-use changes. However, considering that seven of the world's most diverse and threatened areas are in Latin America and the Caribbean (Myers et al. 2000), the continent faces today serious challenges in natural resources management.

2.2 Socioeconomic factors

From an historical perspective, the use of ocean and maritime access has been at the heart of the southern hemisphere's economic and political development. The ports of South America are important nodes in the flow of goods brought into and exported from the Region. For example, the Region's industrial ports are the second leading destination for containerized U.S. exports. Expanding ports and maritime trade are often accompanied by intensified transportation corridors in coastal ocean areas, as is happening off Argentina, Brazil, Ecuador and Uruguay. The ports of South America are also a significant factor in land use changes in the coastal zone. Most commodity ports serve as development poles for manufacturing and processing activities, often contributing to both the urbanization and increasing industrial character of coastal areas. South American economies are increasingly dependent on trade agreements like the Mercosur that has Argentina, Brazil, Paraguay, and Uruguay as participating countries, and Bolivia and Chile as associate members. These have developed with the main purpose of speeding up socioeconomic development, and have important effects on the economy and, indirectly, on the environment of the region. Even with a strong economic perspective at its core, much of these agreements are gradually considering environmental issues. Indeed, a growing environmental awareness is evident in local legislations in the region. Furthermore, a large percent of coastal populations, especially those with low income, still depends on natural resources exploitation (e.g. artisan fisheries, small-scale aquaculture and farming). As a result, sustainable development strategies and integrated management are rather timely in order to decrease conflicts.

2.3 Environmental awareness

Over the last decades, and especially after Río 1992, South American countries have been developing a strong environmental awareness. This is evident in the ratification of international and regional conventions and agreements, and in the adaptation into the national legislation of many countries to reflect the need of a sustainable development. Two major steps of South American government have initiated a path for preventive actions regarding the use of natural resources: the ratification of the recommendations of the United Nations Conference on Environment and Development (UNCED 1992), and the incorporation of recommendations from the Agenda 21 into national legislation.

As environmental concerns become more pressing, they are receiving more attention on the international political agenda. South America is no exception to this. In line with United Nations Resolution A/52/629 calling for cooperation to incorporate sustainable development programs at national, regional and global levels, countries in South America have been engaged in accomplishing the objectives of such development programs. The large majority of countries follow the recommendations made by the United Nations Commission for Sustainable Development (UNCSD), and the Economic Commission for Latin America and the Caribbean (ECLAC) is assisting them in the integration process of relevant disciplines and sectors. There are also a significant number of regional agreements and a vast body of laws, rules, and regulations to ensure systematic and coordinated actions for protecting the environment and promoting sustainable development (Bertucci et al. 1996, Solano 1997). Most South American governments have developed and implemented comprehensive environmental legal frameworks with relevant laws and procedures for specific resources and activities like marine resources, coastal areas, tourism, etc. However, local empowered stakeholder participation is still in an early stage (Bachmann et al. 2007).

2.4 Challenges in integrated coastal zone management

Habitat transformation (for infrastructure expansion, aquaculture, agriculture, etc.), and sewage and garbage disposal are among the most recurrent problems in South American coastal zones. Water resource management has been identified as a guiding objective by South American countries, with the following subdivisions: (1) Water Supply; (2) Watershed Management; Management of marine coast and related resources; and (3) improved quality of groundwater (UNEP 2003). The legal frameworks for coastal management are being revised in many countries and in some cases modified to change the sectorial focus to a more integrated management approach. This is being pursued having in mind the multiple uses of resources by different sectors like agriculture, fishing, aquaculture, industry, domestic consumption, energy and recreational use. From this perspective, the challenge of coastal ecosystem management in South America can be address with common and shared methods.

3 DRIVERS OF CHANGE IN COASTAL AREAS

Coastal and marine areas of South America undergo fast and frequently drastic transformation. Many of these changes, typical of coastal areas, are experienced as environmental, economic and social problems. For the region, these can be summarized in the following topics, with their relative importance varying from one location to another (Lemay 1998).

3.1 Degradation of coastal ecosystems

Degradation of coastal systems occurs mainly by the combination of land conversion and the expansion of coastal infrastructures. The loss of mangrove areas is a clear example. It has been estimated that 55% of the entire mangrove coast of Latin America and Caribbean was classified as either critical or endangered, 30% vulnerable and only 15% as stable (Olson et al. 1995). In the tropical Americas, the loss of coastal forests, mainly mangroves, occurs at a rate of approximately 1% year (Ellison and Farnsworth 1996). In some parts of the region this poses a threat to local subsistence because most commercial shellfish and finfish use mangrove forests for nurseries and refuge, and so the fisheries in mangrove regions are declining at a similar rate as mangrove communities (Ewell and Twilley 1998). Many areas are experiencing a rapid and often drastic transformation and degradation to coastal and marine areas. Land conversion is causing degradation of coastal habitats, including mangroves, estuaries and coral reefs. Mangroves, for example, have been disappearing at an alarming rate over the past 20 years. Coastal water quality has been declining throughout the region, due to increasing discharges of untreated municipal waste.

3.2 Depletion of commercial fisheries stock

Depletion of stocks, overcapitalization and plant closures, habitat degradation, non-compliance with management regulations and illegal practices, are among the main problems that the fisheries sector faces in the Region. The expansion of aquaculture, which often depends on wild fisheries stocks for seeding and food, also contributes to enhance pressure on natural stocks.

3.3 Land use and resource allocation conflicts

Land reclamation for residential, industrial, agricultural and tourism purposes has caused the degradation of coastal and marine ecosystems of the sub-region. The massive and largely unplanned investments in sectors like aquaculture, port and industrial facilities expansion, and tourism is coastal and estuarine areas has been pointed out as the reasons of accelerated land use changes and associated conflicts. Frequently new activities compete for the same resource upon which traditional communities depend. When compared to other tropical regions such as Southeast Asia, the importance of aquaculture in South America is relatively small. Nonetheless its importance is growing in countries such as Ecuador, where a significant shrimp mariculture industry has developed mostly in mangrove converted areas and salt ponds. More recently important breakthroughs have taken place in aquaculture in Chile (mostly salmon), induced by attractive export markets and made possible by favorable environmental conditions for their growth (Lemay 1998). This activity has been steadily growing at an impressive rate of 30% a year, when compared with 9.5% worldwide. Indeed salmon farming, induced by favorable export markets, is generating around US$450 million a year in export earnings. The environmental impact of this activity has been generating growing concerns, especially because of the habitat losses; eutrophication associated with effluent discharges, other changes is estuarine water quality and the introduction of exotic species.

Tourism investments represent an important catalyst of land use change in coastal areas, and the South American continent is no exception. Tourism has increased in the last decades and this may lead to important environmental impact when it takes place in estuaries and mangroves (Garreta-Harkot 2003). In addition to generating employment, tourism investments lead to important land use changes in coastal areas. Many rural coastal areas are experiencing a gradual shift from dependence on local fisheries and agriculture towards the provision of tourism services and related activities (WTTC 1993). The development of the tourism sector implies a demand for improved access along the coast in places that until recently had no basic services. Improvements in access, energy distribution and communications needed for resort development or other infrastructures, as well as prospects for employment, attract new residents to the coast. A frequent outcome is the transformation of natural (ecosystems composition) and human landscape (e.g., traditional fishing villages). These changes trigger rising prices for land, competition for resources, conflicts with sectors such as fisheries and agriculture, and may hinder the development of proper management policies.

3.4 Degradation of coastal water quality induced by land-based sources

Estuarine and coastal habitats are receiving waters for significant volumes of municipal and industrial wastewater discharges, combined with urban and agricultural runoff, and other point and non-point sources. In many estuaries there are signs that the natural dilution capacity is being exceeded by the volumes and concentration levels of effluents. Also in the estuaries, the raising levels of pollution represent an increasing public health hazard. For instance, The Global Programme of Action for the Protection of the Marine Environment from Land-based Activities from UNEP has summarized the environmental priorities for the Region as: (1) Inadequate discharge of liquid urban effluents; (2) Industrial effluents pollution; (3) Pollution related to inadequate use of agrochemical products; (4) Degradation of aquatic environments due to expansion of urban limits; (5) Inadequate disposal of urban solid residues; and (6) Activities related to extraction, transport and storage of oil or derivates (Marcovecchio 2000).

Industries dealing with horticulture and aquaculture, oil, lumber, chemicals, textiles, vehicle repairs and ship building have all added large quantities of hazardous materials to rivers, estuaries, wetlands and coastal areas, and have had major impacts on the aquatic and marine environments (Davidson 1990). The disposal of more than 87% of Municipal Wastewater in rivers, lakes, and seas create serious damage to aquatic ecosystems and implies a significant impact to public health; the enormous lack of minimum facilities for the disposal of wastewater contributes significantly to the deterioration of underground water systems, rivers and coastal environments (UNEP 2003).

3.5 Increasing coastal erosion

Deforestation, dredging and filling, poorly designed coastal structures and illegal sand mining has contributed to the increase of coastal erosion and often intensify the risk associated with coastal hazards.

4 A LOOK AT THREE CONTRASTING SITES IN SOUTH AMERICA

A significant number of estuarine areas in South America have been affected by human influence to some degree. Many of these systems show conflicting interests between urban, industrial and agricultural pressures and environmental maintenance. From heavily populated area of Santos Estuary to the near-pristine water conditions of Aysén Fjord, the sites addressed here cover a wide range of ecological and socio-economical conditions, and their inevitable conflicts and challenges in management, which can be found in South America. These systems share some similarities and also some conspicuous differences, but together they face many of the main challenges discussed above. Tables 1 and 2 contain a brief summary of the major features of each system and Figures 2 to 4 bring additional information by adding visual insights.

4.1 Santos estuary

Located at the Southern Brazilian Coast, the estuarine system of Santos comprises three major estuarine channels, namely São Vicente, Santos and Bertioga, interconnected in its inner area. Santos and São Vicente channels comprise an approximate area of 44,100 m^2, with an average depth of 15 m in the central dredged channel of Santos and 8 m in São Vicente channel. Six main rivers discharge in Santos estuary: Piaçaguera, Boturoca, Cubatão, Mogi, Quilombo e Jurubatuba. There are also many tributaries and artificial channels that collect rain drainage water and clandestine domestic waste. The tropical and subtropical climate causes high rainfall in the summer period. The Santos Estuary can be classified as a typical sub-tropical mangrove system under significant anthropogenic pressure. After hundreds of years of urban, industrial and port development, the estuary is a highly changed ecosystem. Its extensive areas of mangrove, with associated fauna and flora have been destroyed over time and are now partially degraded. The estuarine water column can be stratified, mainly due to vertical gradients of temperature. The estuarine system has a considerable ecological importance because it has a natural high productivity and is a natural habitat for many animals like birds, mammals, fish and numerous kinds of invertebrates.

The phytoplankton is dominated by diatom communities from the Genus *Skeletonema* spp. and *Thalassiosira* spp. Nanoplanktonic phytoflagellates share the dominance (alternate) with diatoms because of the adaptation to the changing light and nutrients conditions induced by the spring-neap tide cycles. Red tides caused by *Mesodinium rubrum* in the inner shelf of Santos have been reported (Moser et al. 2005), a species that is not toxic but can cause oxygen depletion problems at the end of the bloom. Other primary producer groups found in the region are the seaweeds, conspicuous in many areas in the soft substratum of mangrove forests, and *Spartina* spp, which occupy many mangrove fringe areas.

The Santos estuarine system holds the larger Brazilian harbor as well as the most important industrial complex of the Brazilian coast. The Santos estuarine complex, regarded as a polluted area (de Sousa et al. 1998), is an area heavily occupied by urban, industrial and port activities. The construction of an underground generating plant by the Light Company in the late fifties lead to the amplification of the capacity of generating energy in this area, turning possible the installation of a petroleum refinery, a petrochemical complex and, later, a metallurgical complex. These development have modified significantly the environmental and hydrodynamic conditions of the estuary. This period of fast growth during the 1950s-1960s, required amplification of the port area, and the need for continuous dredging of the main channels to allow the circulation of heavy ships in the harbor and access to areas of the upper estuary where major industries are located.

The main socio-economic drivers for the Santos estuary are the industrial and port activities, and the resident population. Currently, there are nearly 400,000 people living in this area, but the Santos, Cubatão and S. Vicente cities and adjoining region account for $\sim$1,000,000 inhabitants (almost doubled in the vacation period). The Baixada Santista region is, at the same time, a tourism, industrial and port center. Cubatão city has a remarkable industrial pole with different kinds of industries but mainly associated with the petroleum products, fertilizer production and a very remarkable steel production. The main sources of pollution at this region are: the Santos port and ships which are involved in the spillage and loss of shipped products, the industrial pole of Cubatão, the domestic waste of São Vicente and Santos cities (which are mainly discharged through an emissary), the garbage dump of the Baixada Santista region, besides the discharge of waters from the Billings water reservoir which receives used water from some parts of São Paulo City. The population accounts for the high levels of pollutants discharged in the system through sewage water. The discharges of domestic waste waters are scattered, due to the existence of only 3 WWTPs. Clandestine domestic sewage disposal arises from slum quarters at the channel margins and make up a significant contribution to the eutrophication of the system. Several industrial effluents are discharged in the inner areas of the estuary and port activities act as another anthropogenic impact in the system.

Some kinds of hazardous compounds are dispensed into the estuary such as nutrient salts, heavy metals, organic compounds and petroleum hydrocarbons. These pressures in the estuary have impacts not only in the estuary, by inducing changes in the ecologic dynamics, but also in Santos Bay and adjacent beaches. Water quality is a public health problem in this coastal zone. The main stakeholders in Santos include local government, university and educational system, industrial and harbour consortiums and NGOs (environmental protection). All the uses of the estuary make this a place of conflictive interests and uses, adding serious difficulties to its management and governance.

4.2 Bahía Blanca estuary

Bahía Blanca is a mesotidal coastal plain estuary in the southwest of the Buenos Aires Province. The main channel of this estuary has a total length of 60 km, varying in width from about 3-4 km at the mouth (22 m depth) to 200 m at the head (3 m depth). This channel is partly closed by a modified ebb delta. Three freshwater tributaries enter the estuary: the sauce Chico River (with a drainage area of about 1,620 km^2), discharging into the principal channel about 3km downstream from the head of the estuary, Saladillo (with a drainage area of 830 km^2) and the Napostá Grande Creek (with a drainage area of about 1,260 km^2) that reaches the estuary about 1km downstream of Ingeniero White Port. Both Sauce Chico and Napostá Grande rivers are originated in ”Serra de la Ventana” in the top part of the watershed.

The Bahía Blanca estuarine system shows some striking similarities with Santos estuary. The similarities are related to the intensive anthropogenic pressure and some ecological features such as the dominance of diatoms. However, there are some major differences, especially in the vertical physical structure and in the tidal range. Unlike Santos, the Bahía Blanca estuary is not stratified and has a larger tidal range. Unlike most southern hemisphere temperate systems, phytoplankton bloom takes place in winter months reaching 70 mg Chla m^{-3}. The bloom is dominated by diatoms (especially *Thalassiosira curviseriata*) with flagellates appearing only near the end of the bloom episodes. The typical phytoplankton succession starts with large diatoms, followed by dinoflagellates and ends with small diatoms. This succession suggests a complex nutrient control on primary production. Nutrient concentrations are always high inside the estuary all year round, yet there are no signs of eutrophication in the system.

The inner estuarine area receives high loads of organic matter from WWTPs, limiting light penetration in the vicinity of the discharge points and they can have a significant contribution to nutrient budgets in the upper estuary. The estuary has extensive intertidal areas with high halophytes coverage.

The main identified socio-economic drivers for the Bahía Blanca are the agricultural activity and population growth. This population growth generates a pressure over land use, impacting the water quality of the estuary, and also affecting the coastal zone. To augment production, soil fertility has been increased over the last year by fertilization. Consequently, agriculture practices have contributed to the eutrophication of the estuary through soil leaching and runoff. The city of Bahía Blanca has rapidly expanded to a total of 350,000 inhabitants over the last two decades and still has a large potential for growth.

The demographic growth has been fuelled by a large petrochemical park, fertilizer and thermoelectric plants, as well as expanding port activities since much of Argentina's export moves through Ingeniero White Port (Perillo et al. 2001). As a consequence, the discharge of industrial wastes and untreated or partially treated domestic sewage has generated increasing problems of contamination. Dredging of the Principal Channel from 9.5- to 13.5 m depths has generated over 35×10^6 m^3 of sediments, dumped on tidal flats and in off-shore locations. Dredging and deposition has introduced major changes of circulation patterns in the estuary.

4.3 Aysén fjord

The Aysén Fjord stands in striking contrast with the previous estuarine systems. Aysén is the eleventh administrative Chilean region; located between 45° and 46° S. The region is characterized by a significant oceanic climate range and vast diversity of ecosystems. There are an insular part and a continental part, with a total territory of 108,000 km^2. It is the least populated of the fourteen political regions of Chile (around 90,000 habitants) with a density of 0.8 habitants km^{-2}. Commercial activities have been primarily based on the exploitation of the region's natural resources and include: fishing and aquaculture, mining, livestock production and ranching, sawmills, agriculture and forestry.

As any fjord system, Aysén fjord has an extremely high residence time ($>$200 days) and is permanently stratified. There is a strong seasonal signal in the freshwater inputs, resulting mostly from the ice melting in the watershed and seasonal rain pattern. Aysén River is the main freshwater water input into Aysén Fjord, having a total watershed area of about 12000 km^2. Fresh water only affects the upper layer (less than 10 m deep) of the system. Primary production in the water column is dominated by diatoms and dinoflagelates. The strong oxygen production of the upper layer contrasts with the low oxygen conditions of the bottom (around 2.5 ml l^{-1}) resulting from the bacterial degradation of organic matter in the sediment. Aysén is a 300 m deep fjord and consequently the sediment is exclusively a mineralization compartment.

Given the late colonization of this area (mainly during the first half of the XX century), Aysén is still a low population region with a large governmental organization, given its geopolitical importance. Consequently, many key stakeholders correspond to governmental organizations. Salmon farmers are also key stakeholders and have been identified as the main actors of the future economic development of the area. Other primary stakeholders correspond to artisan fishermen, tourism companies, agricultures and mining companies. Although the Aysén region is often seen as a pristine and undisturbed zone in southern Chile, it has a past record of devastating human impact. Colonization during last century was accomplished using fire as a management tool; the fires were started intentionally by settlers (and supported by government policies of the time) to clear areas for cattle and sheep. Wildfires that burned throughout the 1940s have left a long-lasting mark on the watershed by changing the physical and chemical characteristics of the soils.

Nowadays, the region still holds its almost pristine state, but the estuarine system is slowing being enriched with organic matter originated in the numerous mariculture units scattered in the Fjord. Salmon farming industry is the main socio-economic driver in the Aysén Fjord. This activity, strongly supported by foreign capital investment, is characterized by having some of its industries ranking on the top ten industries nationwide and two of them worldwide. Recent investment initiatives indicate that aquaculture production in the area is in a process of expansion: by 2010 it is projected that the XI region (and Aysén county in particular) will produce 42% of the national salmonid production (up from 20% today). Besides salmon farming, the system has been used for other purposes such as mollusk harvesting and, more recently, industrial development. About 80% of the population in Aysén watershed is concentrated in the urban areas (Coyhaique, Puerto Aysén, Puerto Chacabuco). In the region of Aysén, 70% of the total habitants are connected to sewer system. This fjord receives the liquid residues of Puerto Aisén (a town of 37,000 people, located close to its head). Its also home of the only seaport for the region (Puerto Chacabuco).

5 FUTURE TRENDS

The environmental costs of regional economic expansion have been extremely high, and seem to be growing. The major issues are the accelerating over-exploitation of land and marine resources, increasing conflict over access to and use of water, loss of biodiversity and habitat degradation, urban waste disposal problems. South America interest in a solid economical development in pace with sustainable natural resource usage is expected to increase over the next few decades, stimulated mostly by new trade opportunities, changing markets, heightened awareness of coastal hazards and natural resources conflicts, and the participation in international agreements (Lemay 1998). Undoubtedly, a key challenge for the Region over the next decades will be to cope with rates of change in coastal areas, especially in estuarine systems, recognizing shortcomings of traditional approaches and building on the lessons of emerging policy reforms for integrated coastal zone management.

TABLE 1: *Major biotic and abiotic features of the three estuarine systems (see text for details).*

Feature	Santos Estuary	Bahía Blanca	Aysén Fjord
Zone	Sub-tropical	Warm temperate	Cold temperate
Mixing characteristics	Partly mixed	Well mixed	Permanently stratified
Dilution potential	Moderate	High	Low
Vegetation	Mangrove swamps High emergent vegetation Tidal flats	Spartina fringes Tidal flats Relatively low emergent vegetation	No marine vegetation
Nutrients	Nutrient exporter	Nutrient exporter	Nutrient importing from the sea
Production	Strongly heterotrophic system (mineralizing system)	Neutral (varying from auto to heterotrophic)	Strongly autotrophic system

TABLE 2: *Major socio-economical features of the three estuarine systems (see text for details).*

Feature	Santos Estuary	Bahía Blanca	Aysén Fjord
Drivers	Industrial and port activities Population growth	Agricultural activity Industrial and port activities Population growth	Salmon Farming
Population (in the area)	1,000,000	350,000	90,000
Economic activities	Petrochemical park Refineries and terminals Fertilizer plants Thermoelectric plant Metal industries Port activities	Petrochemical park Refineries and terminals Fertilizer plants Thermoelectric plant Several industries (meat and fish factories, leather and textile plants, etc.) Port activities	Salmon fish farming Artisan fishing Forestry
Pressures	Urban and industrial pollution (wastewater effluents discharges with and without treatment) Dredging	Urban and industrial pollution (wastewater effluents discharges with and without treatment) Dredging	Organic inputs (associated with fish feed and faecal pallets), sediments from terrestrial systems
Major impacts	Eutrophication Habitat degradation (loss)	Eutrophication	Local bottom modification
Human utilization of the system	Occupation (housing) Recreation (bathing in the bay area) Food source	Food source	Habitat Food source Tourism
Overall State	Highly modified Heavily Polluted	Modified Polluted	Near pristine, unpolluted
Key stakeholders	Regional government Industrial consortiums Port authorities NGOs	Regional government Industrial consortiums Port authorities	Regional and national government Salmon farmers

FIGURE 2: *The Santos Estuary is a highly changed system after decades of occupation and development. These snapshots show different areas of the estuary depicting some of the main drivers: urban pressure with an example of occupation in the frontline of the bay area (top), an industrial plant in Cubatão area (middle), located in the inner part of the estuary, and port activities (bottom). (Photos by M. Mateus)*

FIGURE 3: *The large industrial pole of Bahía Blanca (top, partial view) is a major driver in this system. Water pollution is among its most significant impacts, as seen in the middle plate, where a sign warns against toxic residues. Fishing is also a main economic activity in Bahía Blanca and the industrial port is also an harbor for both the fishing fleet (bottom) and cereal and oil/gas exportations. (Photos by M. Mateus)*

FIGURE 4: *The Aysén Fjord is in almost pristine state, maintaining much of its landscape (top) and system functioning unchanged over the recent decades of human settlement and growth in the area. The main drives in the system are the tourism, justified by the search for its esthetic value (top), fishing (middle) and salmon farming (bottom). (Photos by M. Mateus)*

A commitment towards sustainable development of the Region's estuarine resources is gradually emerging. Unreasonable demands from society in the past led to a general decline of the resources and irreversible changes in the nature of many ecosystems. Many of the natural assets of the Region have been undervalued but this trend is now being reversed. The contribution of coastal and marine areas to sustainable development is increasingly gaining recognition among public, private and political sectors in coastal states. Combined with this increasing recognition of importance is an emerging awareness of the need to manage coastal and marine resources hand-in-hand with the optimization of allocation of its uses. The region's central challenge is now to build a political consensus that will maintain stability and economic growth while addressing the growing social and environmental problems (UNEP 1999). Clearly, careful planning and management of all sectoral activities simultaneously will result in greater overall benefits than pursuing sectoral development plans independently of one another. Integrated coastal management approaches are required, combining all aspects of the human, physical and biological aspects of the coastal zone within a single management framework.

The main challenges in estuarine and coastal zone management faced by South American nations have been identified in the three study sites presented here. The remaining of this volume will deal with the work done for each site during the ECOMANAGE project and the results obtained. The research effort undertaken will be used to assist local stakeholders and decision making entities to implement adequate management strategies. This will lead, eventually, to achieve satisfactory results for the improvement of the ecological status of each site, and at the same time helping their steady and sustainable socio-economic development.

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PART B

THE METHODOLOGICAL COMPONENTS

A PHES-SYSTEM APPROACH TO COASTAL ZONE MANAGEMENT

V.H. MARÍN AND L.E. DELGADO

1 INTRODUCTION

In the past, ecological systems and human societies were analyzed independently from each other. However, in the words of Redman et al. (2004): ”The isolated study of ecological and social systems is no longer defensible”. The main reason for this is that societies and their development have influenced and modified most, if not all, ecological systems of the planet. This, in turn, has affected human societies and their development (Meadows et al. 2004). Thus, we can more properly describe today's nature as an interacting set of ecological-social systems.

For this reason, it is essential to consider human societies as reflexive components of the ecological systems upon which we depend. However, in order to apply this conceptualization a change in the dominant scientific paradigm is required. This change is necessary because in its normal way of functioning, science does not, and indeed it does not have the mechanisms to, incorporate social perception. Post-normal science is an emerging paradigm that incorporates both scientific knowledge and social perception as a requisite to understand an inter-subjective reality. Post-normal constructivism (Von Glasersfeld 1984, Jones 2002, Delgado and Marín 2005, Marín et al. 2007) proposes that knowledge cannot be understood as the image or representation of an ontologically objective reality, but rather as the organization and ordination of worlds constituted and generated in our experience. Thus, diverse social actors (including scientists) confronting the same environmental landscape will perceive different components and interactions depending on the mental models or frameworks used in the process of perception (Kolkman et al. 2005).

The present condition in South America, in relation to environmental themes, is that relevant definitions, delimitations, analysis and strategies are generated by groups of experts (technocrats), based on their investigations, publications in scientific journals and previous work on the same topic. The social context in which ecological or environmental problems are studied is seldom considered, including the way in which such problems are perceived by local actors. As a consequence, scientists tend not to be involved in a real or effective way of communicating with the socio-cultural components that impact and interact with ecological systems. Despite this technocratic dominance, important changes are taking place in the manner in which environmental problems and integrated management are perceived and analyzed. This is facilitated through the development of holistic, multidisciplinary and participative visions of science (Costanza and Jørgensen 2002, Kangas and Store 2003, Marín et al. 2007, Marín and Delgado 2007).

In this chapter we develop the concept of Physical-Ecological-Social (PHES) system as a post-normal alternative to the classical, indeed Kuhnian, ecosystem concept. We propose it as a tool to improve the communication between scientists and other social actors in relation to integrated coastal zone management.

2 ECOSYSTEMS AND PHES-SYSTEMS

Arthur Tansley proposed in 1935 a new term for the ecological sciences: the ecosystem (Tansley 1935). Whether due to a need of ecologists to find a term that would unite them below one conceptual umbrella or, as Golley (1993) proposes, the postwar enchantment with systemic sciences, the literature shows that 'the ecosystem' has been one of the most utilized ecological concepts, both inside and outside the discipline of ecology. An uncountable number of scientific articles exists that utilize the term as a geographic reference, a replacement for others ill-defined terms (e.g. region, place, space, landscape, etc.). Other articles use the term as a descriptor of an 'object'. Perhaps given the vagueness of the concept at this moment in history, some contemporary ecologists have seriously questioned its use (O'Neill 2001). The widespread use of the ecosystem concept (Golley 1993) and its multidimensional character (Pickett and Cadenasso 2002, Jax and Rozzi 2004) have resulted in a great gamma of definitions from objects in the 'real world' to abstract concepts.

The ecosystem concept, in its original formulation, only makes reference to the "system from a physical point of view"; emphasizing the interaction between biotic and abiotic components (Pickett and Cadenasso 2002). In this sense, Tansley's ecosystem is not a place on the face of the planet, but rather a non-dimensional conceptual framework. Or in the words of O'Neill (2001): "a way to observe nature". If this is the case, the ecosystem is certainly a constructivist concept where the observer plays a fundamental role. The ecosystem is generated when an observer distinguishes differences in the observed world (Haag and Kaupenjohan 2001). This process is not only dependent on the individuals involved, but also, in its most elaborated form (e.g. scientific observations), on the school to which the observer belongs. For example, ecologists trained as population-community experts will tend to observe ecosystems as comprised by interacting organisms. Ecologists trained in system approaches are likely to describe them in terms of quality and quantity of energy without a single reference to organisms.

Despite the fact that the original ecosystem concept is constructivist in nature, its use by the academy has transformed it into an object independent of the perception of the observer; one that can be defined without ambiguities (Jax and Rozzi 2004). For us, one of the most important implications of the work of Tansley (1935) is that dividing planet Earth into small parts (and labeling each part as an ecosystem) is a necessary exercise only because of our inability to study the whole interconnected totality. This continues to be true, even after more than seventy years since 1935 and the work of global scientific programs.

However, How do scientists define and delimit ecosystems? When ecological systems are the subject of study (or study units), work groups normally define components and border conditions according to the formulated questions. The system defined in this manner, which for simplicity we shall call the "*agreed ecosystem*" may or may not have the properties that theoreticians have attributed to the *theoretical ecosystem* (e.g. Jørgensen and Müller 2000, Likens 1992). For example, Likens (1992) proposed one of the most influential ecosystem conceptualizations so far, where all components and all interactions should be considered. The *agreed ecosystem* almost never contains (nor in practice could contain) all of the organisms and abiotic components in a certain area; rather it contains only those necessary to answer the questions at hand. It is for this reason, that it appears naive to think that in its current use, the ecosystem concept can provide a 'non-ambiguous' definition (*sensu* Jax and Rozzi 2004) to delineate an ecological-social entity that could be the basis for integrated, participative, environmental management. This motivated us to generate a new concept, one that would take into account post-normal conditions of integrated management (where: **facts are uncertain, values in dispute, stakes high, and decisions urgent**; Funtowicz and Ravetz 2000), the absence of which make the old ecosystem concept one of dubious use at the society and nature crossroad.

Based in a large part on the literature on perception and delimitation of systems (Müller and Leupelt 1998, Kay et al. 1999, Jørgensen and Müller 2000), on the analysis of the ecosystem concept (O'Neill 2001, Picket and Cadenasso 2002) and on discussions about post-normal science and the environment (Guba 1990, Lal et al. 2001, Marín and Delgado 2005), we propose that the social-ecological analysis of the environment, especially as it relates to its management, should be carried out on the basis of the Physical-Ecological-Social System (PHES-system) concept. A PHES-system is an observer-dependent, spatially explicit, conceptual model of an ecological system where its components and boundaries depend on the questions being addressed, the observers who formulate them and the social context in which they were proposed (Fig. 1). Thus, a PHES-system is a socially dependent conceptual model on the society-nature relations that arise as part of an integrated approach to analyzing a defined region of the planet. This concept incorporates two new characteristics in relation to previous system concepts used in environmental management: 1) human societies are explicitly incorporated as reflexive components of the system (socio-ecological component); and 2) the bio-ecological components (other species in the defined area) are only those necessary to deal with the proposed questions.

One of the main differences between the use of the ecosystem concept and the PHES-system is the methodology by which these distinctions are generated. The definition of a PHES-system begins with a series of meetings in which different visions of the environmental problem are shared, the effect of these visions on the eco-social structure of the system and its components and the valuation of the different components or processes by the actors or stakeholders; that is, an intensive process of citizen participation. This participation, in order to be effective, should improve the communication among social actors, in such a way that they may share and understand each other's visions (or PHES-systems) of the ecological system being managed. Since a PHES-system is, above all, a conceptual model, we have designed a participative methodology for their generation. This method is presented below (item 4.) and an example shown in Part D (Outcomes of the project).

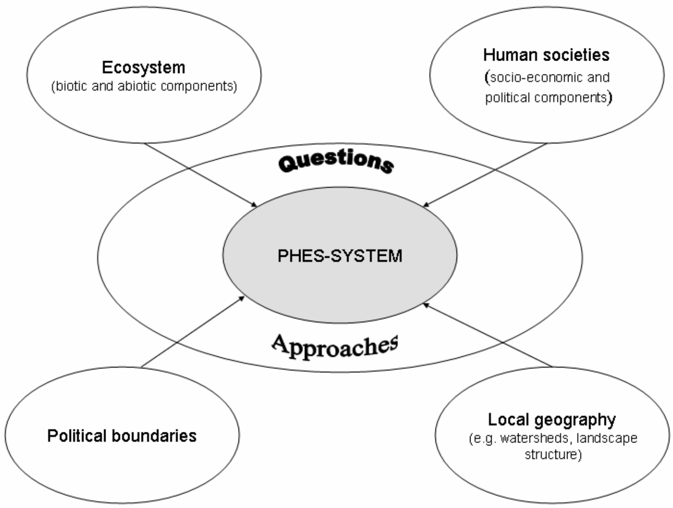

FIGURE 1: *Relationships between the contemporaneous ecosystem concept (Likens, 1992), human societies and the physical structure of the system with the PHES-system. All the information obtained about the system is filtered by the observers in relation to the questions being asked and the proposed methods and approaches to deal with them.*

3 PHES-SYSTEMS AND INTEGRATED COASTAL ZONE MANAGEMENT

Coastal zones, and the watersheds that discharge into them, are among the world's regions with greatest anthropogenic influence. It is estimated that over 50% of the global human population lives within 200 km of the oceans' coasts. This dominant and multifaceted presence of human activity has stimulated the development of concepts and strategies aimed at the integrated and harmonic management of these areas. Integrated Coastal Zone Management (ICZM) has been defined as: "A dynamic process in which a coordinated strategy is developed and implemented in order to establish multiple environmental, socio-cultural, and sustainable uses of the coastal zone" (CAMPNET 1989). Afterward, the World Bank, in a document entitled "Guidelines for Integrated Coastal Zone Management" (Post and Lundin 1996), emphasized that the ICZM should consider the activities of all sectors that affect the coast, including economic, social, environmental, and ecological aspects. Other approaches (e.g. DPSIR) reduce society only to economic impacts, without considering other interactions between them and the environment. Incorporating the perception of other social actors (non-scientists) the description of pressures and effects can, for example, incorporate the historical relationships between societal and ecosystem changes (Saez et al., submitted).

The inclusion of ICZM as one of the main recommendations of Agenda 21, generated during the Rio Earth Summit (United Nations Conference on Environment and Development, Río de Janeiro, 1992), prompted the international acceptation and political prominence of the concept.

One of the requisites for the development of ICZM is integrated economic, social, and ecological knowledge of the coastal zone, and identification of the main components of the corresponding sub-systems, as well as the interactions between them. From an ecological perspective, this form of generating knowledge is an integral part of the ecosystem management approach. This approach emphasizes the analysis of biotic and abiotic components of ecological systems and their interactions. However, in its classical form, this approach does not consider humans as components (O'Neill 2001). Furthermore, since ecosystems definition and delimitation are based only on expert's advice, it does not fulfill one of the main recommendations of Agenda 21: that "*the best way to deal with environmental issues is with the participation of all concerned citizens, at its appropriate level*". Consequently, we propose that the PHES-system concept and its related participatory approaches facilitate several of the requirements of the ICZM such as: the inclusion of human societies as components of coastal zones, the integrated analysis of eco-social components and the use of conceptual models as the basis of narration of the distinct visions held by the social actors that participate in management.

Integrated management is based on a series of elements that have the objective of gathering and articulating a diversity of facets such as plans and programs for economic development. This process considers the coordination, from a multidisciplinary perspective, of conservation and management activities as well as the use of ecosystem services in a spatially defined area (watershed, fjord, forest) with the objective of maximizing the social and economic benefits in the most equitable way. The focus is on the integration of the responsibilities of different levels of government (local, regional, and national), between the private and public sector, incorporating all of the actors in the diverse array of aspects of management (Salomons et al. 1999). This management should include an adaptive vision, that is, the policies, plans, and programs should be evaluated and modified periodically, in a gradual way according to the reality of the situation in which they are being applied. In this way, the management programs can be undertaken and incorporated by all of the actors of the ecosystem, with the goal of achieving a better and more efficient development of the programs applied to the sustainable use of the ecosystems (Sabine et al. 2004). However, if the involved actors do not share the same model of the ecological system, then its integrated management (especially if citizen's participation is empowered) is very difficult. If we accept that each group of social actors involved in the management of an ecological system has its own perspective of it (several PHES-systems of one ecological system), then it is necessary to be able to share these visions in order to arrive at a consensual and integrated management that has a chance of success. Conceptual participative modelling is one of the ways in which such visualization can be carried out (Redman et al. 2004, Marín et al. 2007). This is especially appropriate if the goal is to incorporate models derived from local non-scientific knowledge. Both the techniques of cognitive mapping (Özesmi and Özesmi 2004) and participative modelling (Heemskerk et al. 2003, Marín et al. 2007) have been used successfully in the generation of ecological-social models. The PHES-systemic vision of ICZM is, therefore, an invitation to incorporate human societies as components of that which we manage. Furthermore, it is a proposal to consider as relevant the diverse visions of distinct social actors and that these visions are considered at the time of implementing environmental management plans. Otherwise, we may end up caught in the ”parable of the elephant” (Marín and Delgado, in press).

4 PARTICIPATIVE MODELLING

4.1 On the need of social actor’s participation

Participative modelling, within the context of a PHES-system approach to integrated management, is a way to make explicit the perception that a group of social actors has on the ecological system being managed. Why is this necessary? Because as Waltner-Toews et al. (2003) argue: ”perspective changes everything” and thus, ecosystems should be managed ”inside out”. However, in order to do that: societies have to be incorporated within the system and we should be able to share all visions. One of the problems we have encountered, when using other conceptual modelling approaches (e.g. focus groups), is that models end up having the imprint of the ”dominant scientist” (sensu Bordieu 2003). Thus, we generated an approach to the conceptual modelling of ecological system named: A brainstorming strategy to conceptual PHES-system modelling (Marín et al. 2007). In what follows we describe this approach. Its main objective is to generate an uncritical environment in order to reduce inhibitions so people may express their points of view in a more open manner.

4.2 A brainstorming strategy for conceptual modelling

Conceptual models have proved to be effective communication tools, especially when dealing with complex systems (Heemskerk et al. 2003, Marín and Delgado 2007, van den Belt 2004). However, in highly hierarchical settings (e.g. students-professors, landowners-peasants, general managers-employees) inhibition is highly likely. Nicolson et al. (2002), working to develop heuristics for interdisciplinary modelling, point out that the task of communicating with stakeholders or social actors is vastly underrepresented in many scientific projects. ECOMANAGE was a project based on a strong interaction between scientists and local stakeholders. Thus, from the beginning there was a need to generate innovative participatory methods so we could build the conceptual model of the studied ecosystems incorporating the perceptions of all people involved.

Figure 2 shows a flow diagram of the main steps followed to generate conceptual models in interdisciplinary, hierarchical, experts/non-experts settings. Although the interested reader may find many books on the subject, we would recommend two Internet sites that ended up being very useful in designing our sessions. The process is initiated by sending questions to the potential participants of the modelling session. The development of these questions should be based on: (a) analysis of current private and state development projects which may have an impact on the ecosystems being modelled, (b) current environmental national, local, legislation, and (c) the state of the ecological systems. Although workshop conveners may restrict themselves to the use of the submitted questions during the brainstorming session, their main aim is to provide participants with a global idea of the reason why they have been invited.

In a brainstorming session, narrators play a very specific role: recording all statements of participants without making comments or including extra material. Furthermore, since the ultimate goal is to generate a PHES-system, an extra requirement is to have working knowledge of an iconographic modelling software that may be used to generate the conceptual models (e.g. Stella or Vensim).

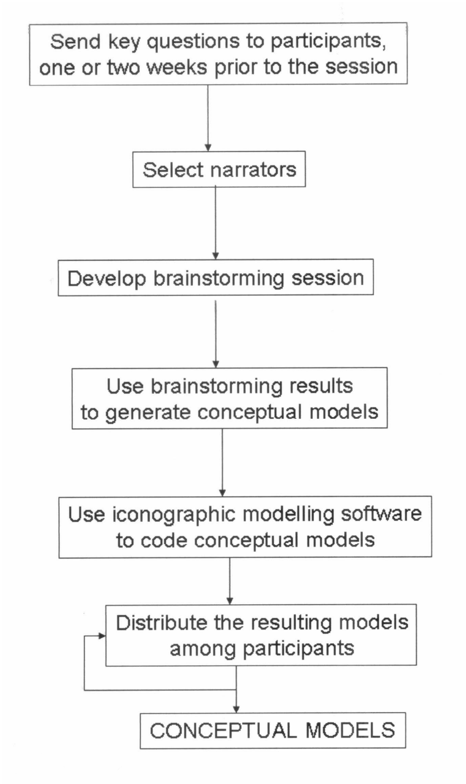

FIGURE 2: *Sequential steps for the generation of conceptual PHES-system models using brainstorming techniques.*

There are several ways to develop a brainstorming session, most of which are fully described in the web pages cited above. Among the alternatives (unstructured or free flow, silence-written, structured or in circle), we recommend the later. The reason for this is simple, most of the time the invitees will be new to this sort of sessions; thus, any other format could inhibit the participation of some of the members (in the case of unstructured format) or prevent the building of ideas based on other member's comments (in the case of silence format). Although all the details given in the previous paragraphs are important to take into consideration in a PHES-system modelling session, there is one which is of the utmost importance: At the beginning of the session the convener should explicitly read the four basic rules of brainstorming:

1. Postpone and withhold judgment of ideas
2. Concentrate on quantity rather than quality
3. Encourage the building upon ideas put forward by other participants
4. Every person and every idea has equal worth

The first rule, out of the four, is by far the most important. Brainstorming techniques are successful only to the extent that through them it is possible to generate the necessary uncritical environment, which should encourage an open exchange of ideas.

Once the brainstorming session is over, which should take between 1.5 to 2.0 hours maximum, the convener and narrators should end with a full record of the ideas of all stakeholders involved. The next step is then to generate the conceptual model based on the ideas gathered during the brainstorming session. Although is not necessary to use iconographic modelling software (see for example Heemskerk et al. 2003), experience show that it is a useful approach to modelling (van den Belt 2004, Marín et al. 2007).

The result of the previously described process should be a conceptual model of the ecosystem (a PHES-system) from the point of view of a group of stakeholders. If analyzed along with the notes and transcripts of the brainstorming session, ecosystem managers will have a better perspective of the way stakeholders see and interact with the ecological system. When many PHES-systems are generated, modelers may attempt to join them into a single model that represents the common ecosystem perception of the stakeholders. This can then be used as a powerful management tool. However, there may be some cases where one or two PHES-systems will simply be non-compatible. That is, perceptions may be so divergent that it is not possible to put them together. In those cases modelers should warn managers that they may be facing potential conflicts among stakeholders and act accordingly.

Integrated coastal zone management should be, above all, a coordinated effort to allocate multiple uses in areas close to the coastline. In order to attain this goal, it is necessary to incorporate the visions and perceptions of social actors. The PHES-system approach we have described here was conceived with that objective in mind. Further along this book we show examples of its use within the context of the ECOMANAGE project.

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DEFINITION OF STATE INDICATORS FOR THE MANAGEMENT OF INTERACTIONS BETWEEN INLAND AND ESTUARINE SYSTEMS

T.E. LEITÃO

1 INTRODUCTION

This chapter briefly describes the methodology used to define State indicators used in the management of estuarine systems during the course of the ECOMANAGE project.

An indicator is defined as ”a statistic or measure, which facilitates interpretation and judgment about the condition or an element of the world or society in relation to a standard or goal” (USEPA 1995). ”Indicators help to reflect and communicate a complex idea [. . .]. We use them to observe, describe, and evaluate actual states, to formulate desired states or to compare an actual with a desired state. These simple numbers, descriptive or normative statements can condense the enormous complexity of the world around us into a manageable amount of meaningful information. [. . .] It aims to communicate information on the system or process. The dominant criterion behind an indicator's specification is scientific knowledge and judgment” (WWDR 2003). Within the water-related field, indicators must be used to manage and systemize information due to the large amount of data available and the increasing complexity of policy problems, providing information for political decision-making processes, and also information for the public.

The type of indicators hereinafter proposed for ECOMANAGE project are ”basic indicators”, defined in Chapter 1 of the WWDR (2006) as to ”provide fundamental information not directly linked to policy goals (e.g. water resources, GNP and population), well established, widely used and corresponding to data generally widely available around the world”. The use of an indicator depends on the objectives for which they are used. In the following sections, a proposal for ECOMANAGE indicators will be explained based on the 3 project case-study areas needs.

2 OBJECTIVES

The use of indicators within ECOMANAGE Project has the objective of connecting the driving forces of the watershed activities to the state of coastal ecosystems and resources, through pressures and pathways, using the conceptual index framework of DPSIR (driver-pressure-state-impact-response) initially proposed by OECD (1993).The indicators proposed serve two main purposes within the Project: (1) reflect the present state of waters (inland and estuary) due to the existing pressures; and (2) evaluate effects deriving from new scenarios to be modelled regarding various environmental management measures of land use - land cover changes. Their final purpose is to help choosing the watershed scenarios that better contribute to a sustainability integrated management of the selected coastal zones.

The indicators in this project are '**state**' **indicators** (DP**S**IR), meaning indicators that describe the actual State conditions of the systems. These indicators are divided in quantity and quality, and are proposed for the 3 media studied: groundwater, surface water and estuary water. Sonak et al. (2003) refer a set of other type of indicators for coastal ecosystems analysis, also using the DPSIR framework such as: 'Pressure' indicators, which refer to the stresses created by different drivers on the coastal area; and 'Impact' indicators which can be described by the effect on coastal ecosystems and resources as reflected by changes in the characteristics of the ecosystem. However, this type of indicators - 'Pressure' and 'Impact' - is not envisaged to be used for ECOMANAGE project.

Since ECOMANAGE project scenarios are centered in land use - land cover changes, 'state' indicators directly attempt to reflect the main effects of these changes, both in the watershed (groundwater and surface water) and the estuary waters:

- In the case of Santos estuary (Brazil), these aspects concern essentially the social changes: movement, increase and density of population and their effects in water demand, water pollution, and the sustainability of mangrove.
- In the case of Bahía Blanca estuary (Argentina), the analysis is focused on the changes in land use for agriculture and for livestock, as well as on the placement of the city sewage treatment plant.
- In the case of fiord Aysén (Chile), the analysis is centred on the effects of intensifying and extending the salmon fish farming for several scenarios.

3 INDICATORS PROPOSAL

3.1 Quantity indicators

'State' quantity indicators are defined only for inland waters, as the changes in pressures are not expected to be reflected in the quantity of the water in the estuary (although their effects can be reflected in the quality aspects, e.g. changes in the salinity due to freshwater discharge decrease caused by higher freshwater demand, as later referred to in the quality indicators). Quantity indicators will assess the changes in water amount due to the different scenarios of pressures for each case-study.

For groundwater, the quantity 'state' indicator proposed is the piezometric level. Additional pumping, either due to scenarios of growth of population consumption or to agriculture demands, will result in a decrease in this level. Note that if the source for irrigation water comes from surface water, it is possible that an increase in the groundwater level is observed.

A more complex form of assessing the changes in the equilibrium between groundwater and sea-water due to new pressures caused by water demand, i.e. aquifer vulnerability to sea-water intrusion in coastal aquifers (that may affect the some mangrove health due to changes in the salinity) is GALDIT index (originally done in the framework of the EU-India INCO-DEV COASTIN project, Chachadi and Lobo Ferreira 2005). The indexes congregate a set of indicators which describe the most important factors controlling seawater intrusion: **G**roundwater occurrence (aquifer type; unconfined, confined and leaky confined); **A**quifer hydraulic conductivity; Depth to groundwater **L**evel above the sea; **D**istance from the shore (distance inland perpendicular from shoreline); **I**mpact of existing status of sea water intrusion in the area; and **T**hickness of the aquifer, which is being mapped. The acronym GALDIT is formed from the highlighted letters of the parameters for ease of reference. These factors, in combination, are determined to include the basic requirements needed to assess the general seawater intrusion potential of each hydrogeologic setting. GALDIT factors represent measurable parameters for which data are generally available from a variety of sources without detailed examination. A numerical ranking system to assess seawater intrusion potential in hydrogeologic settings has been devised using GALDIT factors. The system contains three significant parts: weights, ranges, and ratings. Each GALDIT factor has been evaluated with respect to the other to determine the relative importance of each factor.

For surface water, the chosen indicator for ECOMANAGE project is the flowrate discharge of rivers into the estuaries. Flowrate changes reflect the seasonal availability, the climatic changes and also reflect the extra demands (Pressures) due to urban and agriculture activities. A decrease in the flowrate will be expected in scenarios of population and agriculture growth. Moreover, the water quality is expected to be worsening due to these activities, as can be seen in the next section. Salt water intrusion can also affect the possibility of using surface fresh water of upstream reservoirs.

3.2 Quality indicators

A 'State' quality indicator helps assessing the current status of water quality (due to existing pressures) and to further weigh up the effects of the different scenarios that are modelled. As stated before, the scenarios for the 3 case-study areas are focused on changes due to population changes, to land use changes due to agriculture and livestock, and to changes in fish farming. All these activities have analogous effects in what concerns the main type of pollutants associated to them. Therefore, the selection of common indicators is possible and, therefore, hereinafter proposed. The main types of pollutants expected are described in Table 1, based on some indications by Chapman (1996).

From the main type of pollutants referred, the following indicators are proposed:

- Faecal and other pathogens coming from livestock and human waste; Indicator: *Escherichia coli* (*E. Coli*).
- Nutrients such as nitrogen and phosphorus coming from fertilizers, manures, and sewage; Indicator: nitrate (NO_3^-), ammonium (NH_4^+) and phosphate (PO_4^{3-}).
- Organic wastes coming from slurries, silage liquor, surplus crops, and sewage sludge.

Indicator: chemical oxygen demand (COD) or biological oxygen demand (BOD) or dissolved oxygen (DO) or oxygen (O_2%).

- Heavy metals have little expression in this type of pollution, and therefore no indicators are proposed.
- Salinity indicates the freshwater and salt water interactions; Indicator: electrical conductivity (EC) or salinity and clorinity (Cl^-).
- Total suspended solids (TSS) is also an important indicator of loads, i.e. soil particles (resulting in suspended solids) coming from farming, upland erosion, forestry, urban areas and construction and demolition sites.
- Other indicators of the water quality status: pH, Silica (Si), Chlorophyll-a (Chl-a), benthic oxygen demand, Secchi depth (the depth of the water where the disk vanishes and reappears).

Urban, agriculture and livestock can also led to other important indicators that were not chosen for this project since their analyses are frequently unavailable. They refer to pesticides, veterinary medicines, biocides, and endocrine-disrupting substances (particularly estrogenic steroids deriving from human contraceptive pills, linked to feminization of male fish).

TABLE 1: *ECOMANAGE project activities potentially causing water pollution and main type of pollutants associated. Category legend: P - Point, D - Diffuse and L - Line.*

		Main type of pollutants				
Activity	**Category**	**Faecal pathogens**	**Nutrients**	**Organic micropollutants**	**Heavy metals**	**Salinity**
Urbanisation						
Unsewered sanitation	P-D	x	x	x		x
Land discharge of sewage	P-D	x	x	x		x
Stream discharge of sewage	P-L	x	x	x		x
Sewage oxidation lagoons	P	x	x	x		x
Sewer leakage	P-L	x	x	x		x
Landfill, solid waste disposal	P		x	x	x	x
Highway drainage soak-aways	P-L			x	x	x*
Wellhead contamination	P	x	x	x		
Agricultural						
Fertilisers	D		x	x		
Irrigation	D		x	x		x
Sludge and slurry	D	x	x	x		x
Wastewater irrigation	D	x	x	x		x
Livestock rearing/crop processing						
Unlined effluent lagoons	P	x	x	x		
Land discharge of effluent	P-D	x	x	x		x
Fish farming	P-D	x	x	x		x
Stream discharge of effluent	P-L	x	x	x		

*In countries where de-icing procedures are applied.

3.3 Synthesis of the State indicators proposed

Table 2 presents a synthesis of the 'state' indicators proposed for ECOMANAGE project, based on the 'pressure' scenarios that will be modelled for each case-study site, as referred in previous sections.

As can be seen in Table 2, when feasible, ECOMANAGE indicators are the same in the three media: groundwater, surface water and estuary. Common indicators between inland and estuary are essential to help integrating the effects of the different watershed pressure scenarios in the estuary water quality and quantity. However, in same cases it is not reasonable to have the same indicators: TSS is a meaningless measure in groundwater since the geological material acts as a natural filter that reduces its value; phosphorus and ammonia are usually not a problem in groundwater quality due to its strong retention into the soil particles; etc. Also the measurement of oxygen content in water is typically different for each media; so the indicators proposed for each media reflect the most common expression used, corresponding to the data more widely available.

The indicators referred in Table 2 must be measured in different temporal and spatial scales to supply useful information. So, for example surface flow discharges must be given in terms of seasonal and annual discharges, as well as groundwater discharge rates; oxygen and most other quality parameters must be measured for the different seasons, considering not only several points in the area but also their profile in the water, especially for the case of estuaries.

TABLE 2: *Synthesis of the 'state' indicators for ECOMANAGE project.*

<table>
<tr><th rowspan="2">Case study</th><th rowspan="2">Pressure</th><th colspan="3">'State' quantity indicators</th><th colspan="3">'State' quality indicators</th></tr>
<tr><th>Groundwater</th><th>Surface fresh water</th><th>Estuary</th><th>Groundwater</th><th>Surface fresh water</th><th>Estuary</th></tr>
<tr><td>Aysén Fjord</td><td>Fish farming</td><td>-</td><td>Flowrate</td><td>-</td><td rowspan="3">pH, E. Coli, NO_3^-, DO, COD, EC, Cl^-</td><td rowspan="3">E. Coli, NO_3^-, NH_4^+, PO_4^{3-}, BOD, O_2, EC, Cl, TSS, Si</td><td rowspan="3">E. Coli, NO_3^-, NH_4^+, PO_4^{3-}, O_2, salinity, TSS, Si, Chl-a, Benthic oxygen demand, Secchi depth</td></tr>
<tr><td>Bahía Blanca</td><td>Agriculture + population</td><td>Piezometric level</td><td>Flowrate</td><td>-</td></tr>
<tr><td>Santos Estuary</td><td>Population + sewage</td><td>Piezometric level</td><td>Flowrate</td><td>-</td></tr>
</table>

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MODELLING COASTAL SYSTEMS: THE MOHID WATER NUMERICAL LAB

P.C. LEITÃO, M. MATEUS, F. BRAUNSCHWEIG, L. FERNANDES AND R. NEVES

1 THE MOHID MODELLING SYSTEM

MOHID Water is a numerical model included in MOHID Water Modelling System (Braunschweig et al. 2004), an integrated water modelling software that can be used to simulate water bodies, porous media flow and infiltration, and watersheds (http://www.mohid.com). Over the past years MOHID Water has been used to simulate a variety of processes and scales in marine systems. This chapter presents a brief record of these applications, along with a description of the transport processes simulated by the MOHID system and its modelling philosophy. MOHID Water is the latest version of MOHID long set of evolutions which started back in 1985. Since then, a continuous development effort of new features has been maintained. Model updates and improvements have been made available on a regular basis and used in the framework of many research and engineering projects. Initially, MOHID was a two-dimensional tidal model written in FORTRAN 77 (Neves 1985). This version also gave the present name to the model, which derives from the Portuguese abbreviation of "MOdelo HIDrodinámico" (Hydrodynamic Model). Traditionally known as a hydrodynamic model, it was first used to study estuaries and coastal areas using a classical finite-differences approach. Further developments included a 3D setup and the addition of baroclinic effects (Santos 1995), and full discretization to a finite volumes approach, allowing the use of generic vertical coordinates (Martins 2000).

A substantial increase in the number of users has occurred since the model was made available on Internet, backed up by an online user forum. Model robustness in hydrodynamics set the basis for the development and coupling of a transport model, including fine sediment transport (Cancino and Neves 1999). This development also allowed the coupling of a water quality (eutrophication) module (Portela 1996, Miranda 1999, Pina 2001, Saraiva et al. 2007) which increased the variety of model applications and transformed the model into a fully integrated tool. In time, the increase of MOHID programmers and users proved to be unsustainable due to the multidisciplinary nature of the endeavour and to FORTRAN 77 language limitations. So it was necessary to establish a methodology which allowed reusing the code systematically and improving its robustness. The model was restructured and converted to ANSI FORTRAN 95, profiting from its new features such as the ability to use object oriented programming methods. This migration began in 1998, implementing object oriented features as described by Decyk et al. (1997) with significant changes in code organization (Miranda et al 2000), leading to an object oriented model for surface water bodies which integrates different scales and processes (Leitão 2003). The object oriented strategy proved to be reliable and robust, though it has increased the code and the execution time twofold or threefold, depending on the nature of the applications (Miranda et al. 2000). Presently the MOHID development is a relatively straightforward task due to the use of this philosophy.

2 APPLICATION EXAMPLES

MOHID Water has been applied in numerous studies, integrating a wide variety of processes and scales. Among the recent applications to marine systems we find:

- Estuaries: Sado estuary, Portugal (Martins et al. 2001); Tagus estuary, Portugal (Leitão et al. 2003, Braunschweig et al. 2003, Mateus 2006); Guadiana, Portugal (Saraiva et al. 2007);
- Coastal lagoons: Ria de Aveiro (Trancoso et al. 2005, Vaz et al. 2007); Ria Formosa (Silva et al. 2002), Óbidos (Silva et al. 2005);
- Coastal areas: Ria de Pontevedra; Spain (Villarreal et al. 2002), Brazilian Coast (Leitão et al. 2004), Nazare Canyon, Portugal; Galician coast (Carracedo et al. 2006, Fig. 1);
- Oceans: Cadiz Gulf, Spain (Leitão et al. 2005), Iberian West Coast (Coelho et al. 2002, Santos et al. 2005).

Some of these modelling studies focused only on hydrodynamic processes: Cadiz gulf circulation (Leitão et al. 2005); Poleward current (Coelho et al. 2002); Barotropic 3D flows in an estuary (Martins et al. 2001) and in a costal lagoon (Silva et al. 2002); Baroclinic 3D flow in an estuary (Leitão 2003) and in a Galician Ria (Villarreal et al. 2002); Waves and currents interaction effect on the sea level in a coastal lagoon (Silva et al. 2005); Non hydrostatic processes associated with internal waves (Theias 2005). Some studies were also focused on the dynamics of fine sediment: In the Western Scheldt and Gironde estuaries (Cancino and Neves 1999); Dredge material contaminated with release in a coastal area off-shore of the Santos estuary (Leitão et al. 2004); The effect of internal tides on fine sediment transports in the Nazare Canyon, located on the Portuguese coast. Other studies focused on water quality issues: The Prestige oil spill (Carracedo et al. 2006); Influence of nutrient loads in Portuguese estuaries (Saraiva et al. 2007); Modelling of Macroalgae in a shallow temperate estuary (Trancoso et al. 2005); Modelling phytoplankton dynamics in the Tagus estuary, Portugal (Mateus 2006); A methodology to estimate renewal time scales in estuaries applied to the Tagus Estuary case (Braunschweig et al. 2003).

3 TRANSPORT PROCESSES

Transport processes play a key role in marine environments and so one main goal of MOHID is to simulate them accurately. This modelling system simulates the transport processes of momentum, mass and heat in the water column, and the vertical transport of mass in the sediment column. Mass can be transported in the dissolved phase and the particulate phase, both in the water and the sediment columns (Figure 2). The particulate matter tends to be adsorbed to the fine sediments. The fine sediments settle at the water-sediment interface (fluff layer). They can be eroded or undergo a consolidation process; if they are consolidated the adsorbed particulate matter can be transferred to the dissolved phase and be dispersed in the sediment column by diffusion processes (Figure 2). The accurate simulation of hydrody-

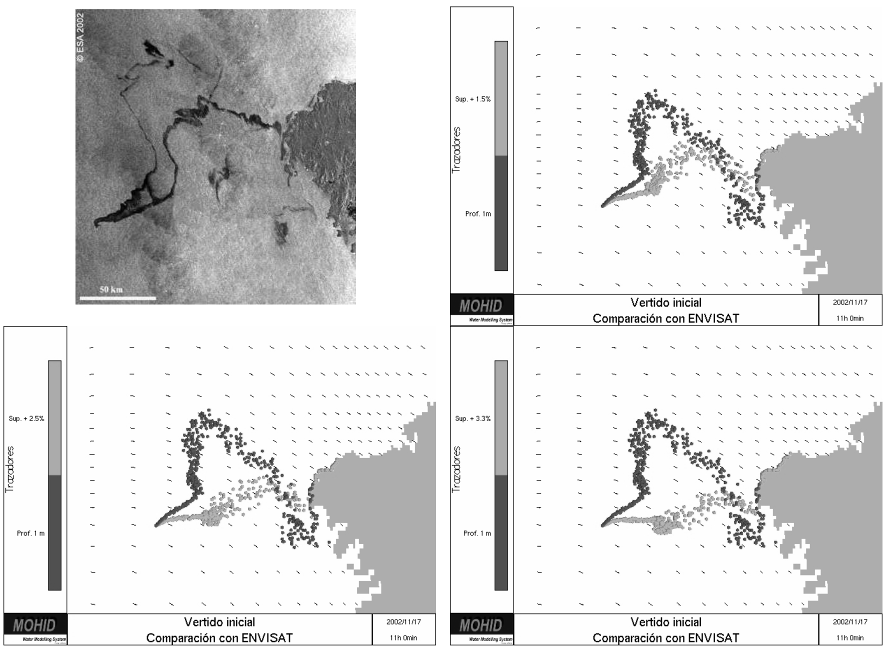

FIGURE 1: *Lagrangian particles are released at different depths: surface particles (dark grey), 1 m depth particles (light grey). Surface particles are directly dragged by wind with a velocity proportional to the wind velocity in a percentage of (b) 1.5%, (c) 2.5%, (d) 3.3%. Results are compared with an ENVISAT satellite image (a) taken on November 17th (adapted from Carracedo et al. 2006).*

3.1 Eulerian referential

MOHID is able to simulate in an Eulerian referential (fixed grid) the transport processes of momentum, mass and heat in the water column. The evolution of the non-turbulent flow properties (hydrodynamics) is computed using the Navier-Stokes equations for a rotating fluid. The geophysical fluid is constrained to the hydrostatic and the Boussinesq approximations, as a practical result of dimensional analysis. The spatial discretization is done using a finite-volumes approach (Martins et al. 2001) similar to Chu and Fan's (2004) method. MOHID also solves a seawater density non-linear state equation, depending on pressure, salinity and potential temperature using the algorithm of Millero and Poisson (1981). The system uses a structured grid: an Arakawa C grid type in the horizontal and a generic vertical coordinate with the possibility to choose different types of discretizations (e.g. z-level, sigma and double-sigma coordinates). A "partial step" approach is recommended for bottom layer discretization for z-level vertical discretization in 3D models. This methodology is better than the traditional "full step" or "staircase" approach. Adcroft et al. (1997) show that this approach minimizes the traditional problems associated with the staircase topography of z-level models ("staircase noise"). The baroclinic pressure gradient term is always calculated using a z-level approach, with a linear interpolation, to minimize spurious pressure-gradients (Kliem and Pietrzak 1999).

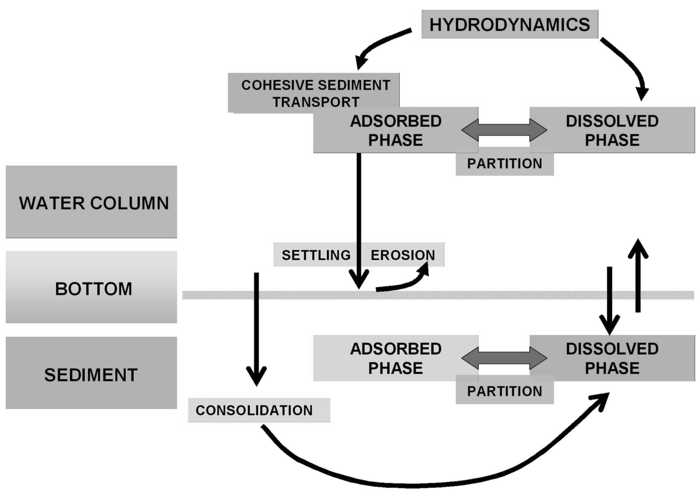

FIGURE 2: *Transport processes model by MOHID.*

The temporal discretization is done using an alternate direction semi-implicit (ADI) method for the 2D mass balance equation (used to compute the SSH). For the 3D momentum (zonal and meridional velocities), heat and salt balance equations in the vertical direction are computed implicitly while the horizontal directions are calculated explicitly. The advection of momentum, heat and salt is computed using a total variation diminishing (TVD) scheme with a Superbee limiter. A biharmonic filter for the velocities is used to dissipate high frequency noise in applications where the dissipation rate is low (e.g. open ocean applications). The advantage of this methodology, relatively to the Fickian diffusion, lies in its ability to dissipate the high-frequency processes without significantly changing the lower frequency processes. To calculate the turbulent vertical mixing, the GOTM (Burchard 2002) code is embedded in MOHID. The parametrization proposed by Canuto et al. (2001) is used by default in the MOHID system. Finally, the hydrodynamic model can be forced with tide, momentum and atmospheric heat fluxes, wind waves and fresh water discharges.

3.2 Open boundary

To prescribe coherent open boundary conditions (OBC), good external data are mandatory (Blayo and Debreu 2005). There are several sources of external solutions for coastal applications. Several global tidal solutions became very common approximately 15 years ago (e.g. CSR4, FES2004, GOT00.2, NAO. 99b, TPXO6.2). The MOHID system has the necessary software tools to generate the external solution from the FES2004 tidal SSH atlases (Lyard et al. 2006). Pre-operational models have been made available over the last years (Mercator, HYCOM-US, Topaz and FOAM), providing a best estimate on the current state of the ocean low frequency processes. The MOHID system also has the necessary software to use the Mercator and HYCOM-US for external solutions and initial conditions (Leitão et al. 2005). The MOHID system allows the user to construct a tree of one-way nested models with no limitations on the number of nesting levels from a software perspective (Braunschweig et al. 2004). By default, for each nesting level the external data for the OBC is the upper level in the MOHID nesting system. However, the user can add another solution linearly to the upper nesting levels. This nesting capability allows overlapping different scales in an efficient way to study local processes.

3.3 Lagrangian referential

Mohid can simulate the transport of mass (dissolved and particulate) in the water column using a lagrangian approach (Leitão 1996). The velocities of lagrangian particles at any point in space are calculated with a linear interpolation between the points of the hydrodynamic model grid. Turbulent transport is responsible for dispersion. The effect of eddies on particles depends on the ratio between eddies and particle size. Eddies bigger than the particles make them move at random. On the other hand, eddies smaller than the particles cause entrainment of matter into the particle, increasing its volume and mass according to the environment concentration. The random movement is calculated following the procedure proposed by Allen (1982). The random displacement is calculated using the mixing length and the standard deviation of the turbulent velocity component, as given by the turbulence closure of the hydrodynamic model. Particles retain the velocity during the necessary time to perform the random movement, which is dependent on the local turbulent mixing length (Leitão 1996). It is also possible to associate to the particle the main processes of the fine sediments dynamics described below: settling velocity, adsorption/desorption, deposition and erosion.

3.4 Fine sediments in the water column

Particulate properties transported in the water column are governed by a 3D advection-diffusion equation where the vertical advection includes the particle settling velocity. Two different approaches are followed to compute settling: a constant settling velocity and a settling velocity dependent of fine sediment concentration. In the first case, each particulate can have its specific and constant settling velocity, which can be derived from literature (depending on its size and biogeochemical characteristics). The latter approach, however, needs some considerations. Since the settling velocity algorithm was developed for fine sediment modelling, it raises the question of how the settling velocity of particulates with other properties can be computed. The model assumes the same velocity as for the fine sediment settling velocity, therefore reinforcing the importance of fine sediments in the distribution and fate of the adsorbed contaminants fraction. The algorithm follows a formulation widely used in literature (e.g. Mehta 1988), where the general correlations for the settling velocity in the flocculation range are:

$$W_S = K_1 C^m \quad for \quad C < C_{HS} \tag{1}$$

and in the hindered settling range:

$$W_S = K_1 C_{HS}^m \left[1.0 - K_2 \left(C - C_{HS}\right)\right]^{m_1} \quad for \quad C > C_{HS} \tag{2}$$

where W_S ($\mathrm{m\ s^{-1}}$) is the settling velocity, C ($\mathrm{kg\ m^{-3}}$) is the concentration, and the subscript HS refers to the onset of the hindered settling (of about 2 to 5 $\mathrm{kg\ m^{-3}}$). The coefficients K_1 ($\mathrm{m^4\ kg^{-1}\ s^{-1}}$) and K_2 ($\mathrm{m^3\ kg^{-1}}$) depend on the mineralogy of the mud and the exponents m and m1 depend on particle size and shape.

3.4.1 Adsorption/Desorption

Adsorption and desorption are considered as a reaction process, that can be included in the sinks and sources terms of the transport equation. This reaction involves the dissolved and the particulate phases of the contaminant being simulated, where the two phases tend to an equilibrium, which is given by a partition coefficient. The equilibrium can be described by the following system of equations (Hayter and Pakala 1989):

$$\frac{\partial C_d}{\partial t} = k\left(D\% \times C_p - P\% \times C_d\right) \tag{3}$$

$$\frac{\partial C_p}{\partial t} = k\left(P\% \times C_d - D\% \times C_p\right) \tag{4}$$

where C_p and C_d are the particulate and dissolved contaminant concentrations respectively; k ($\mathrm{s^{-1}}$) is the equilibrium kinetic rate for adsorption-desorption between dissolved and particulate phase; D% is the dissolved contaminant fraction; and P% the particulate contaminant fraction. The kinetic constant defines the rate at which the two phases tend to equilibrium. To account for the fact that, in the presence of low suspended matter concentrations, the adsorption process is less likely to occur (the probability of a contaminant ion to collide with a particle is lower), a direct relation between the kinetic rate and the suspended particulate matter was implemented:

$$\begin{cases} k = k_{ref} \cdot \frac{C_{SPM}}{C_{SPMreference}} & for \quad \frac{C_{SPM}}{C_{SPMreference}} < 1 \\ k = k_{ref} & for \quad \frac{C_{SPM}}{C_{SPMreference}} \geq 1 \end{cases} \tag{5}$$

where C_{SPM} is the concentration of the suspended particulate matter approximately equal to the fine sediment concentration. The k_{ref} is the kinetic rate for a C_{SPM} of reference ($C_{SPMreference}$).

3.5 Water-sediment interface model

The water sediment interface model also known as fluff layer computes and manages boundary conditions for the water column and sediment compartments.

3.5.1 Fine sediments fluxes

For fine sediments at the bottom, a flux term, F_b (mass of sediment per unit bed area per unit time) can be defined, corresponding to a source or sink for the suspended particulate matter in conditions of erosion or deposition, respectively. Consequently, at the bottom:

$$F_b = F_E - F_D \tag{6}$$

where F_E and F_D are respectively the erosion and deposition fluxes. It is assumed that, when bottom shear stress is smaller than a critical value for deposition, there is addition of matter to the bottom, and, when the bottom shear is higher than a critical value, erosion occurs. Between those values, erosion and deposition balance each other. The erosion algorithm used is based on the classical approach of Partheniades (1965). Erosion occurs when the bottom shear stress exceeds the threshold of erosion. The flux of eroded matter is given by:

$$\begin{cases} F_E = E\left(\frac{\tau}{\tau_E} - 1\right) & for \quad \tau_b > \tau_{CSE} \\ F_E = 0 & for \quad \tau_b < \tau_{CSE} \end{cases} \tag{7}$$

where τ is the bed shear stress, τ_{CSE} is a critical shear stress for erosion and E is the erosion parameter (kg m^{-2} s^{-1}). This erosion algorithm is computed at the sediment-water interface. If this layer is eroded, erosion occurs from the underlying sediment layer, which has a higher level of compaction, therefore increasing the erosion shear stress thresholds. This is obtained by defining τ_{CSE} as depth dependent, reflecting the increasing resistance of the sediment to be eroded as scouring reaches deeper layers. Wave induced shear stress can also be computed by the model by a linear wave theory, given wave characteristics such as wave period and wave significant height. Estuarine local waves can be important in terms of sediment resuspension, especially in shallow water where the wave stresses effect reaches the sediment bed. Pina (2001) presents a detailed description on the formulation implemented in the model. On the other hand, the deposition flux can be defined as:

$$F_D = -p(W_S C)_b \tag{8}$$

where p is the probability of sediment particles to settle down on the bed; W_S is near-bed the settling velocity; and C the near-bed fine sediment concentration. The probability of deposition (Krone 1962), can be defined as:

$$p = (1 - \frac{\tau_b}{\tau_{CSD}}) \tag{9}$$

where τ_b (Pa) and τ_{CSD} (Pa) are the bottom shear stress and the critical shear stress for deposition respectively. This concept reflects the fact that the deposition of flocks is controlled by near-bed turbulence. For a flock to stick to the bed, gravitational forces must be strong enough to withstand the near bed shear stress. The deposition algorithm (Krone 1962), like the erosion algorithm, is based on the assumption that deposition and erosion never occur simultaneously, i.e., a particle reaching the bottom has a probability of remaining there that ranges from 0 to 1 as the bottom shear stress varies between its upper limit for deposition and zero respectively. Deposition is calculated as the product of the settling flux and the probability of a particle to remain on the bed:

$$\begin{cases} F_D = (CW_S)_B(1 - \frac{\tau}{\tau_{CSD}}) & for \quad \tau_b < \tau_{CSD} \\ F_D = 0 & for \quad \tau_b > \tau_{CSD} \end{cases} \tag{10}$$

The critical shear stress for deposition depends mainly on the size of the flocks. Bigger flocks have a higher probability of remaining on the bed than smaller flocks. As only a single characteristic class of fine sediment is considered in the model, parameters must be calibrated, starting from reference values found in literature, in order to achieve good approximations in the final results. Consolidation is considered to occur in recently deposited sediments at the sediment-water interface and is modelled as a sediment flux, $F_{consolidation}$ ($kg_{sed}\ m^{-2}\ s^{-1}$), between the fluff layer and the first sediment layer at a certain rate, $k_{consolidation}$ (s^{-1}), dependent on the sediment mass per unit of surface area deposited at the fluff layer. It is assumed that consolidation only occurs when shear stress (τ_b) is lower than the critical shear stress for deposition (τ_{CSD}):

$$\begin{cases} F_{consolidation} = 0 & for \quad \tau_b > \tau_{CSD} \\ F_{consolidation} = M_{sediment} \cdot k_{consolidation} & for \quad \tau_b < \tau_{CSD} \end{cases} \tag{11}$$

This consolidation flux is one of the governing processes for particulate contaminant fractions to enter the sediment compartment.

3.5.2 Particulate properties fluxes

Particulate properties fluxes at the sediment-water interface depend on erosion and on consolidation processes. As the erosion algorithm was developed specifically for fine sediment modelling, when computing other particulate properties fluxes at the bed, the erosion rate parameter cannot be the same. Thus, a specific proportionality factor for the erosion constant is computed, E_{prop}, for each property, relating the quantity of property ($M_{property}$ in $kg_{property}\ m^{-2}$) to the quantity of fine sediment deposited in the bed ($M_{sediment}$ in $kg_{sed}\ m^{-2}$). The particulate property erosion flux is then computed similarly to fine sediments but with a specific E_{prop}:

$$E_{prop} = E\left(\frac{M_{property}}{M_{sediment}}\right) \tag{12}$$

In this way, critical shear stress values are considered equal for all particulate properties, with the specific erosion constant being the differentiating factor. When consolidation occurs a similar algorithm is followed, relating the sediment consolidation flux to the particulate property deposited mass. Thus, the property consolidation flux (F^{prop}) can be computed with the following expression:

$$F^{prop}_{consolidation} = F^{sediment}_{consolidation}\left(\frac{M_{property}}{M_{sediment}}\right) \tag{13}$$

3.5.3 Dissolved properties

Dissolved properties fluxes across the water-sediment interface depend both on erosion / consolidation processes and on concentration gradients between the water column lower layer and on the interstitial water of the sediment upper layer. As stated before, when the fluff layer is active (i.e. there are recently deposited sediments on the bed), interstitial water between those sediment particles is not considered. Thus, when erosion occurs there is no dissolved properties influx from the fluff layer to the water column. The interstitial water in the sediments upper layers (containing solutes such as dissolved contaminant fractions, nutrients, etc) is flushed to the water column when consolidated sediment is eroded (upper sediment compartment layer). On the other hand, when consolidation occurs, water overlying the sediment bed becomes part of the sediment interstitial water. These processes constitute an additional flux of solutes to and from the water and sediment columns. Thus, a water flux (F^{water} in $m^3\ s^{-1}$) can be computed, corresponding to the amount of porewater dragged along with the eroded sediments or to the amount of overlying water captured in the consolidation process:

$$F^{water}_{erosion/consolidation} = F_{erosion/consolidation} \cdot A \cdot \phi_k \cdot \frac{1}{\rho_{sed} \cdot (1 - \phi_{kn})} \tag{14}$$

where, $F_{erosion/consolidation}$ is the fine sediment flux ($kg_{sed}\ m^{-2}\ s^{-1}$) between the sediment-water interface and the sediments' upper layer, ϕ_{kn} is the porosity in the upper (k=n) sediment layer, ρ_{sed} is the sediment dry density ($kg_{sed}\ m^{-3}_{sed}$) and A is the area (m^2) of the sediment-water interface. Respectively, solute fluxes are given by:

$$F^{solute}_{erosion/consolidation} = \frac{F^{water}_{erosion/consolidation} \cdot C^{solute}}{A} \tag{15}$$

where C is solutes' concentration ($kg\ m^{-3}_{water}$) in the sediment upper layer or in the water column bottom layer, depending on the type of flux (erosion or consolidation). As mentioned above, the concentration gradients between the water column bottom layer and the sediment surface layer can also produce a mass flux through the sediment-water interface. Solutes in a turbulent flow can be transported by a mean advective flux, turbulent diffusion and molecular diffusion. It is usually considered that solutes diffusion coefficient is equal to the fluids turbulent viscosity, which are usually several orders of magnitude higher. Nonetheless, when approaching the sediment bed water flow is reduced, and the same is true of turbulent motion, leading to an increase in the importance of molecular diffusion relative to the turbulent one. Thus, a sub-diffusive layer (Boudreau 1997) is formed, where a linear concentration gradient can be considered, and a diffusive flux, $F_{diffusive}$ ($kg_{solute}\ m^{-2}\ s^{-1}$), can be computed representing the rate at which this gradient tends to be eliminated:

$$F_{diffusive} = \frac{D_{molecular}}{\delta} \cdot A \cdot (C_{water} - C_{intertidal}) \tag{16}$$

in which $D_{molecular}$ is the molecular diffusion coefficient ($m^2\ s^{-1}$), and δ (m) is the sub-diffusive boundary layer thickness, which is dependent on near-bed turbulence:

$$\delta = \frac{2 \cdot \nu_{water}}{u_+} \tag{17}$$

where ν_{water} is the water kinematic viscosity ($m^2\ s^{-1}$) and u_+ is near-bed shear velocity ($m\ s^{-1}$).

4 SEDIMENT COLUMN MODEL

The sediment compartment consists of saturated porous media, formed by sediments and by water that fills the interstices between the sediments. Properties in this compartment can either be dissolved (in the porewater), or particulate (adsorbed on to sediments). The sediment-water interface handles processes occurring between the water and the sediment column. Since it is very difficult to define physically, this interface really is an abstraction. In the model it can be seen as a thin sediment layer (fluff-layer) with transient characteristics, depending basically on temporal scales associated with hydrodynamics and transport in the water column, namely erosion and deposition. This layer has a separation function, which allows dissociating processes that occur on the sediment deposit, at a very slow scale, "filtering" the high frequencies of erosion/deposition fluxes that shape it, therefore leading to consolidation. Dissolved properties can be produced in the interface but their mass is not part of it, becoming part of the water column by means of a boundary condition flux. Contrastingly, particulate properties are part of the sediment-water interface. This can be the case when sediment deposition occurs but the sediment is not yet consolidated. Thus, a particulate property deposited mass is tracked in order to know how much of is available when erosion conditions occur. Following this concept, it is considered that dissolved properties can exchange fluxes directly between the water column and the sediment interstitial water. In erosion conditions, if this transient layer is completely eroded, then scouring takes place from the sediment compartment upper layer, where consolidated sediment is present. When this happens, interstitial water is entrained along with the sediment, constituting a flux to the water column. In the same way, when the fluff-layer consolidates and becomes part of the sediment column there is an input of overlying water (and its properties) to the sediment compartment. The sediment column model is a set of 1D vertical models defined below the 3D water column model (Figure 3). Both models share the same horizontal discretization, but compute independent vertical coordinates. Adsorption and desorption processes are simulated with a similar approach as in the water column.

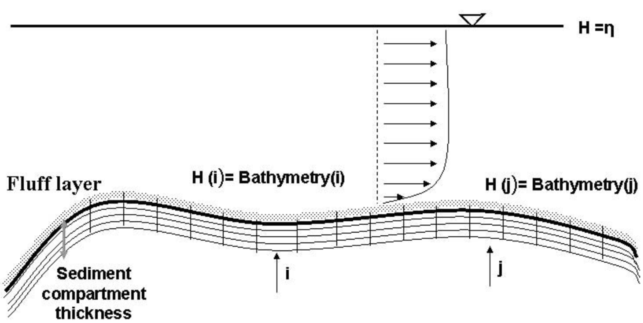

FIGURE 3: *Sediment compartment discretization.*

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MODELLING POLLUTION: OIL SPILLS AND FAECAL CONTAMINATION

M. MATEUS AND R. FERNANDES

1 INTRODUCTION

Human activities in coastal areas degrade biota and affect human health. Pollution caused by oil spills and sewage disposal are among the most obvious examples of this reality. Both have clear detrimental effects, with obvious socio-economic consequences and, over the last decades, have been at the top of the list of estuarine management concerns. Public health risks posed by sewage discharge to sea led the United Nations Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP) to place this threat on the top of its list of concerns, in 1990. While not harming the environment in the same way as oil pollution, sewage impairs human health by the transmission of enteric diseases. Human sewage contains enteric bacteria, pathogens and viruses, and the eggs of intestinal parasites. Contamination of food or drinking and bathing waters may therefore pose a public health hazard. There have been some instances of hepatic and enteric diseases contracted through bathing in contaminated waters (Clark 1992). In most developing countries, discharge of raw sewage in coastal areas and estuaries is still common practice, posing serious threats for the population that makes use of the waters.

Faecal and oil pollution assessment requires descriptive and predictive tools, such as numerical simulation models, that are able to reproduce the dynamics of the systems in study, and simulate the fate of the pollutants under a broad range of scenarios (Bach et al. 1995, Christodoulou et al. 1995, Rodriguez et al. 1995, Garvey et al. 1998, Mahajan et al. 1999, Noutsopoulos et al. 1999). This chapter presents synthetic description of the oil and faecal coliform modules of the MOHID modelling system. They have been applied in the ECOMANAGE study sites to study the influence of the hydrodynamic regime on the fate of these pollutants, their impact on the system, and in some cases, to test different management options.

2 MODELLING FRAMEWORK

Attaining an adequate description of the dynamics of contamination is imperative to define water quality management strategies and to develop realistic contingency plans to deal with potential threats. This is frequently achieved through the use of numerical model simulations. A major advantage of the use of models in pollution studies is that they can be used as a diagnostic tool (identifying and studying actual problems) and as a prognostic tool (testing different scenarios). In addition, numerical tools can be implemented to render forecast capacity to pollution assessments. The numerical tool must be able to achieve basic goals such as (1) an accurate assessment of the dispersion of the pollutant, and (2) a reproduction of the basic processes that affect the state and fate of the pollutant in the environment.

Contrary to oil spills that usually have a tremendous visual impact, feacal pollution can be unnoticed. A striking difference between these two forms of pollution is their residence time in the water; hydrocarbons can last for long periods of time when compared with faecal agents. Faecal bacteria are bioindicators, meaning that their residence time in aquatic systems is comparatively low, usually ranging from less than an hour to a few hours, and occasionally up to some days. However, from a modelling perspective, oil spills and faecal pollution share same basic requirements in respect to the underlying physical mechanisms of transport in water. Hydrodynamic processes govern the dispersion of contaminants in the receiving area. So, by coupling general hydrodynamic processes and pollutant-specific processes, models can determine the plume evolution, enabling the prediction of affected areas and ambient concentrations over time. In summary, models enable the assessment of the magnitude of the pollution.

The effects that transport, ambient conditions, domain geometry and bathymetry, and other variables have on the dynamics of these pollutants in aquatic environments, require that models have the ability to reproduce their contribution to the dynamics of the system. The MOHID numerical platform (more details in the previous chapter) is such a tool because of the wide range of processes it simulates. Within this modelling framework there are independent modules that deal with the dynamics of these pollutants. This chapter deals with these modules, addressing in a brief way the processes they simulate and the baseline modelling philosophy.

3 THE LAGRANGIAN MODULE

Lagrangian transport models are very useful to simulate localized processes with sharp gradients (submarine outfalls, sediment erosion due to dredging works, hydrodynamic calibration, oil dispersion, etc.). The MOHID model uses the concept of lagrangian tracers to assess the spatial-temporal evolution of the contamination plume, determined by tidal regime and local circulation. Tracers are transported by currents calculated by hydrodynamic model and each tracer has the ability to be associated with one or more properties (physical, chemical or biological). This model is a subset of the MOHID modelling system and has been used in other instances also to study pollutant dispersion (Gomez-Gesteira et al. 1999). At the present stage the model is able to simulate oil dispersion, water quality evolution and sediment transport. The lagrangian module interacts with other modules such as the oil dispersion module to simulate oil dispersion and the *Escherichia coli* decay module. Sediment transport can be associated directly to tracers using the concept of settling velocity.

3.1 Tracer concept

Tracers are characterized by their spatial coordinates, volume and a list of properties, each with a given concentration. The most important property of a tracer is its position in space (*x,y,z*). The tracer can be a water mass, a sediment particle or group of particles, a molecule or group of molecules, or even a phytoplankton cell. The movement of tracers can be influenced by the velocity field from the hydrodynamic module, by the wind from the interface water-air module, by the spreading velocity from oil dispersion module and by random velocity. Both volume and properties concentration of each tracer vary in time in response to different parameters and ambient conditions. For *E. coli*, for example, volume is affected by turbulent mixing while fecal concentration depends on environmental factors like irradiance, temperature and salinity. Tracers belonging to the same origin have the same list of properties and use the same parameters for random walk, *E. coli* decay, etc. Origins can differ in the way they emit tracers. There are three different ways to define origins in space: (1) Point Origins - emits tracers at a given point; (2) Box Origins - emits tracers over a given area; (3) Accident Origins - emit tracers in a circular form around a point. Origins, in turn, emit tracers in two different ways: (i) Continuous - emits tracers during a period of time; (ii) Instantaneous - emits tracers at one instant.

3.2 Tracer Movement

The major factor responsible for particle movement is generally the mean velocity. Spatial coordinates are given by the definition of velocity:

$$\frac{dx_i}{dt} = u_i(x_i, t) \tag{1}$$

where u is the mean velocity and x the particle position. This equation is solved using a simple explicit method:

$$x_i^{t+\Delta t} = x_i^t + \Delta t \cdot u_i^t \tag{2}$$

Higher order accuracy requires the use of an iterative procedure. For most natural flows, the explicit method is sufficiently accurate. Velocity at any point in space is calculated using a linear interpolation between the points of the hydrodynamic model grid. The lagrangian module allows splitting the calculation of the trajectory of the tracers into sub-steps of the hydrodynamic time step.

3.3 Turbulent Diffusion

Turbulent transport is responsible for dispersion. The effect of eddies over particles depends on the ratio between eddies and particle size. Eddies bigger than the particles make them move at random, while eddies smaller than the particles cause entrainment of matter into the particle, increasing its volume and mass according to the environment concentration. Random movement is calculated following the procedure proposed by Allen (1982). The random displacement is calculated using the mixing length and the standard deviation of the turbulent velocity component, as given by the turbulence closure of the hydrodynamic model. Particles retain that velocity during the necessary time to perform the random movement, which is dependent on the local turbulent mixing length. The increase in volume is associated with small-scale turbulence and is reasonable to assume it as isotropic. Under these conditions, small particles keep their initial form and their increase in volume is a function of the volume itself.

4 OIL SPILLS

The prediction and simulation of the trajectory and weathering of oil spills are essential to the development of pollution response and contingency plans, as well as to the evaluation of environmental impact assessments. In order to predict the fate of oil products spilled in coastal zones, the oil weathering model predicts the evolution and behavior of the processes (transport, spreading, evaporation, etc.) and properties (density, viscosity, etc) of the oil products.

Oil density and viscosity, and many different processes such as oil spreading, evaporation, dispersion, sedimentation, dissolution, emulsification and oil beaching have been included in the oil module. Depending on the characteristics of the computational mesh or the magnitude of the spill, the model considers different alternative methods to simulate some of these processes. The oil weathering module (OWM) uses mainly the hydrodynamics and lagrangian transport modules. The hydrodynamic module simulates the velocity field necessary for the lagrangian module to calculate oil trajectories. These oil trajectories are computed assuming that oil can be idealized as a large number of particles that independently move in water.

Water properties and atmospheric conditions are introduced in the lagrangian module and used by the oil module to determine oil processes and properties. Except for the spreading and oil-beaching, all weathering processes and properties are assumed to be uniform for all tracers, like water properties and atmospheric conditions. These are assumed to be equal to the environmental conditions at the accident's origin. Oil temperature is assumed equal to water temperature, neglecting solar radiation or any other energy transfer process that may influence oil temperature. In its current setup MOHID OWM is not a 3D application. It simulates the amount of oil that leaves the water surface (by different processes like evaporation, or dispersion in water), without simulating the evolution at the subsurface and variations in the water column.

4.1 Modelled processes

Only a description of the modelled processes is presented here, since detailed information on the governing equations and model parameterization is available in the form of a User's Manual for download at the MOHID's model website (`http://www.mohid.com`).

4.1.1 Spreading

For an instant spill accident, the initial area of spilled oil is calculated according to Fay's formulation (Fay 1969). Two different algorithms are available to estimate oil spreading. One of the algorithms determines random velocities assuming a uniform distribution inside a range (in directions *x* and *y*), proportional to diffusion coefficients, which are calculated assuming that lagrangian tracers spreading is equivalent to Fay's formulas solution (Fay 1969). The only phase simulated in spreading is the gravity-viscous phase, from solutions proposed by Fay.

The other algorithm proposed for oil spreading is based in thickness differences inside the oil slick, presuming that the existence of a thickness gradient generates a "spreading force" in the direction of minor thickness. Therefore, a tracer will move from the computational cell with larger oil thickness to the thinner one. This formulation uses a coefficient to approach the solution to the Fay solution, in order to make results sensible to some factors, like different oil densities, originating different behaviors. In the oil module, velocities are calculated in the faces of cells where oil is present, in directions *x* and *y*. Subsequently, in the lagrangian module tracer velocities are interpolated based on cell faces velocities and tracer position. If the average oil thickness becomes too thin (less than a value between 0.1 and 0.01 mm, depending of product viscosity), oil spreading is stopped.

4.1.2 Density and viscosity

Oil density is estimated considering the density of the emulsion at ambient temperature, the density of fresh oil at a reference temperature and the water temperature. The oil's initial density is obtained from the algorithm proposed by the American Petroleum Institute (API). Only oil products with lower density than water are modelled, because higher density products will sink. In the model the oil viscosity is controlled by three major factors: temperature, evaporation and emulsification.

4.1.3 Evaporation and emulsification

In MOHID the oil evaporation process can be estimated by two different methods: an analytical method, also known as the evaporative exposure method (Stiver and Mackay 1984), and by a more recent methodology proposed by Fingas (1998), where the relevant factors are time and temperature. Square root equations can also be used in some refined oils and in short term simulations (1-2 days). The emulsification process consists in the incorporation of water in oil, usually starts after a certain amount of oil has evaporated. An emulsification constant is used, which means the percentage of oil evaporated before emulsification starts. By default, this constant is 0%. When emulsification starts, incorporation of water in oil can be simulated by two different algorithms: the widely used equation of Mackay et al. (1980) and the Rasmussen equation (Rasmussen 1985).

4.1.4 Dispersion

This is the process where oil droplets entrain the water column. Two different methods are available to predict this weathering process, based on the formulations proposed by Delvigne and Sweeney (1998), and Mackay et al. (1980). The latter method is a simplified algorithm developed for vertical dispersion as a function of squared wind velocity, for conditions where turbulent energy is difficult to determine.

4.1.5 Dissolution

Dissolution is quantified through the Cohen method, considering the analytical solution for the solubility of typical oil proposed by Huang and Monastero (1982).

4.1.6 Sedimentation

Although the process of oil sedimentation is relatively difficult to estimate, the MOHID model uses a formulation developed by Science Applications International (Payne et al. 1987) for this purpose. Only droplets greater than 70 microns and smaller than 200 microns are considered for sedimentation. Bigger droplets are less likely to stick to sediment particulate matter, and those smaller than 70 microns are already estimated in the dispersion process.

4.1.7 Oil-Beaching

When oil reaches a coastal zone, it might become beached. This model estimates the amount of beached oil when the model user predefines a beaching probability (or different beaching probabilities for different coastal zones).

5 FAECAL CONTAMINATION

The reduction of faecal contaminant loads in the water is achieved by a combination of three main factors: initial dilution, dispersion and bacterial decay. Dynamic models usually take into consideration the physical processes of dilution and dispersion but frequently ignore the influence of abiotic effects on bacteria mortality by assuming a fixed mortality rate. Under certain conditions, this limitation hinders the results. For the specific case of the decline in faecal indicator bacteria, several studies have shown a particularly relevant role of abiotic factors like temperature, salinity and irradiance (Pereira and Alcantara 1993, Sarikaya and Saatci 1995, Serrano et al. 1998). A more realistic assessment of temporal and spatial faecal contamination can be achieved by using a feacal decay model with a dynamic T90 as a function of instant solar radiation, water temperature and salinity. Faecal dispersion can be simulated using both the lagrangian and the eulerian transport schemes.

5.1 Faecal decay model

TC and FC groups have similar decay rates (Marais 1974) and have been commonly used as indicators to assess water quality state. The die-off rate of this class of organisms is represented by a first-order equation (Chick's Law for disinfection), which states that the rate of loss is proportional to the concentration:

$$\frac{\partial N}{\partial t} = -kN \tag{3}$$

where N is the initial bacterial concentration in the effluent and k, the first-order decay rate (day^{-1}) or die-off coefficient, usually described in the form of a sum of individual parameters:

$$k = k_b + k_i + k_s + k_p \tag{4}$$

where k_b is the base mortality - a function of temperature and salinity - k_i is the death rate due to solar radiation, k_s is the net loss/gain due to settling/resuspension, and k_p is the mortality rate induced by predation. In the MOHID framework the fecal mortality model accounts for the impacts of temperature, salinity and ambient light in the decay of fecal indicators. It is derived from in situ and laboratory studies of mortality rates of *E. coli* made in the Cantabrian Sea (Canteras et al. 1995). The contribution of settling/suspension and grazing were not considered. As such, the simultaneous combination of all factors considered is expressed as:

$$k = 2.533 \times 1.04^{(T-20)} \times 1.012^{S} + 0.113 i_z \tag{5}$$

where S and T are the surrounding water salinity and temperature (°C), respectively, and i_z is the irradiance (watt m^{-2}) at depth z (m). Irradiance levels in the water environment are estimated by the hydrodynamic model where the light extinction is already parameterized and the irradiance is known for each vertical layer (depth integrated). This implies that for 2D-horizontal settings a mean value is calculated from irradiance levels at surface, considering the light attenuation effect of water molecules over the height of the water column. Bacterial decay is usually expressed as T90, the time in which 90% of population is no longer detectable, meaning 1 log reduction in number of pathogens. Assuming a first-order loss, the 90% mortality time is obtained by:

$$T_{90} = 2.303 k^{-1} \tag{6}$$

Together with FC concentrations, the model also outputs T90 values since these can be used to explain the underlying dynamics of the contamination patterns. This is particularly useful is systems with a high spatial and temporal changes in ambient conditions.

5.2 The influence of abiotic parameters

Fixed T90 values are still widely used in modelling studies to assess the impact of bacterial inputs in water bodies (Kashefipour et al. 2002). However, when this methodology is applied to areas with strong daily fluctuations of irradiance, usually associated with tidal movements ressuspending sediments and blocking light, it may fail to consider the major influence of light in bacterial decay. The explicit modelling of abiotic effects on FC decay, on the other hand, accounts for the variation in ambient conditions. The wide range of T90 values over a daily cycle is obvious when we consider a typical diel period in late spring and summer at mid latitudes. It has been pointed that impact studies of FC contamination usually consider T90 values lower than the values measured in culture experiments (Guillaud et al. 1997). This assumption can compromise the quality of model predictions, limiting the role of models has predictive tools. This is further aggravated because static values are used from the entire simulation period.

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LOAD AND FLOW ESTIMATION: HARP-NUT GUIDELINES AND SWAT MODEL DESCRIPTION

P. CHAMBEL-LEITÃO

1 INTRODUCTION

When eutrophication is considered an important process to control it can be accomplished reducing nitrogen and phosphorus losses from both point and nonpoint sources and helping to assess the effectiveness of the pollution reduction strategy. HARP-NUT guidelines (Guidelines on Harmonized Quantification and Reporting Procedures for Nutrients) are presented by OSPAR as the best common quantification and reporting procedures for calculating the reduction of nutrient inputs. In 2000, OSPAR adopted the Harmonized Quantification and Reporting Procedures for Nutrients (HARP-NUT guidelines) on a trial basis. They were intended to serve as a tool for OSPAR Contracting Parties to report, in a harmonized manner, their different commitments, present or future, with regard to nutrients under the OSPAR Convention, in particular the ”Strategy to Combat Eutrophication”.

OSPAR 2000 adopted HARP-NUT guidelines number 1-9, except for number 6 on diffuse sources. OSPAR 2000 agreed that the further development of draft Guideline 6 within OSPAR should only start when the results of the EC Fifth Framework Programme Euroharp project on an intercomparison of quantification models for losses from diffuse anthropogenic sources were available. OSPAR 2004 adopted revised versions of HARP-NUT guidelines 1,3,4,5,7,8 and 9 and noted a report on progress on the euroharp project, indicating that the output from this project was expected to become available at the end of 2004 for use in the further development of HARP-NUT Guideline 6. OSPAR 2007 adopted the HARP-NUT GL 6 on a trial basis for the 2007/2008 and 2009/2010 implementation reporting rounds under PARCOM Recommendation 88/2.

There is no such approach to quantify loads of nutrients specific for all South America, which is why HARP-NUT guidelines were used in ECOMANAGE. The use of a standard methodology like the one of HARP-NUT guidelines is a first iteration to study the eutrophication in South America. As such, this application could be the first step in the development of HARP-NUT guidelines for South America. In this chapter the methodology and application of HARP-NUT guidelines is described with special emphasis in: i) HARP-NUT description ii) SWAT model; iii) Applicability of HARP-NUT and SWAT to South America.

2 HARP-NUT GUIDELINES GENERAL DESCRIPTION

HARP-NUT Guidelines (Borgvang and Selvik 2000, Schoumans 2003) were developed to quantify and report on the individual sources of nitrogen and phosphorus discharges/losses to surface waters (Source Orientated Approach). These results can be compared to nitrogen and phosphorus figures with the total riverine loads measured at downstream monitoring points (Load Orientated Approach), as load reconciliation. Nitrogen and phosphorus retention in river systems represents the connecting link between the ”Source Orientated Approach” and the ”Load Orientated Approach”. Both approaches are necessary for verification purposes and both may be needed for providing the information required for the various commitments.

Guidelines 2,3,4,5 are manly concerned with the sources estimation (Figure 1). They present a set of simple calculations that allow the estimation of the origin of loads. Guideline 6 is a particular case where the application of a model is advised, in order to estimate the sources of nutrients from diffuse sources associated with land use/land cover. The model chosen for this was SWAT model because it is suggested in the guideline 6 and because it is widely used in the world (including South America). The development of HARP-NUT Guidelines to quantify and report on the individual components of nitrogen and phosphorus discharges/losses to inland surface waters is intended to allow the aggregation of the discharges/losses of nitrogen and phosphorus in each catchment (Source Orientated Approach). By taking account, where appropriate, of nitrogen and phosphorus retention processes in river systems and background losses of nitrogen and phosphorus, it is possible to compare the aggregated nitrogen and phosphorus figures on discharges/losses at source with the total riverine loads measured at downstream monitoring points (Load Orientated Approach), as a load reconciliation (guideline 7). Nitrogen and phosphorus retention (guideline 9) in river systems represents the connecting link between the ”Source Orientated Approach” and the ”Load Orientated Approach”. Associated with each guideline there is a set of minimum data requirements needed. In Figure 1 presents the data needed for each guideline, as well as the time rate typically available of each data.

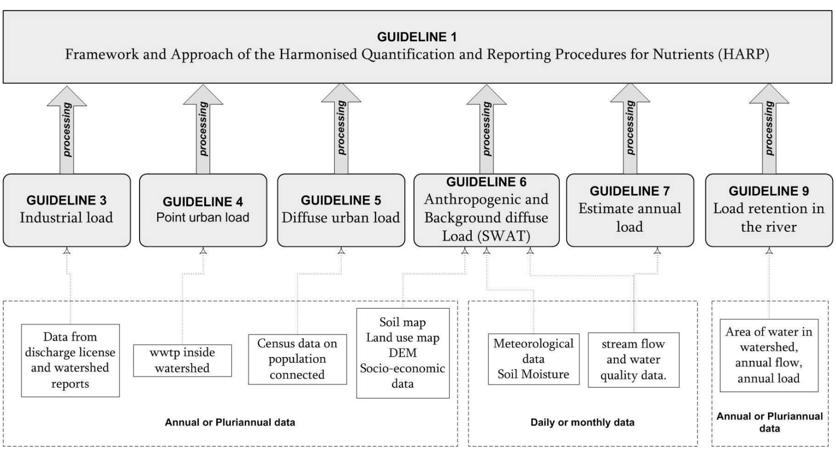

FIGURE 1: *HARP-NUT guidelines work flow.*

3 SWAT MODEL

In SWAT simulated basin is divided in several sub-basins. Each sub-basin can be divided in many HRU (Hidrologic Response Units - units with the same land use and soil type) or it can be only one HRU. Each HRU has as superior boundary soil surface, and as inferior boundary the aquifer (Figure 2). It receives for the superior boundary precipitation, of which part is converted into run-off and another part is converted in infiltration. The part that is converted into run-off is directed to the sub-basin channel. The part that infiltrates is carried along the soil profile, being able to evapotranspirate, to be percolated to the aquifer or carried laterally along the soil profile until it reaches the channel. The water that reaches the aquifer is lost for the channel or the deep aquifer or finally for the atmosphere (the effect of capillary rise is simulated like this because SWAT soil hydrodynamics is one way: only allows water to percolate and no capillary rise). Because the evapotranspiration rate is strongly influenced by a number of vegetative surface characteristics, PET is the rate at which evapotranspiration would occur from a large area uniformly covered with growing grass, completely shading the ground, of uniform height and never short of water. Penman-Monteith method describes the equation to estimate PET (Monteith 1965, Allen 1986, Allen et al. 1989). Once total potential evapotranspiration is determined, actual evaporation must be calculated. SWAT first evaporates any rainfall intercepted by the plant canopy. Next, SWAT calculates the maximum amount of transpiration and the maximum amount of sublimation/soil evaporation based on Richtie's (1972) method.

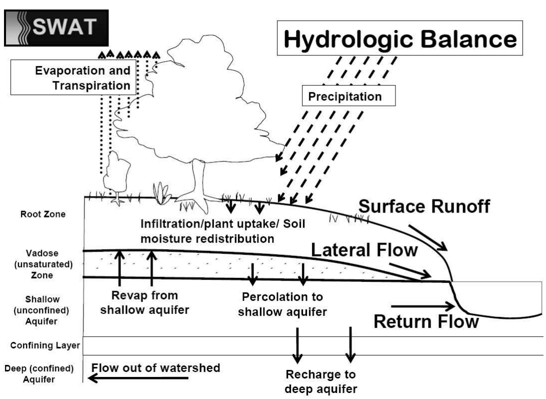

FIGURE 2: *Example of the Hidrologic Response Units (HRU).*

Plant growth in SWAT is estimated using the heat unit theory. This theory postulates that plants have heat requirements that can be quantified and linked to time to maturity. Because a plant will not grow when the mean temperature falls below its base temperature, the only portion of the mean daily temperature that contributes towards the plant's development is the amount that exceeds the base temperature. To measure the total heat requirements of a plant, the accumulation of daily mean air temperatures above the plant's base temperature is recorded over the period of the plant's growth and expressed in terms of heat units (Barnard 1948, Phillips 1950). Run-Off in SWAT is based on the SCS runoff equation, which is an empirical model that was the product of more than 20 years of studies involving rainfall-runoff relationships from small rural watersheds across the U.S. The model was developed to provide a consistent basis for estimating the amounts of runoff under varying land use and soil types (SCS 1972, Rallison and Miller 1981).

Manning's equation for uniform flow in a channel is used to calculate the rate and velocity of flow in a reach segment and also to estimate overland flow. Percolation is calculated for each soil layer in the profile. Water is allowed to percolate if the water content exceeds the field capacity water content for that layer. Water that percolates out of the lowest soil layer enters the vadose zone. The vadose zone is the unsaturated zone between the bottom of the soil profile and the top of the aquifer. An exponential decay weighting function proposed by Venetis (1969) and used by Sangrey et al. (1984) in a precipitation/groundwater response model is utilized in SWAT to account for the time delay in aquifer recharge once the water exits the soil profile. The baseflow recession constant is a direct index of groundwater flow response to changes in recharge (Smedema and Rycroft 1983). Values vary from 0.1-0.3 for land with slow response to recharge to 0.9-1.0 for land with a rapid response. Although the baseflow recession constant may be calculated, the best estimates are obtained by analyzing measured streamflow during periods of no recharge in the watershed.

4 HARP-NUT AND SWAT APPLICABILITY TO SOUTH AMERICA

For the application of Source Orientated Approach of HARP-NUT guidelines, a great amount of data is needed because a study of a catchment is complex and needs the use of many sources of data in order to represent and clarify the various aspects that affect water quality. For the Source oriented approach (guidelines 2, 3, 4, 5) data was obtained in the three sites from national census, environmental agencies and agriculture agencies. For guideline 6 input data are the ones needed by SWAT model: topography, land use/land cover, soil type and meteorology. The digital elevation model (DEM) was obtained from the Shuttle Radar Topography Mission (SRTM) DEM data (Hounam and Werner 1999). This mission covered the entire South America. Land Use/Land Cover maps were obtained for Argentina and Brasil from Landsat satellite images (ETM+) downloaded from the free site `http://glcf.umiacs.imd.edu.data`. The type of classification applied was Supervised Classification. For Chile maps of Land use were already available, and they were the ones used for watershed modelling. Often there is no available data on soil properties like porosity, wilting point, hydraulic conductivity and field capacity. This was the case in all study sites. However some data like texture was available. Soil properties were derived in ECOMANAGE study sites using pedotransfer functions. These functions consist in predictive functions of certain soil properties from other more available, easily, routinely, or cheaply measured properties like texture. Many studies have developed statistical correlations between soil texture and selected soil potentials (versus water content) using a large data base, and also between selected soil textures and hydraulic conductivity.

Saxton et al. (1986) developed a procedure to estimate soil-water characteristics from readily available inputs using results of previous statistical analyses of a large data base.

Climate Information obtained from national meteorology institute as well as water institutes included: i) average maximum air temperature for month, ii) average minimum air temperature for month, iii) standard deviation for maximum air temperature in month, iv) standard deviation for minimum air temperature in month, v) average daily solar radiation in month, vi) average dew point temperature in month, vii) average wind speed in month and finally daily precipitation. For the load oriented approach water quality and quantity was obtained. For water quality monthly values of concentration of nutrients were obtained for basin and sub basins. For water quantity daily flows were obtained. Retention of Nitrogen and phosphorus in river systems represents the connecting link between the "Source Orientated Approach" and the "Load Orientated Approach". Retention depends manly on: i) The portion of lakes, river stretches and wetland in each catchment; ii) The hydrological and morphological conditions within the river system. Nitrogen and phosphorus retention used in ECOMANAGE were quantified on the basis of the mass balance of investigated lakes and rivers in Europe. In future studies retention results should be complemented with local studies of retention.

The applicability of HARP-NUT and SWAT can also be evaluated by the quality and usefulness of the results. In the case of Aysén and Santos flows and the load of nutrients estimated were used for the calibration of the MOHID estuary model, which produced reasonable results. For Bahía Blanca, the estimated flow values confirmed the limited effect on the Bahía Blanca estuary. Moreover data on rivers and on meteorology was very poor, but a set of data from the forties allowed calibrating flow in SWAT for Naposta watershed. Less than 10% of the precipitation was transformed in flow. This fact associated with the relative small scale of the watersheds resulted in an insignificant influence of flows and loads of nutrients on the Bahía Blanca estuary. In Santos estuary main sources are industries. It's difficult to make the analysis of industries effluents. However major contributors to nutrient loads were identified by using the source and load reconciliation strategy of HARP-NUT. In Aysén most pollution is diffuse, and there was available data to run and calibrate SWAT model. However, changes to SWAT code had to be made (within this project) in order to adapt it to South America particularities. One of the most important changes was to include organic nitrogen in the rain. This change decreased the differences between "Source Orientated Approach" and the "Load Orientated Approach". One of the strong aspects of HARP-NUT guidelines is the integrated approach to the eutrophication problem in coastal areas, which prove to produce reasonable results which were used as an input for the MOHID estuary model in ECOMANAGE study sites. This application implied the application of SWAT model which generated flows that were also used has input for MOHID.

5 FUTURE WORK

Application of HARP-NUT in ECOMANAGE consisted in an overall nutrient budget that has to be confirmed and improve with new data and more detailed studies. In the case o Santos main improvements include the monitoring of main potential sources of nutrients which are located in Moji and Piaçaguera. In the case of Aysén one of the main improvements suggested is the understanding of soil organic matter distribution in the watershed, as well has the mineralization rate. The reason for this is the importance of soil organic matter in overall budget. As discussed before, the watersheds that drain to Bahía Blanca (Sauce Chico, Saladillo and Naposta) are irrelevant for global studies of estuary hydrodynamics and eutrophycation. They could be relevant for local studies of areas adjacent to river mouths in the estuary, especially in terms of fecal coliforms. The application of HARP-NUT guidelines and the application of SWAT contributed for estimating the global budget of water and nutrients. To obtain this, SWAT model had to be calibrated for the watershed. The implementation of this model could be a valuable tool for future management plans of the watershed.

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GROUNDWATER RECHARGE ASSESSMENT

M.M. OLIVEIRA, J.P. LOBO FERREIRA AND M. YARROW

1 INTRODUCTION

The choice of a model or method to compute recharge derives from the conceptualization of the recharge process of a study area. This conceptualization is based on the physical system, its geometry, all the inputs and outputs of water and its locations. The computation of recharge is based on mass balances between water entering, leaving or being stored in the water system. These mass balances are generally water-mass balances but can also be any substance-mass balance diluted in water. Models to compute recharge may be grouped into mass balances above saturated zone and mass balances in the saturated zone.

The **water mass balances above the saturated zone** are predictive models as they quantify recharge by computing the processes prior to recharge occurrence (precipitation, infiltration, water stored in the surface and in the vadose zone). The **soil daily sequential water balance** is an appropriate method to estimate deep percolation that, in some conditions may be assumed to be equal to recharge. This method requires knowledge of the climatic data to characterize precipitation and reference evapotranspiration, and knowledge of medium characteristic parameters, that depend on the complexity of the selected model. These models allow for estimation of distributed recharge in a region, produce results by recharge episode and may be applied to any geological medium (intergranular, fissured, karstic or more than one type). However, the more general application is for intergranular, as the soil storage is more easily quantified, and preferential pathways are less important.

The **water mass balances in the saturated zone** are response models as they represent the reaction of the groundwater medium to the recharge process. Several methods are available depending on the hydrogeological setting, for instance: (1) surface flow hydrograph separation, (2) spring discharge quantification, (3) flow quantification in aquifer sections, (4) saturated zone storage change (water level change), (5) combination of these methods, also including human water abstractions. These methods are integrative for a region and may compute recharge by episode. In the **surface flow hydrograph separation method** baseflow and direct runoff are separated. Baseflow is an estimate of recharge that occurs in the area defined by a watershed when all groundwater flow inside the watershed discharges to the surface water streams inside that watershed (i.e. there is a coincidence between the watershed and the hydrogeological basin). The hydrogeological settings more favourable to observe this requisite are local systems of metamorphic and igneous rocks, with intergranular or fissured porosity. In some cases of sedimentary rocks with intergranular porosity, even if stratified, this requisite may still be found. The surface flow hydrograph separation method is probably the easiest recharge calculation method to use, as it does not require medium characteristic parameters, and only requires knowledge of daily precipitation and flow series.

The **spring discharge method** provides a direct measurement of the amount of water that recharged the system. It requires the knowledge of the area drained by the spring, which is not an easy value to obtain. Due to the structure of the groundwater flow paths and the springs' significant water volumes this method is mainly applicable for karstic hydrogeological settings. For the other hydrogeological media, despite the possible occurrence of large flow springs, it is likely that it exists diffuse discharge in important amounts that difficult the quantification of discharge. The **flow quantification in aquifer sections** is applicable to any hydrogeological medium requiring the knowledge of the recharge area upgradient the measuring section, the constant monitoring of the piezometric level in both sides of the section and the aquifer transmissivity along the measuring section. These requirements turn the application of the method more difficult. The **water level change method** is based on the direct consequences of recharge. The time step for the application of this method is very short. For the application of this method the difference between groundwater flow entering and leaving the system should be negligible in relation to the water level rise. This method also requires the characterization of effective porosity in the depth of water level oscillation.

Among the methods referred to above, two of them are applied in the case study areas: the **soil daily sequential water balance** and the **surface flow hydrograph separation method**. These methods are described in more detail in the next sections.

2 SOIL DAILY SEQUENTIAL WATER BALANCE

For the conceptual case of an area where there is no artificial recharge, no surface flow entering the area, and the groundwater level is always below the soil zone, the water balance equation for the soil of that area can be expressed by (Figure 1):

$$P - RET - \Delta A_l - Sr - Dp = \varepsilon \qquad (1)$$

where P is the precipitation, RET is the effective evapotranspiration, ΔAl is the variation (*final - initial*) of the water stored in the soil, Sr is the surface runoff, Dp is the deep percolation and ε is the calculation error of the balance. The sequential mass balance approach intends to measure or estimate and compute P, RET, Sr and ΔAl processes, computing Dp by solving equation (1) considering $\varepsilon = 0$. The sequential water balance is carried out in a determined time step, for instance the daily time step. Recharge (R) is then assumed to be equal to Dp:

$$R = Dp = P - RET - \Delta A_l - Sr \qquad (2)$$

The soil daily sequential water balance method is a good method to forecast differences on total recharge in response to changing daily precipitation pattern. Moreover as a general characteristic of the method it allows for the determination of seasonal recharge. However it must be taken into account that the presented method provides a value of the water available for deep percolation, and that this deep percolation will take some time to reach the aquifer. A soil daily sequential water balance methodology was implemented in the BALSEQ numerical model (Lobo Ferreira 1981, Lobo Ferreira and Delgado Rodrigues 1988), originally written in Fortran. Figure 2 shows the flowchart of the BALSEQ model. In this model the runoff curve number (*NC*) that depends on soil hydraulic conductivity and on land use, is used in the process of estimating surface runoff. *NC* values vary between 0 (corresponds to an area with infinite permeability, where all water infiltrates into the soil), and 100 (corresponds to a completely impermeable zone).

The effective evapotranspiration is calculated using the potential evapotranspiration (the evapotranspiration that would occur if the water available in the soil was not a limiting factor) and the amount of water available in the soil. This water available in the soil is calculated by a sequential water balance that daily updates the water stored in the soil. The computation of deep percolation depends on the maximum amount of water available in the soil for evapotranspiration (AGUT = $(sr - wp) \, . \, rd$) in which sr is the specific retention (or field capacity), wp is the wilting point and rd is the depth of the plant roots. If after the process of evapotranspiration the water stored in the soil is above the AGUT value, the water in excess of AGUT becomes deep percolation.

The BALSEQ numerical model has been subject to changes and new methods have been implemented to calculate surface infiltration, effective evapotranspiration and deep percolation. These methods, developed in Oliveira (2004a) have all been included in the BALSEQ_MOD numerical model, written in Visual Basic. The surface infiltration is computed using formulas obtained to generalize the results of the Philip infiltration model (Philip 1957, in Lencastre and Franco 1984, Rawls and Brakensiek 1989) applied to several situations that depend on the water content of the soil, the soil texture, the daily precipitation and its distribution. These formulas depend on the daily precipitation and on two tabulated coefficients (Oliveira 2004a, Oliveira et al. 2008) that depend on the textural soil class and on the initial soil moisture.

The effective evapotranspiration is estimated based on the formulation presented in Allen et al. (1998):

$$RET = (K_a.K_{cb} + K_e).ET_o \tag{3}$$

where ET_o is the reference evapotranspiration, K_{cb} is the basal crop coefficient, K_e is the soil water evaporation coefficient and K_a is the water stress coefficient. The procedures to characterize the parameters in equation (3) are presented originally in Allen et al. (1998). The ET_o represents the evaporation from a hypothetical reference crop under determined weather conditions. The K_{cb} and K_e terms of the equation integrate the physical and physiological differences between the specific field crop and the reference crop, hence their values vary with time (depending on the vegetative stage). The use of the two different coefficients, K_{cb} and K_e, constitutes the dual crop coefficient approach. The K_a term is related to the stress conditions in which the crop develops and depends on the water available in the soil during the crop growth. Also the K_e term is dependent on the soil moisture in the bare soil part. The computation of the soil moisture, on which K_a and K_e depend, is achieved by a daily sequential water balance.

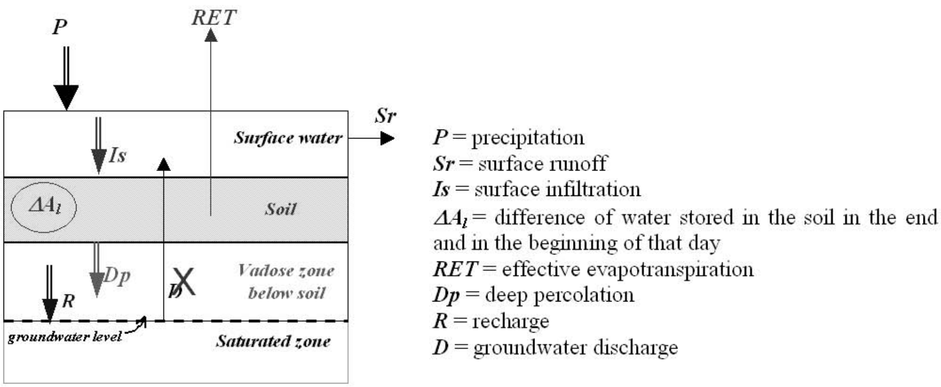

FIGURE 1: *Soil water balance of an area with no discharge of groundwater and no surface flow entering in the system.*

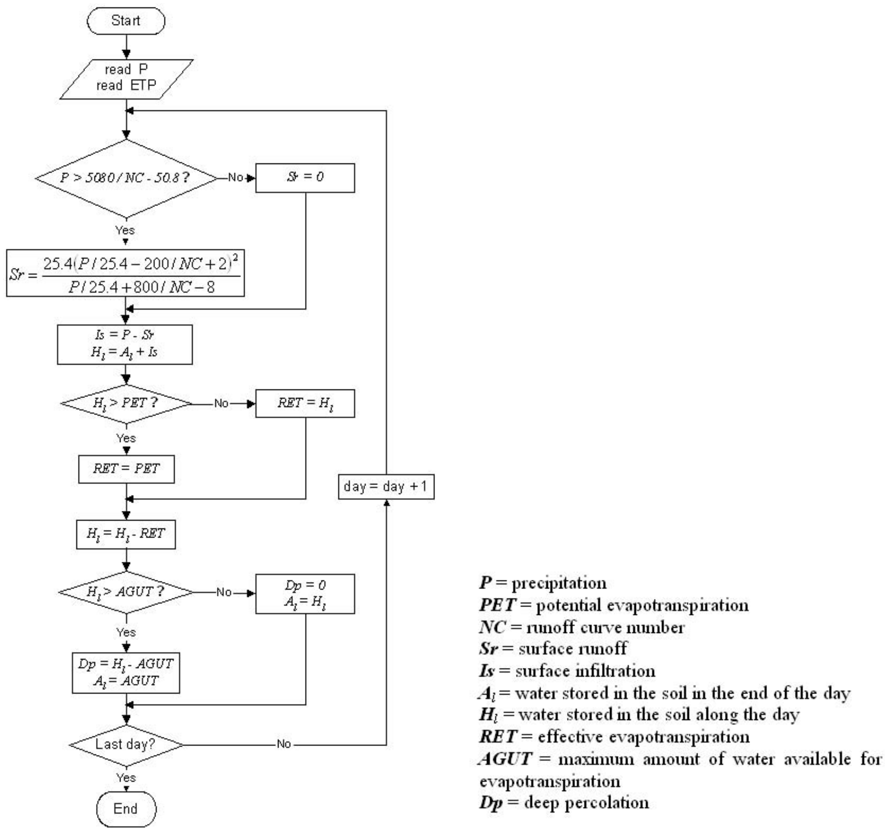

FIGURE 2: *Flow chart of BALSEQ model for daily sequential water balance in the soil.*

The methodology that was implemented in BALSEQ_MOD allows for the existence of up to two land covers and a bare soil surface. The area occupied by each cover may vary in time. For instance, in the case of two different vegetation covers, the area occupied by each cover can change according to the vegetation development period. These features required the soil area to be divided in up to three parts. For each soil part, a daily sequential water balance is carried out. The water balance of one part is dependent on the water balance of the other two parts. So the sequential water balance must be performed simultaneously.

The following information is required to estimate effective evapotranspiration: (1) daily surface infiltration; (2) daily reference evapotranspiration; (3) the fraction of the area occupied by each land cover; in the case of vegetation cover it is necessary to know the area fraction occupied by the vegetation during mid-season and late-season stages, and the area fraction occupied by vegetation during the initial stage of the development; for static land covers these fractions are equal; (4) the soil depth subject to evapotranspiration; for the vegetation cover, two soil depths are defined accordingly to the development stage of the vegetation: the initial stage and the mid-season and late season crop development stage; for bare soil a depth of 15 cm subject to evaporation is assumed; (5) the basal crop coefficients, for initial, middle and late seasons of the vegetation cover; these depend on the vegetation height, the air relative humidity, the wind speed, and the fraction of land surface covered by the vegetation; (6) the first day of the initial stage, and the length of each crop growth stage: initial stage length, crop development length, mid-season length and late-season length; (7) threshold values for the minimum amount of water stored in the soil that allow the effective evapotranspiration to occur at the maximum rate, both for the vegetation cover and for the bare soil.

The deep percolation is calculated depending on the soil saturated hydraulic conductivity and on the water that exists in the soil that can drain under the force of gravity. This aspect also depends on the sequential water balance in the soil, which is different depending on the vegetation cover. Details about the methods to compute each one of the referred to processes may be consulted in Oliveira (2004a) and in Oliveira et al. (2008). Surface or total flow (F) of a river is mainly composed of (1) direct runoff or overland flow (Fd), produced in the watershed above the place where it is measured, resulting from precipitation that does not infiltrate into the soil surface and that is not retained (for example in the plants canopy, buildings, dams, etc.), and (2) baseflow (Fb), resulting from water that infiltrates into the soil, goes through the subsurface and eventually comes to the surface, being the discharge of groundwater to the watershed:

$$F = Fd + Fb \tag{4}$$

The hydrograph represents surface flow against time (Figure 3). The two large flow components of surface flow (Fd and Fb) may be separated in the hydrograph. Several methods exist (cf. e.g. Linsley et al. 1975). One of these consists in connecting total flow that exists in the beginning of the rising limb of a new direct runoff episode due to the occurrence of precipitation to the total flow that exists in the end of this direct runoff episode. Linsley et al. (1975) present the following equation to estimate time from the hydrograph peak to a point located in the end of the recession curve that reflects the end of direct runoff (A is watershed area above the measuring station in km^2 and n is number of days):

$$n = 0.8A^{0.2} \tag{5}$$

The hydrograph separation (HS) method, using a daily basis, was initially developed and programmed in the QuickBasic computer code DECHIDR (Oliveira et al. 1997) and later updated to Visual Basic 6.0 in DECHIDR_VB program (Oliveira 2001, Oliveira 2004a). It was developed for the estimation of groundwater recharge in the fractured massifs of Portugal, since knowledge about recharge in these regions was scarce, and data was available on surface flow and precipitation daily time series. The applications of DECHIDR_VB have been disseminated in several papers, for instance Oliveira (2004b) and Oliveira (2006).

The general technique for the separation followed the method represented in Figure 4. The method consists of plotting a straight line linking the hydrograph origin of the precipitation/total flow (P/F) episode under analysis with total flow calculated in the beginning of day $n + 1$. Day n [computed with equation (5)] refers to the number of days with direct runoff after the hydrograph peak (Figure 4A) or the end of the precipitation if this exceeds the hydrograph peak (Figure 4B).The area above the line represents direct runoff of the episode under analysis while the area below the line represents its baseflow.

The HS turns out to be a more complex process due to the occurrence of different superimposed episodes, which can result in the recession of several P/F episodes in the same day. To deal with this situation, a set of procedures was developed in order to isolate distinct P/F episodes. The separation is carried out sequentially considering the input data series: date, total flow and precipitation. Oliveira (2004a) and Oliveira et al. (2008) describe these procedures.

The advantages of the HS method in estimating recharge are: (1) it is easy to apply with commonly available precipitation and flow data; (2) it only requires the definition of two parameters (1- the number of days in which there is direct runoff, and 2- the precipitation threshold - if this parameter is considered); (3) it is not constrained to fixed parameters of the watershed because each P/F episode is considered separately; (4) it is able to control and maintain the mass balance between precipitation and the produced total flow; (5) it integrates all the processes of the hydrological cycle that take place in the watershed, measuring the response of the system to those processes; (6) it is applicable to the whole watershed, not requiring the definition of recharge and discharge areas of the groundwater medium.

The following limitations are referred to: (1) it is vulnerable to errors in the determination of total flow; (2) it is dependent on the quality of the estimation of precipitation in the watershed, mainly if the balance between precipitation and total flow is used; (3) it considers that streams are only receiving bodies (does not consider bank storage) and that all groundwater discharges to those streams derive from inside the watershed; (4) it may not be used directly if there are dams that inhibit natural flow.

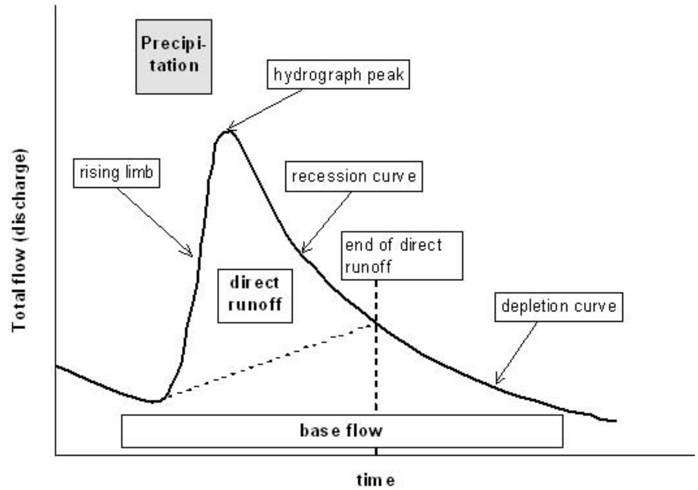

FIGURE 3: *Surface flow hydrograph and separation of surface flow into direct runoff and baseflow.*

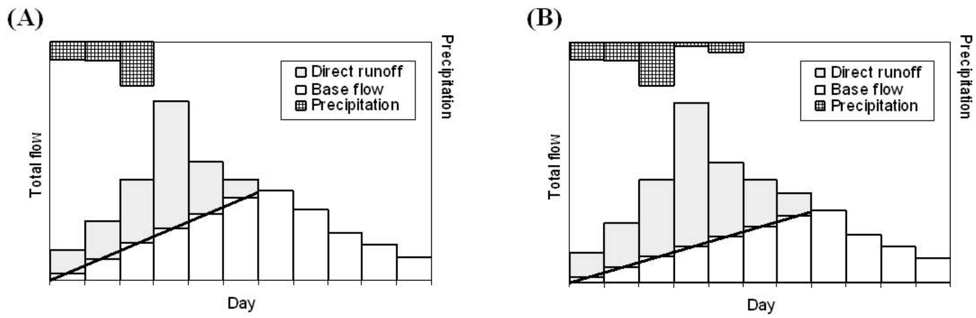

FIGURE 4: *Example of the hydrograph separation process, for n = 2 day, using as criterion (A) the day of the hydrograph peak, (B) the last precipitation day.*

Baseflow is an estimator of the recharge that occurs in the area defined by a watershed. In between the occurrence of the recharge process and the subsequent baseflow, the process of discharge from the saturated groundwater medium to the surface medium must be considered. Baseflow is a measure of the groundwater medium discharge to the surface medium if: (1) there is no storage of surface water; (2) there is no evaporation of surface water; and (3) there is no abstraction of surface water. On the other hand, groundwater discharge may translate the groundwater recharge if: (1) recharge is the only water source of the saturated medium; (2) there is no abstraction of groundwater; (3) all the water that leaves the saturated zone flows to the surface medium; and (4) there is no evapo(transpi)ration from groundwater.

3 SNOWMELT

In many temperate regions of the world, snowmelt represents an important source of water and can, to varying degrees, change the patterns observed in hydrographs. Moreover, the chemical properties of meltwater change over the snowmelt period. Tracer studies are often used to illuminate the origin and principal flow paths of water in a basin (Sueker et al. 2000). In the case where information on chemical tracers is not available, simple energy-budget equations may be used to predict the quantity and timing of snowmelt in specific watersheds. The snowmelt model calculates precipitation as rain, new snow, snowpack, and ultimately meltwater. The water available for movement as baseflow or surface flow (meltwater plus precipitation as rain) is calculated on a daily basis and imported into DECHIDR_VB in order to obtain values for baseflow.

Snowmelt depends on the balance of a series of energy sources and sinks, including shortwave and longwave radiation, convection from the air (sensible energy), vapour condensation (latent energy), conduction from the ground, and the energy contained in rain (USACE 1998). Once snow melts, it can follow the same flow paths as rain. Thus, snowmelt calculations can effectively be separated from the processes related to the movement of the snowmelt into the river network.

The generalized energy-balance snowmelt equations are often simplified based on certain meteorological or forest-cover conditions. In general, snowmelt can be divided into two types: rain-free and rain-on-snow. Equations for rain-on-snowmelt can be greatly simplified because solar radiation can be considered a relatively minor energy input. Snowmelt equations depend on the categories of forest density. These equations can be consulted in USACE (1998), and are also reproduced in Yarrow and Oliveira (2006).

The snowmelt model follows the flowchart in Figure 5. It is important to note that snow is measured in units of water equivalent (mm) so as not to introduce further uncertainty as to snow depths and snowpack dynamics. Using information about temperature and precipitation, new snow is calculated daily. A daily snow balance equation is used to calculate the evolution of the snowpack over time.

To run the hydrograph separation method, the daily snowmelt and the precipitation as rain are summed to get the daily water available for infiltration (eventually baseflow) and direct runoff. In basins where snow represents an important percentage of the annual precipitation, the unmodified use of the hydrograph separation method could lead to erroneous conclusions. Thus, it is important to consider how snowmelt might influence the hydrographs of the associated river and how to account for this process in the hydrograph separation method. Without accounting for snowmelt, in periods without precipitation, baseflow would be overestimated because total flow would be considered as baseflow originating from the last precipitation episode (Yarrow and Oliveira 2006). During the spring snowmelt season, one could expect hydrograph peaks due only to snowmelt.

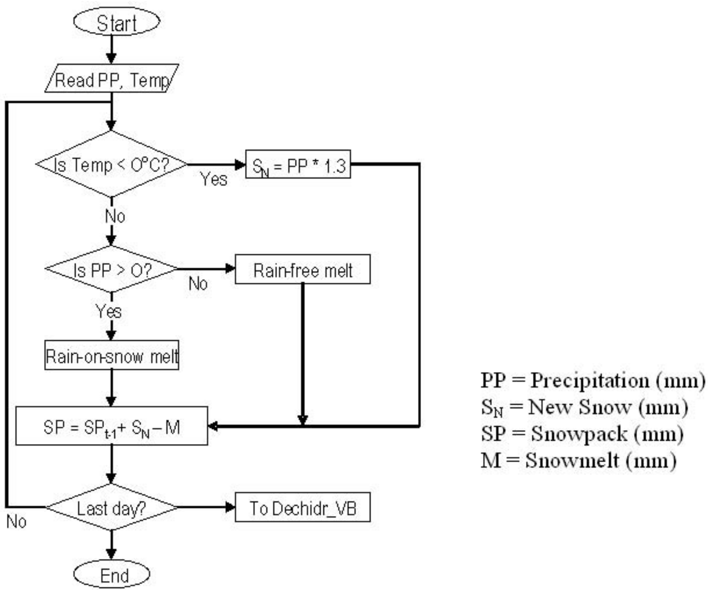

FIGURE 5: *Flow chart for Snowpack/snowmelt model.*

4 CONCLUSIONS

The choice of a model or method to compute recharge derives from the conceptualization of the recharge process of a study area, and also on the available data. Recharge methods may be classified as predictive, when the mass balance is carried out above the saturated zone or as response methods, when the mass balance is made for the saturated zone.

Concerning the two kinds of methods presented here, the **soil daily sequential water balance** method, a predictive one, requires knowledge of the climatic data to characterize precipitation and reference evapotranspiration, and knowledge of medium characteristic parameters, that depend on the complexity of the selected model. The use of more complex models, such as the described BALSEQ_MOD model, has the advantage of more closely approximating the processes that occur in natural systems. However, sometimes it may be difficult to characterize all the parameters required for these models, and the insufficient characterization of these parameters may produce erroneous estimates of recharge.

The **surface flow hydrograph separation method**, a response method, is probably the easiest method for recharge calculation, as it does not require medium characteristic parameters, and only requires knowledge of daily precipitation and flow series.

A snow-melt model, although requiring the knowledge of a few land cover and climatic data, that are not always readily available, can be coupled to the hydrograph separation method, in order to allow the application of this method for situations where snow occurs. This snow-melt model can also be coupled with the soil daily sequential water balance method.

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GROUNDWATER VULNERABILITY TO POLLUTION AND TO SEA WATER INTRUSION IN COASTAL AQUIFERS

J.P. LOBO FERREIRA AND M.M. OLIVEIRA

1 INTRODUCTION

In integrated coastal zone management, groundwater plays an important role as it discharges to the rivers, estuaries, and sea, contributing with fresh water to these systems and transporting dissolved substances. These substances may be natural, due to the chemical reactions of the water with the underground medium, or may result from man-originated pollutants. These pollutants exist on the land surface, and are transported to the groundwater medium by the infiltrating surface water (including rainfall) that recharges the aquifer medium and, after flowing in the aquifer, during a period that may range from days to thousands of years, discharges to the surface medium. In the underground medium, the pollutant load may be reduced, due to the underground medium properties that react with the polluted groundwater, and the discharged groundwater may present better chemical characteristics. Also, the opposite direction may be considered, i.e. when instead of groundwater discharging to the surface water bodies, it is the surface water bodies that infiltrate or move into the groundwater body. This is the case of the marine salt-water intrusion due to the equilibrium between the groundwater medium and the sea-water medium. This equilibrium may be affected by the pumping of groundwater that induces saline water flow from the sea into the aquifer.

The groundwater vulnerability may be considered an indication of the likelihood that a groundwater body may be contaminated either by surface water infiltration or by the movement of the sea-water intrusion interface. On the other hand it gives an indication of the likelihood that a pollutant may be retained in the groundwater medium, thus influencing the quality of the groundwater discharged to the surface medium. Thus, groundwater vulnerability assessment is a measure to protect groundwater. Several works have been made in order to access mainly groundwater vulnerability to pollution. This chapter gives a general overview of some groundwater vulnerability assessment methodologies, presents the concepts and describes with more detail some techniques, one of which was applied in the ECOMANAGE Project.

2 CONCEPT OF VULNERABILITY TO POLLUTION

According to Lobo Ferreira and Cabral (1991) it is believed that the most useful definition of vulnerability is one that refers to the intrinsic characteristics of the aquifer, which are relatively static and mostly beyond human control. It is proposed therefore that the groundwater vulnerability to pollution be defined, in agreement with the conclusions and recommendations of the international conference on ”Vulnerability of Soil and Groundwater to Pollutants”, held in 1987 in The Netherlands, as (Duijvenbooden et al. 1987): ”*the sensitivity of groundwater quality to an imposed contaminant load, which is determined by the intrinsic characteristics of* *the aquifer*". Thus defined, vulnerability is distinct from pollution risk. Pollution risk depends not only on vulnerability but also on the existence of significant pollutant loading entering the subsurface environment. It is possible to have high aquifer vulnerability but no risk of pollution, if there is no significant pollutant loading; and to have high pollution risk in spite of low vulnerability, if the pollutant loading is exceptional. It is important to make clear the distinction between vulnerability and risk. This because risk of pollution is determined not only by the intrinsic characteristics of the aquifer, which are relatively static and hardly changeable, but also on the existence of potentially polluting activities, which are dynamic factors which can in principle be changed and controlled.

Considerations on whether a groundwater pollution episode will result in serious threat to groundwater quality and thus to its (already developed, or designated) water supply are not included in the proposed definition of vulnerability. The seriousness of the impact on water use will depend not only on aquifer vulnerability to pollution but also on the magnitude of the pollution episode, and the value of the groundwater resource. Given the definition of vulnerability proposed as above, it is important to recognise that the vulnerability of an aquifer will be different for different pollutants. For example, groundwater quality may be highly vulnerable to the loading of nitrates at the surface, originated in agricultural practices, and yet be little vulnerable to the loading of pathogens.

In view of this reality, it is scientifically most sound to evaluate vulnerability to pollution in relation to a particular class of pollutant, such as nutrients, organics, heavy metals, pathogens, etc., i.e. to create specific vulnerability maps. This point of view has been expressed by other authors (e.g. Foster 1987), and some work has been done in specific vulnerability mapping. An example is the work of Canter et al. (1987) for nitrate pollution of agricultural origin. Alternatively, vulnerability mapping could be performed in relation to groups of polluting activities (Foster 1987), such as unsewered sanitation, agriculture, and particular groups of industries. This has been attempted for some activities. An example is the work of Le Grand (1983) for waste disposal. Although it is recognised that this specific vulnerability mapping is scientifically sounder, one must realise that there will generally be insufficient available data to perform specific vulnerability mapping. Therefore, it is necessary to adopt a mapping system that is simple enough to apply using the data generally available, and yet is capable of making best use of those data in a technically valid and useful way. Various such systems of vulnerability evaluation and ranking have been developed and applied in the past. Examples are Albinet and Margat (1970), Haertl (1983), Aller et al. (1987), and Foster (1987). Some of the systems for vulnerability evaluation and ranking include a vulnerability index which is computed from hydrogeological, morphological and other aquifer characteristics in some well-defined way. The adoption of an index has the advantage of, in principle, eliminating or minimising subjectivity in the ranking process.

The same definition provided to the aquifer system may be given to a groundwater pumping well. In the case of the exploitation of an aquifer system, one may be interested on knowing how vulnerable a well in terms of water pollution is. The vulnerability of a producing well to pollution could be defined as "*the sensitivity of the pumped groundwater quality to an imposed contaminant load, which is determined by the intrinsic characteristics of the aquifer that contribute with flow to the well*". A well corresponds to a 1D part of the aquifer system where it is installed, and hence the same methodology for the aquifer system could be applied. However a well is pumping water only at specific depths. Hence, it is important to define the part of the well which is screened and is pumping the water. In the case of coastal aquifers the sea-water may be regarded as a contaminant load that, in the case of aquifer exploitation, may contribute with non-fresh water to a pumping well. Chachadi and Lobo Ferreira (2001) handled this situation and presented the concept of groundwater vulnerability to sea water intrusion as "*the sensitivity of groundwater quality to an imposed groundwater pumpage or sea level rise or both in the coastal belt, which is determined by the intrinsic characteristics of the aquifer*".

3 METHODS FOR VULNERABILITY ASSESSMENT

3.1 Introduction

The pumping well vulnerability may be regarded as wellhead protection zoning, i.e. the vulnerable area directly depends on the travel time of water towards the pumping well, being this travel time a function of the hydraulic properties of the aquifer. On the other hand, the travel time could also be defined for a pollutant as a function of the retardation properties of the hydrogeological medium in relation to the pollutant or the decay properties of the pollutant. Often, the travel time is a defined value in legislation, and the need to know the retardation properties is disregarded. When the travel time is a defined value, several kinds of approaches may be considered, depending on the consideration of two or three dimensions of the domain. These approaches are based on the groundwater numerical modelling or on approximations that require hydrogeological data.

The simpler approach is the calculated fixed radius method. A more elaborated approach, requiring the hydraulic head knowledge was developed by Krijgsman and Lobo Ferreira (2001). These approaches are applicable in two dimensions horizontal domain. More elaborated approaches depend on numerical modelling, that allow two or three dimensional groundwater flow modelling with particle tracking (dependent on the numerical flow models). The WELLFLOW mathematical model allows the computation of travel time distances in a radial cross-section (vertical 2D, cf. Feseker and Lobo Ferreira 2001). These methodologies can also be found in Lobo Ferreira et al. (2004).

The aquifer vulnerability may be characterised as a function of the physical system properties in terms of more or less favourable conditions in the system for a pollutant load to contaminate the aquifer. Or it may be characterised using numerical flow modelling. The first approach may be considered a parametric one, in which different variables are characterised and put together to produce an index, as is the case of the DRASTIC index for a pollutant load occurring in the ground surface (cf. Aller et al. 1987) or the GALDIT index for the case of coastal aquifers in relation with marine water (cf. Chachadi and Lobo Ferreira 2001). The second approach, which may inclusively consider the flow in the unsaturated (vadose) zone, rely on the numerical flow models, and is related to the travel time of groundwater flow. This modelling may be coupled with the attenuation conditions of the medium that dictate the pollutant concentration on the aquifer system. In the next sections parametric methods are described in more detail: the DRASTIC index and the GALDIT index.

3.2 The DRASTIC index for the assessment of groundwater vulnerability to pollution

The index of vulnerability DRASTIC (Aller et al., 1987) was created for the following conditions: (1) the contaminant is introduced at the ground surface; (2) the contaminant is flushed into the ground water by precipitation; (3) the contaminant has the mobility of water; and (4) the area evaluated with DRASTIC is 100 acres (0.4 km^2) or larger. The index of vulnerability DRASTIC corresponds to the weighted addition of seven values corresponding to seven parameters that characterise the subsurface medium and its specificity: (1) Depth to the water (D); (2) Net Recharge (R); (3) Aquifer material (A); (4) Soil type (S); (5) Topography (T); (6) Impact of the unsaturated zone (I); (7) Hydraulic Conductivity (C).

A value between 1 and 10 to each parameter, except R for which the value ranges between 1 and 9, is attributed, depending on local conditions. High values correspond to high vulnerability. The attributed values are generally obtained from tables, which give the correspondence between local hydrogeologic characteristics and the parameter value. DRASTIC index is computed by:

$$DRASTIC = D_R D_W + R_R R_W + A_R A_W + S_R S_W + T_R T_W + I_R I_W + C_R C_W \tag{1}$$

where R = rating and W = weight. Thus, each parameter has a predetermined, fixed, relative weight that reflects its relative importance to vulnerability. The most significant factors have weights of 5 and the least significant a weight of 1. A second weight has been assigned to reflect the agricultural usage of pesticides. Table 1 shows the ratings to be assigned to each DRASTIC parameter and the weights for standard DRASTIC applications and for DRASTIC pesticide applications. In some cases a rating interval and a typical rating (in brackets) is shown, which represents, in the case of the rating interval, the values that the parameter can assume, depending for instance on the clay content, the weathering conditions, or the fissuring in the case of the aquifer media. The minimum value of the standard DRASTIC index is 23 and the maximum value is 226. Such extreme values are very rare, the most common values being within the range 50 to 200. Whereas the corresponding minimum and maximum values for pesticide DRASTIC index are 26 and 256 respectively.

Depth to the water (D) refers to the distance from the ground surface to the top of the aquifer. In the case of a non-confined aquifer, the top of the aquifer is given by the water table. In the case of a confined aquifer it is given by the base of the confining layer. The depth to water is an important factor in the evaluation of groundwater pollution vulnerability primarily because it determines the thickness of material through which a contaminant must travel before reaching the aquifer. The depth to water is also important because it provides an opportunity for oxidation and it may help to determine the contact time of pollutant with the surrounding media. In general, there is a greater chance for attenuation of pollutants to occur as the depth to the water increases because deeper aquifer tops imply longer travel times in the vadose zone with an exception that if the vadose media is fractured or karstified then the travel time is independent of the depth of the aquifer. The quantity of **net recharge (R)** represents the amount of water per unit area of land, which penetrates the ground surface and reaches the water table. This recharged water is thus available to disperse, dilute and transport a contaminant vertically into the vadose zone to the water table and horizontally within the aquifer. The greater the recharge, the greater the potential for groundwater pollution. However, at certain quantity of recharge the pollution event may in turn decrease due to dilution of the contaminant. **Aquifer media (A)** refers to a lithological unit that serves as an aquifer. Aquifer media plays an important role in dissipation and transportation of the pollutants once introduced into them. The parameters like effective porosity, grain size, clay contents and aquifer thickness are the four main characteristics that control dissemination and transportation of the contaminants in the aquifer. **Soil media (S)** refers to that uppermost portion of the vadose zone which is characterised by significant biological activity. In the DRASTIC classification, the soil media is referred to be the upper weathered zone of the earth which averages to a depth of two metres or less from ground surface. Eleven different soil types which were given ratings of between 1 and 10 were defined by Aller at al. (1987). The parameter **topography (T)** refers to the slope and slope variability of the land surface. Topography has an influence on the soil development and therefore has an effect on contaminant attenuation. The topography also determines the contact time of contaminants with the soil and in case of unconfined aquifers the topography can provide information about the groundwater gradients and velocities. Smaller ground slopes are always potential for groundwater pollution compared to steep slopes. In DRASTIC method percent slope is considered for rating topography. **Impact of the vadose zone media (I):** By definition, the vadose zone includes all the unsaturated media below the ground and above the water table, including the soil zone. As the soil was already considered in the S parameter, the I parameter of the DRASTIC method will refer only to the unsaturated media below the bottom of the soil layer and above the water table in case of unconfined aquifer. When evaluating a confined aquifer, the ”confining layer” must be treated as a vadose zone and be always assigned a rating of 1. The type of vadose zone media determines the attenuation characteristics of the material below the soil horizon and above the water table. Biodegradation, neutralisation, mechanical filtration, chemical reactions, volatilisation and dispersion are the processes that may occur within the vadose zone. The media also controls the path length and routing, thus affecting the time available for attenuation and the quantity of material found. The routing is strongly influenced by any fracturing present in the vadose media. **Hydraulic Conductivity of the aquifer (C):** The rate of contaminant movement in the saturated zone is also controlled by the rate of groundwater movement. This parameter is used to measure the rate of water flow in the aquifer. By definition the aquifer hydraulic conductivity is the ability of the aquifer to transmit water. The higher the conductivity the higher the rate of contaminant movement.

The relation between DRASTIC index and aquifer system vulnerability may be considered as: Very high vulnerability (>199); High vulnerability (160-199); Moderate vulnerability (120-159); Low vulnerability (<120). The DRASTIC index was applied to the Santos estuary area (Figure 1, Oliveira et al. 2005).

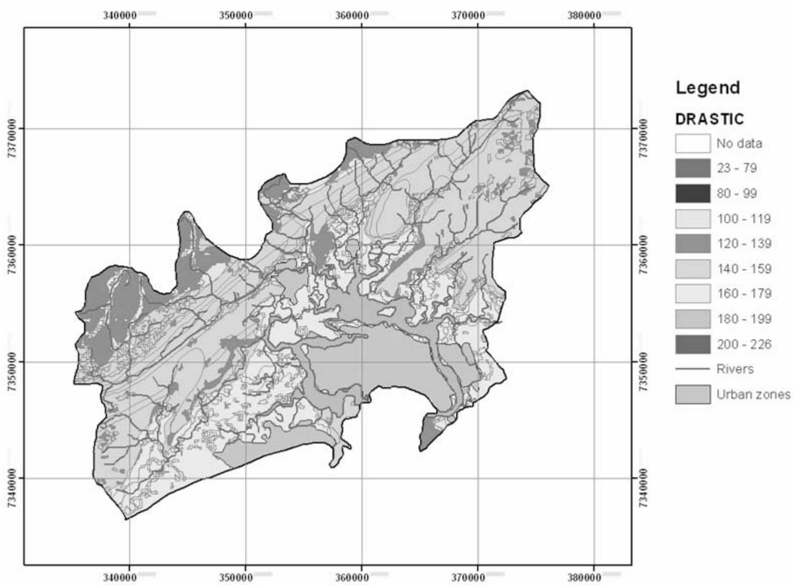

FIGURE 1: *Characterization of the DRASTIC vulnerability of groundwater to pollution in the Santos estuary area.*

3.3 The GALDIT index for the assessment of the coastal aquifer vulnerability to sea water intrusion

The description of the GALDIT index methodology is based on the work of Chachadi and Lobo Ferreira (2005, 2007), where more in detail information may be found. Inherent in each hydrogeologic setting is the physical characteristics that affect the seawater intrusion potential. After extensive discussions and consultations with the experts, academicians etc., the most important mapable factors that control the seawater intrusion were found to be: (1) Groundwater occurrence (aquifer type) (G); (2) Aquifer hydraulic conductivity (A); (3) Height of groundwater level above sea level (L); (4) Distance from the shore (D); (5) Impact of existing status of seawater intrusion (I); (6) Thickness of aquifer being mapped (T).

These factors, in combination, give the acronym GALDIT. They are determined to include the basic requirements needed to assess the general seawater intrusion potential of each hydrogeologic setting. GALDIT factors represent measurable parameters for which data are generally available from a variety of sources without detailed reconnaissance. The GALDIT system contains three significant parts: weights, ranges and importance ratings. Each GALDIT factor has been evaluated with respect to the other to determine the relative importance of each factor. The basic assumption made in the development of the tool is that the bottom of the aquifer(s) lies below the mean sea level. The GALDIT index is computed through the multiplication of the rating attributed to each parameter by its relative weight, adding up the six products and dividing the result by the sum of the weights (which is 15):

$$GALDIT = (G_R G_W + A_R A_W + L_R L_W + D_R D_W + I_R I_W + T_R T_W) \; / \; (G_W + A_W + L_W + D_W + I_W + T_W) \quad (2)$$

where R = rating and W = weight. The indicator weights depict the relative importance of the indicator to the process of seawater intrusion. After identifying the indicators, a group of people consisting of geologists, hydrogeologists, environmentalists, students, in-house experts was asked to weigh these indicators in the order of importance to the process of seawater intrusion. The feedbacks from all such interactions were analysed statistically and the final consensus list of indicators weights was prepared. The most significant indicators have weights of 4 and the least a weight of 1 indicating parameter of less significance in the process of seawater intrusion. The indicators must be considered as constants and may not be changed under normal circumstances.

Each of the indicators is subdivided into variables according to the specified attributes to determine the relative significance of the variable in question on the process of seawater intrusion. The importance ratings may assume the values: 2.5, 5, 7.5 or 10. Higher importance rating indicate higher vulnerability to seawater intrusion. Table 2 shows the classes and the ratings to assign to each parameter.

Groundwater occurrence (Aquifer type) (G): The extent of the seawater intrusion depends on the basic nature of groundwater occurrence. **Aquifer hydraulic conductivity (A):** This parameter is used to measure the rate of flow of water in the aquifer and hence to the sea. It is similar to parameter C of DRASTIC index. The magnitude of seawater front movement is influenced by the hydraulic conductivity of the aquifer. The higher the hydraulic conductivity the greater the inland movement of the seawater front. **Height of groundwater level above sea level (L):** This indicator determines the hydraulic pressure availability to push back the seawater front. The higher the water level in relation to sea level the lower the rating. **Distance from the shore (D):** The impact of seawater intrusion generally decreases as one moves inland at right angles to the shore. The maximum impact is witnessed close to the coast. Parameter D is measured in relation to the high tide. **Impact status of existing seawater intrusion (I):** The ratio $Cl^-/[HCO_3^- + CO_3^{2-}]$ was recommended by Chachadi and Lobo Ferreira (2001) as a criterion to evaluate seawater intrusion into coastal aquifers. Chloride is the dominant ion in the seawater and it is only available in small quantities in groundwater, while bicarbonate, which is available in large quantities in groundwater, occurs only in very small quantities in seawater. **Thickness of aquifer being mapped (T):** Aquifer thickness plays an important role in determining the extent and magnitude of seawater intrusion in coastal areas. The larger the aquifer thickness the larger the extent of seawater intrusion.

TABLE 1: *Ranges and ratings for the DRASTIC parameters (adapted from Aller et al. 1987).*

D - Depth to water (m) **(Weight - standard: 5; - pesticide: 5)**	**Rating**
< 1.5	10
1.5 - 4.6	9
4.6 - 9.1	7
9.1 - 15.2	5
15.2 - 22.9	3
22.9 - 30.5	2
> 30.5	1

A - Aquifer media **(Weight - standard: 3; - pesticide: 3)**	**Rating**
Massive Shale	1-3 (2)
Metamorphic/Igneous	2-5 (3)
Weathered Metamorphic/Igneous	3-5 (4)
Glacial Till	4-6 (5)
Bedded Sandstone, Limestone and Shale Sequence	5-9 (6)
Massive Sandstone or Massive Limestone	4-9 (6)
Sand and Gravel	4-9 (8)
Basalt	2-10 (9)
Karst Limestone	9-10 (10)

I - Impact of the vadose zone media **(Weight - standard: 5; - pesticide: 4)**	**Rating**
Confining Layer	1
Silt/Clay	2-6 (3)
Shale	2-5 (3)
Limestone	2-7 (6)
Sandstone	4-8 (6)
Bedded Limestone, Sandstone, Shale	4-8 (6)
Sand and Gravel with significant Silt and Clay	4-8 (6)
Metamorphic/Igneous	2-8 (4)
Sand and Gravel	6-9 (8)
Basalt	2-10 (9)
Karst Limestone	8-10 (10)

R - net Recharge (mm yr^{-1}) **(Weight - standard: 4; - pesticide: 4)**	**Rating**
< 51	1
51 - 102	3
102 - 178	6
178 - 254	8
> 254	9

S - Soil media (texture) **(Weight - standard: 2; - pesticide: 5)**	**Rating**
Thin or Absent, Gravel	10
Sand	9
Peat	8
Shrinking and/or Aggregated Clay	7
Sandy Loam	6
Loam	5
Silty Loam	4
Clay Loam	3
Muck	2
Nonshrinking and Nonaggregated Clay	1

T - Topography (slope in %) **(Weight - standard: 1; - pesticide: 3)**	**Rating**
< 2	10
2-6	9
6-12	5
12-18	3
> 18	1

C - Hydraulic conductivity (m d^{-1}) **(Weight - standard: 3; - pesticide: 2)**	**Rating**
< 4.1	1
4.1 - 12.2	2
12.2 - 28.5	4
28.5 - 40.7	6
40.7 - 81.5	8
> 81.5	10

TABLE 2: *Ratings for the GALDIT parameters (Chachadi and Lobo Ferreira 2005, 2007).*

G - Groundwater occurrence (Aquifer type) **(Weight: 1)**	**Rating**
Confined Aquifer	10
Unconfined Aquifer	7.5
Leaky confined Aquifer	5
Bounded Aquifer (recharge and/or impervious boundary aligned parallel to the coast)	2.5

A - Aquifer hydraulic conductivity (m d^{-1}) **(Weight: 3)**		**Rating**
High	>40	10
Medium	10-40	7.5
Low	5-10	5
Very low	<5	2.5

L - Height of groundwater level above sea level (m) **(Weight: 4)**		**Rating**
High	<1.0	10
Medium	1.0-1.5	7.5
Low	1.5-2.0	5
Very low	>2.0	2.5

D - Distance from the shore (high tide) (m) **(Weight: 4)**		**Rating**
Very small	<500	10
Small	500-750	7.5
Medium	750-1000	5
Far	>1000	2.5

I - Impact status of existing seawater intrusion - Range of Cl^- / [HCO_3^- + CO_3^{2-}], ratio in epm in ground water **(Weight: 1)**		**Rating**
High	>2	10
Medium	1.5-2.0	7.5
Low	1-1.5	5
Very low	<1	2.5

T - Aquifer thickness (saturated) (m) **(Weight: 2)**		**Rating**
Large	>10	10
Medium	7.5-10	7.5
Small	5-7.5	5
Very small	<5	2.5

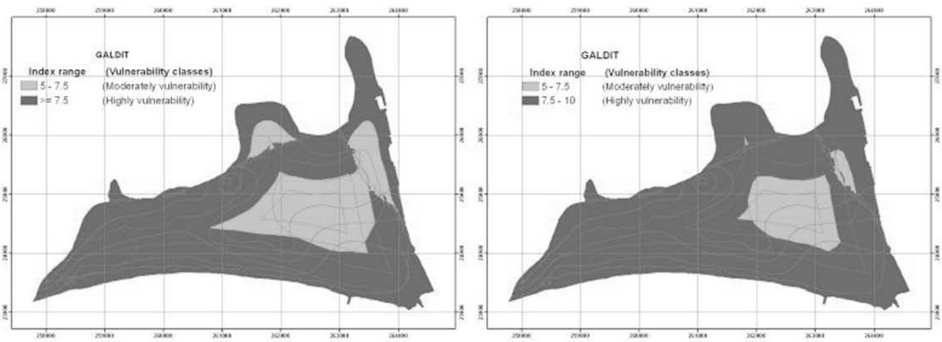

FIGURE 2: *Computed GALDIT index for today's sea level (left) and for sea level rise of 0.5 m (right).*

Once the GALDIT index has been computed it is possible to classify the coastal areas into categories of seawater intrusion vulnerability as follows (Chachadi and Lobo Ferreira 2005, 2007): High vulnerability ($\geq$7.5), Moderate vulnerability (5 to 7.5), Low vulnerability ($<$5). The GALDIT index methodology was firstly applied to the Bardez aquifer in Goa, India, first using a former methodology, with weights and importance ratings different from the actual, (Chachadi and Lobo Ferreira 2001, 2002), and with the actual methodology (Chachadi and Lobo Ferreira 2005, 2007).

This methodology was also applied to the Monte Gordo aquifer system in South-eastern Portugal Algarve region, considering three different scenarios regarding the parameter L: today's sea level, sea level rise of 0.25 m, and sea level rise of 0.5 m. Figure 2 shows the computed GALDIT index for the today's sea level scenario and for the sea level rise of 0.5 m scenario. The three scenarios application show how important it is to assess on due time the impact of sea water level rise due to climate changes. These results are also important to observe the negative effects of overexploitation of aquifers, which affects regional groundwater level, causing in coastal zone salt water intrusion. The same study considering the effects of sea water rise in the Bardez aquifer in Goa, India is available in Chachadi et al. (2002).

4 CONCLUSIONS

The vulnerability mapping can be used as a tool for the management of the land use in terms of groundwater protection (DRASTIC index) or for the management of the coastal groundwater resources (GALDIT index). The GALDIT applications can also be made for the island aquifers so that optimal management practices can be evolved for groundwater use. The maps can be prepared using GIS or if the area is small, point values of the vulnerability indices can be obtained and then contoured using a contour drawing program to get a vulnerability score map. The point values of GALDIT index can be used in ascertaining the wellhead protection areas in the coastal belts to prevent seawater mixing.

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MATEDIT: A SOFTWARE TOOL TO INTEGRATE INFORMATION IN DECISION MAKING PROCESSES

N. BURBA, E. FEOLI AND M. MALARODA

1 INTRODUCTION

In this era of omnipresent Information and Communication Technology (ICT), the availability of software tools is immense, however to find the one tailored to satisfy all our specific needs is almost impossible. This is the reason why, instead of spending time in searching for the "perfect software", it is better to spend time to write directly the software that does exactly what we want it to do. MATEDIT is a product of this philosophy. It is non-commercial software that extracts information from data matrices. The acronym stands for MATrix and EDITing; while the term matrix is obvious, the term editing deserves an explanation. It is there to cope the idea that the matrices of ecological data have information that has to be enlighten in a process that is analogous to that of the editor of a book (or a paper) when she/he is keeping the decision to publish or not the book (or the paper).

The current version of MATEDIT was mainly developed as a free software tool within the project ECOMANAGE as a mathematical tool of general applicability suitable both for Integrated Assessment (IA) and for transdisciplinary actions (TA). IA and TA are considered today the fundamental steps in decision making processes. Rotmans and van Asselt (2001) defined IA as "an interdisciplinary process of structuring knowledge elements from various scientific disciplines in such a manner that all relevant aspects of a complex societal problem are considered in their mutual coherence for the benefit of decision-making", while Ahern (2005) considers TA (which he calls transdisciplinary modelling), as the process that integrates planning with research, "enabling the multidimensional challenge of sustainability to be understood more rigorously with many disciplines involved, and with the involvement of the public (i.e. stakeholders, elected officials)". Ahern (2005) also stresses that the level of TA has become a key indicator of rigorous sustainability.

2 MATEDIT

MATEDIT was designed to structure data matrices (or data tables) in order to show relationships between objects and variables. Structuring data matrices, also called block clustering, may be achieved in different ways and it is the basis to classify simultaneously variables and objects (Podani and Feoli 1991, Miklos et al. 2005). MATEDIT is an expanded WINDOWS version of an old one written for DOS. The first version is the only officially published (Burba et al. 1992), the current one has been submitted (Burba et al. 2007) to the international journal "Community Ecology". MATEDIT is based on similarity theory, a theory that is emerging independently within different disciplines and it is considered under different perspectives by several authors, also within the Artificial Intelligence (AI) research framework (e.g. Stull 1988, Rusthon 1988, Zilitinkevich et al.1998, Janowicz 2006, Jia-Rong et al. 2007).

MATEDIT is based on fuzzy set theory (Zadeh 1965, Zimmerman 1996), whose fundamental axiom is that "any element, of a universe real or imaginary, may be considered as belonging to several sets with a degree of belonging to each set". MATEDIT produces fuzzy sets once a set of data elements (objects or variables) are grouped in classes. The degrees of belonging of the elements are calculated on the basis of the similarity they have with the classes defined in a classification process. MATEDIT compares and tests with permutation techniques (Good 1994, Manly 1997, Pillar 1996) the significance of similarity between objects and/or variables and the sharpness of classifications of the same sets of elements (objects or variables) based on different sets of characters (in case of objects) or objects (in case of variables) and/or different similarity functions and clustering methods. MATEDIT compares also different classifications of the same objects or variables by continency tables according to the method suggested by Estabrook and Estabrook (1989).

MATEDIT implements only few similarity functions among those most commonly used, but includes some similarity functions that are rarely found in other software tools, such as the index of Gower (Gower 1971) and the probabilistic similarity index of Goodall (Goodall 1964, Goodall 1966, Goodall et al. 1991, Goodall 1993). All the functions are adapted to consider missing values. MATEDIT can import matrices from OpenOffice Calc and Microsoft Excel and can export data directly to Excel or in free format. MATEDIT performs hierarchical cluster analysis with the three fundamental criteria: single, average and complete linkage (Sneath and Sokal 1973) and extracts eigenvalues and eigenvectors from squared matrices. It performs other simple matrix operations such as matrix sum, differences and multiplication, and also it performs product of the elements of a matrix by the corresponding elements of other matrices of the same size. This is useful for weighting variables or criteria both for comparisons between objects and for multicriteria evaluation (Saaty 1980, Saaty 1999). MATEDIT produces dendrograms and scattergrams and allows to produce subsets of the data matrices by clicking on the dendrogram branches and by delimiting clusters drawing closed lines around points with the cursor in the scattergrams. It rearranges rows and columns of a data matrix according to the dendrogram sequences or other sequences given by the user, so it is very useful for structuring data tables and to obtain concentration values in the "blocks" corresponding to the sub-matrices of structured tables (Feoli and Orloci, 1979). It standardises and transforms variables by many ways and normalises rows and column vectors. It also gives ranks to variables by the sum of squares criterion. It reduces the data matrix by grouping column and row vectors according to a specified dendrogram classification or clusters isolated from the scattergrams or by other external grouping criteria.

The scoring in the groups can be done in several ways (sum, average, minimal value, maximal value, deviation from expectation), to produce degrees of belonging of objects or variables to sets. MATEDIT includes options for identification purposes (supervised classification) by assigning new objects or variables to clusters already established using the available similarity functions and all the three criteria implemented in the program namely single, average and complete linkage) or by using the neural network algorithms of the public domain software tool

SNNS (Java/Stuttgart Neural Network Simulator) to which it is interfaced (`http://www-ra.informatik.uni-tuebingen.de/SNNS/`). MATEDIT has also the possibility to use a script language in format XML that allows making in one run different procedures of data analysis. Based on the capabilities listed above MATEDIT can be considered a flexible decision support system with many capabilities not present in other (commercial) software for DSS and SDSS (e.g. DEFINITE, Janssen and Herwijnen 1994).

3 MATEDIT FOR MULTICRITERIA EVALUATION

MATEDIT can be used to build the matrix ALTERNATIVES/EFFECTS in a participatory approach that can accomodate discussions and brainstorming with stakeholders. Once the matrix is obtained on the basis of consensus among different ”participants” in the decision process, MATEDIT ranks the alternatives according to their similarity to the alternative that is considered the best in terms of the effects (Malczewski 1999).

Several options are available, the easiest to understand is the one based on the transformation of the scores of the effects according to the formula (xi-xmin)/(xmax-xmin) and on the comparison of the alternatives with the vectors representing the worst and best alternatives by scoring the effects as zero or one depending on whether they represent costs or benefits. Many different tables can be obtained introducing the uncertainty values and weights according to the interaction among the stakeholders involved in the decision making process.

4 CONCLUSION

Integrated Coastal Zone Management (ICZM) requires multidisciplinary data collection, interdisciplinary data analysis and transdisciplinary knowledge integration. MATEDIT is a mathematical software tool that can help in all the three phases. It can be used to manipulate the data stored in matrices by applying the matrix algebra, useful for representing the multidimensional spaces as defined by matrices describing landscape states or alternatives in terms of the effects (Multicriteria analysis). These spaces may be represented both in terms of eigenvectors (e.g. Principal component analysis, Orloci 1978) and/or fuzzy sets (Feoli and Zuccarello 1986, 1988).

ICZM can be seen in the perspective of ”deliberation matrices” (O’ Connor 2000, 2004). In this context MATEDIT can be used to compare the matrices produced in the deliberation procedure, being capable to compare the matrices of the ”deliberation cube” in all the three dimensions. These dimensions are: the classes of stakeholders (one dimension of the cube) judging a set of scenarios, the set of scenarios (second dimension of the cube) described by a set of key governance or decision issues (third dimension of the cube). MATEDIT is a tool that can help the development and the application of the concept of increasing policy integration in ICZM since the tabular (matrix) rearrangement and the ordination scattergrams produced by MATEDIT can be easily understood by the stakeholders in participative planning activities (Cundill et al. 2005).

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SPATIAL DECISION SUPPORT SYSTEM (SDSS) FOR INTEGRATED COASTAL ZONE MANAGEMENT (ICZM)

E. FEOLI AND M. SCIMONE

1 INTRODUCTION

Coastal zones are among the most complex spatial systems where to take decisions. To cope with problems related to conflicts and/or find optimal solutions, Fedra and Feoli (1998) propose the integration of Geographic Information Systems (GIS), Image Processing Systems (IPS), Remote Sensing (RS), with Data Management, Data Analysis and Modelling, as a necessary exercise. Definitions of decision support systems (DSS) range from: ”interactive computer based systems that decision makers utilize with data and models to solve unstructured problems” (Gorry and Morton 1971) ”or semi-structured problems” (Malczewski 1999), to ”any system that makes some contribution to decision making” (Sprague and Watson 1986). We can say that decision support systems are the results of the ”approaches for applying information systems technology to increase the effectiveness of decision makers in situations where the computer can support and enhance human judgement in the performance of tasks that have elements which cannot be specified in advance” (Sol 1983). Spatial decision support systems (SDSS) are special cases of DSS. They are meant to optimally locate in space human activities (e.g. cropping, logging, selling, etc.) and manufactures (buildings, infrastructures etc.) that can be called for brevity ”land uses” according to some specific objectives. The application of SDSS should follow a decision making process that already selects the type of land use among different alternatives.

The concept of suitability is central in DSS and SDSS applications. According to this concept, the choice done is not the best in absolute terms, but just the best in the relative context in which DSS and SDSS are applied. For this reason, the ultimate objective of a computer based decision support system (DSS or SDSS) is merely to improve the decision making processes by providing useful and scientifically sound information to the actors involved in these processes, and tools that can flexibly and easily deal with such information in iterative and recursive way. DSS and SDSS belong to the ”domain of decision making” that is given by a network of actors (stakeholders) whose main nodes are: decision and policy makers, investors, entrepreneurs, the ”general public” and the scientific/ technical institutions that can offer knowledge and tools to support the decision making. According to Fedra and Feoli (1998) in this network where ”problem owners and various actors in the decision making process have a central role; supporting their respective tasks requires man-machine interfaces that are easy to use and easy to understand: the paradigm of the thematic map offered by GIS is a powerful tool for this purpose. . . ” an effective decision support system must first of all provide a common, shared information basis, framework and language for dialogue and negotiation. Fedra and Feoli (1998) stated also that ”an information system that can cater to all these needs must be based on more than good science and solid engineering. The need for better tools to handle ever more critical environmental and resource management problems is obvious, and the rapidly developing field of information technology can provide the necessary machinery. The biggest challenge, however, seems to be the integration of new information technologies and more or less mature formal methods of analysis into institutional structures and societal processes, that is, putting these tools to work in practice". This contribution is in line with these sentences suggesting a unifying view of DSS and SDSS under the perspective of Multi Criteria Evaluation (MCE).

2 A UNIFIED VIEW OF DSS AND SDSS

The theoretical and technical description of DSS and SDSS have been extensively and adequately covered in the literature (Geoffrion 1983, Craig and David, 1991, Densham, 1991, Goodchild and Densham, 1990, Moon, 1992, NCGIA 1992, Paruccini, 1994, Malczewski 1999 and references therein). Concerning the technical aspects, DSS and SDSS have in common the following components: Data bases, Data analysis, Modelling and Expert systems, Simulation and Optimisation, Multicriteria Evaluation (MCE) and User Interface. However SDSS requires one more component that is known as Geographic Information System (GIS).

2.1 DSS and definition of alternatives

The word "alternative" is the key word in the domain of decision making. A decision may be defined as a choice between alternatives. The alternatives may represent different options of action following different hypotheses among which a choice is desirable based on some specific criteria. To establish alternatives is the first step in a decision making process. Even if a scenario of socio-economic development is delineated the scenario may present a set of possible alternatives that are difficult to be objectively evaluated and ranked according to the needs of the people. The ranking of alternatives is a procedure that requires comparative analysis in terms of costs and benefits. The questions: who pays the costs and who receive the benefits are fundamental to be answered within the "domain of decision making". The evaluation and ranking the alternatives follows always a hierarchical process that can be logically and scientifically codified (Saaty 1980, 1999). An example of a hierarchical process is given in Table 1. Different scenarios can be built to describe the possible socio-economic development of a given area on the basis of different alternatives: e.g. development based on industrial sector or development based on tourism. A first application of DSS could be done in order to rank the scenarios as alternatives. Once a rank is obtained for "industry " or "tourism", a set of alternatives considered within each scenario (e.g. converting existing industries, building new industry etc.) have to be ranked .

2.2 DSS and definition of criteria

In any DSS application criteria are the factors that act positively or negatively in favour of a specific alternative. The ranking of alternatives is conditioned by the choice of criteria and by the weight given to them. For that reason the participatory approach is necessary to ensure the democratic discussion and the respect of the equity principle. It is obvious that the criteria are determined on the basis of the objectives to be achieved. If for example the objective is sustainable development, criteria are to be chosen within socio-economic and ecological parameters (indicators). In theory, there is no limit for the choice of the criteria. In terms of the DPSIR model the criteria are based on the State - Impact indicators. They are always determined by experts who act on the basis of their experience with or without the analysis of available data and/ or modelling and in consultation with stakeholders and interested public. Also the choice of the criteria is always based on cost-benefit considerations and has always a strong component of subjectivity.

2.3 DSS and the effects/alternatives matrix

Once a set of criteria is defined, they have to be evaluated or measured in terms of the effects the different alternatives have on the state of the corresponding indicators (variables). For example, if biodiversity is a criterion to be used, the effects, negative or positive (costs and benefits) the alternatives may have on it, have to be measured. The estimated number of species lost can be a measure of the effect (cost) of the alternatives on biodiversity. When scientific/technical reports, databases, indicator tables, etc. are available, or when it is possible to apply models (e.g. predicting soil erosion, biomass productivity, etc.), for some criteria the effects may be measured by ratio/interval scale, however for the majority of the effects expert judgement in ordinal scale is the only possibility. The first step in any decision process is the construction of the matrix effects/alternatives. This matrix should be built once all the alternatives and effects have been well defined. The matrix consists of m effects and n alternatives, where the x(i,j) scores represent the effects of the j-th alternative on the i-th criterion (variable). Table 2 represents such a matrix. This is the basis to compare the alternatives and the effects using matrix algebra.

TABLE 1: *Example of a hierarchical structure to be submitted to DSS in order to rank the different alternatives.*

Scenario level	**Industry**	**Tourism**
Alternatives	(1) Converting existing industries	(1) Improving existing resorts
	(2) Building new industries	(2) Building new resorts
	(3) Improving infrastructures	(3) Improving infrastructures
	etc…	etc…

TABLE 2: *The matrix effects/alternatives for three alternatives.*

	Alternative 1	**Alternative 2**	**Alternative 3**
Effect a	X(a,1)	X(a,2)	X(a,3)
Effect b	X(b,1)	X(b,2)	X(b,3)
Effect c	X(c,1)	X(c,2)	X(c,3)
etc…	…	…	…

2.4 SDSS and definition of the alternatives (Operational Geographic Units, OGUs)

Once chosen the best scenario the question to answer with SDSS is: Where to optimally locate the selected ”land uses” (that were alternatives defined and selected in a previous DSS). For this, it is essential to use a geographic information system (GIS) where all the necessary cartographic information are stored. The first step in the use of GIS for the SDSS is the definition of the operational geographic units (OGU) where to locate the selected ”land uses” (industries, markets, roads, airports, harbours, hotels etc.). The OGUs are the alternatives to be ranked in SDSS. According Feoli and Zuccarello (1996) OGUs can be of different nature and size: pixels of remote sensing images (Landsat, SPOT etc.), administrative units, land cover patches or arbitrary cells of a net (grid, triangulated network, quadtree pattern, hexagonal mesh, thiessen polygons, etc., (see Peuquet 1984, Skole et al. 1993) that are superimposed to a map of a given territory.

2.5 SDSS and definition of the criteria

In SDSS the criteria are factors that act in favour of suitability of a given OGU to host the selected ”land use” and constraints that deny the suitability. The most important key-word in SDSS is *suitability* and the most important tool is the *suitability map*. The GIS component of SDSS is essential to produce suitability maps. Factors on which to rank the OGUs are spatial variables: topographical, geo-morphological, geographical, ecological etc. measured by values of distances (e.g. distance by the rivers, by the roads, by urban areas etc.), degrees of inclination of slopes, aspect, by meters above the sea level etc. or by the quality of the position (e.g. within a valley, on the top of a mountain etc.), or by measures of risks of various nature (soil erosion, water flooding, fire, etc.). It follows that SDSS deals with matrices describing the OGUs with spatial variables representing the factors. Table 3 is such an example.

2.6 Multi Criteria Evaluation (MCE)

Multicriteria Evaluation (MCE) is applied both in DSS and SDSS on the basis of the same theoretical background, it can be defined as a set of techniques that can be used to combine a set of criteria to achieve a single composite basis to select among alternatives according to specific objectives (Eastman 2001, Munda 1995, Nijkamp et al. 1990). After having obtained the matrix effects/alternatives or the matrix factors/OGUs (**X**), that we can call criteria/alternatives matrix, the following are the steps in a decision making procedure of MCE.

2.6.1 Normalisation / standardisation

Both in DSS and in SDSS, the criteria are variables measured in different scales: rationa/interval (temperature, inhabitants of a city, concentration of pollutants in air, water, etc.); ordinal (relative impact of an intervention, classes of risk for a certain factor, etc.); binary (violation or non-violation of a standard, occurring or non-occurring of an event, etc.); nominal (negative, positive, no effect, etc.). To make alternatives comparable, standardisation or normalisation of rational/interval and ordinal variables is required, in order to set the range of the variables representing the criteria between 0 and 1 (dimensionless). Among the most common, there are:

Standardisation to maximum: x'(i,j) = x(i,j) / max (i)

Standardisation within interval: x'(i,j) = (x(i,j) - min (i)) (max(i)-min(i))

Standardisation to goal (i.e. in case of having to respect some standards or to achieve a goal, above or below which the score is considered 0 or 1): x'(i,j) = [x(i,j)- min (i)) / (goal(i)-min(i)]

Standardisation by non-linear functions: x(i,j) = x(i,j) / f(x) [where f(x) could be exp, log, S-shape, etc.]

2.6.2 Giving weight to each criterion

To each criterion (or groups of similar criteria) a weight should be given for defining its relative importance in composing the multicriteria index. Weights of criteria depend on different priorities of objectives or strategic scenarios, finally based on decision-makers choices. Different techniques are used to assign weights. **Direct assessment**: to each criterion a weight is attributed giving an arbitrary score based on judgement (e.g. from 0 to 10). Weights are then normalised to the sum of scores; **Pairwise comparison**: weights are given based on the relative importance of the criteria in pairwise comparisons (e.g. A is 2 times important than B, B is 1/2 times important than C, etc). The scores are reported in a symmetric matrix and the weight of each criterion computed according to the sum of the factor importance scores with regard to all the others; **Expected value** (direct ranking criterion): criteria are directly ranked based on an assigned scale of importance given by the experts. Weights are assigned proportionally to the rank fitting a chosen function (e.g. negative exp).

2.6.3 Ranking alternatives based on the multicriteria index

A multicriteria index (ranging 0-1) is calculated for each alternative summing the weighted contribution of each criterion:

$$I_j = \sum x(i,j) * W_i \tag{1}$$

where x(i,j) is the score of the variable corresponding to the i-th criterion and w_i is the weight given to it. The alternatives are ranked from the highest (best solution) to the lowest (worst) index.

2.6.4 Uncertainty estimation and sensitivity analysis

Uncertainty and *Sensitivity* analysis are useful techniques to estimate the stability of the ranks of the alternatives in function of magnitude of changes in weights and scores of the criteria. They can be performed by several techniques, the most suitable looks to be the one based on randomization.

2.7 Suitability maps and sustainable land use

The above described procedure for MCE has the same steps in DSS and SDSS. The rank in SDSS expresses a measure of suitability of OGUs to host a given land use. The mapping of the suitability indices produces the suitability maps for each land use. The selection of the criteria on which the suitability maps are based is depending on many factors related to the knowledge and perception of the stakeholders about their landscape and to their economic interests. The criteria may be considered arbitrary: different sets of criteria may be used and their weights can be changed. Of course changing criteria and weights may have effects on suitability maps (sensitivity analysis). It follows that suitability maps are more than a methodological issues since they drive the planners to shift from multidisciplinary model to interdisciplinary and transdisciplinary models requiring the participation of all the interested stakeholders in defining the criteria and their weights. The suitability is a not absolute concept, but relative to the context. For example, a piece of land may be suitable for rice cultivation even if it is in a dry area provided there is a system for irrigation that can supply the necessary water quantity for a ”convenient” cost.

If the criteria are chosen in function of a sustainable development, the suitability maps will show the optimal allocation of the selected land use according to sustainability criteria. It follows that suitability maps can be very useful for a cartographic presentation of different scenarios of sustainable development (e.g. Altobelli et al. 2001). It may happen that decisions are needed that satisfy several objectives at the same time. These objectives may be complementary or conflicting in nature (Rosenthal 1985, Carver 1991). In case of complementary objectives, multi-objective decisions can be solved through a hierarchical extension of the multicriteria evaluation process (Saaty 1999). More difficult is to afford conflicting objectives. These must be ranked and discussed in a participatory approach where different perspectives and scenarios should be discussed in a transdisciplinary approach (Rotmans and van Asselt 2001, Asselt and Rijkens-Klomp 2002). In SDSS the problem is more complicated than in DSS because the same OGU can be suitable for all the conflicting objectives and it has to be chosen only for one of them. In this case the ranks of the objectives obtained in DSS should be followed and used to combine the suitability maps into a single suitability map indicating the degree to which the OGUs meet all of the objectives considered (see Voogd 1983). With conflicting objectives, it is often not possible to find a unique solution in SDSS and the most common one is the development of a compromise solution according to the percentage of land allotment given to the different alternatives (Eastman 2001).

3 CONCLUSIONS

Both DSS and SDSS are semi-structured procedures because they have two main phases: the unstructured one dedicated to obtain the matrix effects/alternatives or factors/OGUs (matrices **X**) and the structured one that can follows well established mathematical procedures once the matrix **X** is given. The most important phase is to obtain the matrix **X**. It requires a participatory approach of stakeholders of the ”decision making domain” in a democratic and equity environment. The method of giving weight to the criteria by pairwise comparison is one that it is easily applicable in the participatory approach and that stimulate the active participation of the involved stakeholders. According to the experiences made, stakeholders are very curious to know how much consistent is the matrix they have generated, and they are always keen to work and try to give different weights till the consistency is reached. The method based on the similarity with the ”ideal point” (e.g. Burba et al. 2008) is also well understood by stakeholders since, the ordination graphs can show the mutual position of the alternatives (and /or OGUs) with respect what the stakeholders define as the best and the worst alternative (situation) in terms of the effects on the chosen criteria (see Burba et al., Belchior et al., this volume). Both the approaches allow the application of sensitivity and uncertainty analysis and would be able to implement at any stage expert knowledge. The use of public domain software and simple procedures such as the approximate method of weighting is strongly suggested as best practice since the stakeholders may have an easy control on what is going on without getting lost in the black boxes of commercial software.

TABLE 3: *The matrix factors/OGUs as n alternatives.*

	OGU 1	OGU 2	OGU 3	OGU j	OGU n
Factor a	X(a,1)	X(a,2)	X(a,3)	…………	X(a,n)
Factor b	X(b,1)	X(b,2)	X(b,3)	…………	X(b,n)
Factor c	X(c,1)	X(c,2)	X(c,3)	…………	X(c,n)
etc…	…	…	…	…………..	

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PART C

FROM SHALLOW WATER TO THE DEEP FJORD: THE STUDY SITES

OCCUPATION HISTORY OF THE SANTOS ESTUARY

M.R. GASPARRO, E.C.P.M. SOUSA, F. GIORDANO AND R.C. ARGENTINO-SANTOS

1 SETTLEMENT HISTORY

Nowadays, the Santos Estuarine System is considered one of the most critical areas in São Paulo State, concerning the degradation level of the different compartments which integrate the aquatic system and the seven Conservation Units due to the presence of the petrochemical industry pole in Cubatão city, the Santos Harbor, and also due to the irregular use and occupation of the lands favoring the dissemination of diffuse sources of contamination, complicating the monitoring and the integrated management of the environment. To evaluate such a variety of aspects of the "Baixada Santista", the study focus of ECOMANAGE Project, the detailed characterization of the area in a variety of aspects of knowledge is important. A succinct review of the history, focusing the industrialization process and its benefits and costs should help to learn about the importance of a development planning to minimize economic, social and environmental conflicts.

2 EARLY OCCUPATION

In 1508, a Port of slaves ("Temiuru") was settled in the region controlled by the Portuguese João Ramalho. Few years later, in 1532, Martim Afonso de Souza founded the first village of Brazil, in the Southern part of São Vicente Island. Some time later, in 1536, the Village of Santos started to grow, in a place called "Iguaguassupe" (big "lagamar"). In a natural sedimentation area bordered by the São Vicente and Santo Amaro Islands, a harbor with a little movement of ships was settled. It was the beginning of the history of Santos Harbor (Figure 1). In 1830 Santos Harbor started to export coffee (Figure 2). In 1867 the arrival of the trains of the Railway Company linking the Santos Port to the coffee farms in the Central part of the State of São Paulo brought a great increase in the activities in the region and, consequently, richness. But, on the other hand, the environmental problems in the estuary also increased. In 1895, the first industry ("Companhia Curtidora Fomex") was established in Cubatão City, located at the estuarine head. This industry produced tannin extracted from the black mangrove trees in order to tan animal skins.

During the decade of 1910 the first chemistry industries were settled down in the region and in the decade of 1920 the São Paulo road linking Santos to São Paulo cities was inaugurated. Also hydroelectric power plant Henry Borden started to capture the water from the area of São Paulo City and deposited it in the Estuary of Santos. The Baixada Santista is located on the mid-southern coast of São Paulo State enclosing the municipalities of Bertioga, Cubatão Guarujá, Itanhaém, Mongaguá, Peruíbe, Praia Grande, Santos and São Vicente. The highest population density is in São Vicente Island in the municipalities of Santos and São Vicente, occupying a territorial area of 271 km^2, of which 39.4 km^2 are in the insular domain and the greater portion (231.6 km^2) is located on the mainland.

The Santos - São Vicente estuarine System receives the discharge of important rivers which integrate the "Unidade de Gerenciamento de Recursos Hídricos (UGRHI)" (the Hydrological Resources Management Unit) of the Baixada Santista. These rivers are Cubatão, Perequê, Moji, Quilombo and Itapanhaú, besides other rivers, which come from other seven different Conservation Units, e.g. the State Parks of Serra do Mar and Xixová. The Baixada Santista region possesses a well established and developed social economic complex in which the Santos Harbor, the siderurgic-petrochemical pole in Cubatão, the commercial and service centre in Santos and the tourism exploitation are the more important factors.

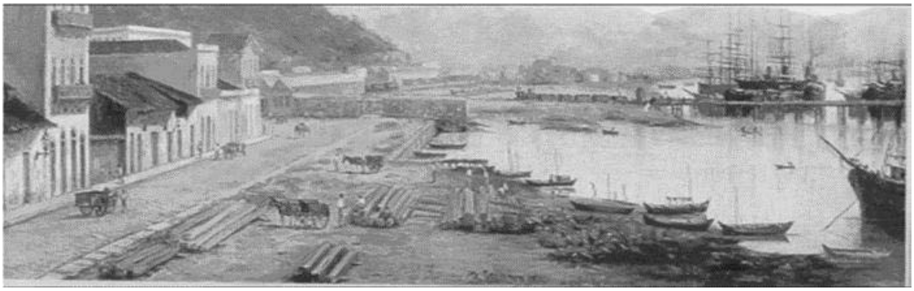

FIGURE 1: *Picture of the Santos Harbor in the 18th Century from a famous artist from Benedito Calixto (Porto do Bispo, Santos Colonial - Óleo sobre tela Benedito Calixto, c. 1765-1775. Source: Biblioteca Nacional).*

FIGURE 2: *Picture from the Coffee Loading Zone at Port of Santos during the XIX Century (Source: Fundação Arquivo e Memória de Santos).*

The geomorphological characteristics of Santos Bay, widely open to the sea and the depth of the Santos channel were decisive factors to the harbor settlement and increase. With the increase of the coffee trading, Santos had an intensive development, mainly in the second part of the XIX century. This process continued until the decade of 1940, when the housing boom started due to tourist demand (Goldenstein 1992). Until the 1940s, the Santos industrial park was not important. It consisted of industries linked to harbor activities and some others of consumption of goods.

Santos Harbor is the most important Brazilian harbor responsible for the traffic of the products from the biggest industrial park in Brazil comprising the States of São Paulo, Minas Gerais, Mato Grosso do Sul e Goiás (Silva Filho 1992). Santos Harbor is responsible for 30% of the total national export and for 60% of the total container shipments in the country. Nowadays the annual transport volume is approximately 60 million tons of a variety of products, contrasting with the 125 thousand tons transported in 1892. Santos Harbor possesses 12 km of piers on both sides of the Santos channel. However, it is still growing by the acquisition of new lands in the neighboring areas (`www.portodesantos.com.br/historia`).

During the Second World War, Brazil went through many difficulties related to petroleum derivates, which could not be imported. At that time the country had a minimal capacity of petroleum refining, so the industrial and transport sectors were deeply affected. Such situation favored the decision to create a pole composed by petrochemical and siderurgical industries near the Serra do Mar slopes, in Baixada Santista. It was a political and economic decision based on strategic reasons, which met the demand of economic groups from São Paulo. This decision did not take into account spatial planning criteria neither the theories of best locations for industry (Lineu 1992).

The chosen area was in the Cubatão municipality, located 14 km from Santos, covering an area of 160 km^2 in Serra do Mar slopes and in estuarine lands; the former utilisation was based on modest agricultural exploitation. Until the creation of the Anchieta road, in 1948, this place did not have any importance to the region; it was only a space between São Paulo and Baixada Santista beaches. The area became more and more attractive to the industries as the conditions for their settlement were improved. As a result, in the late 1970´s and early 1980´s, Cubatão became worldwide recognized as the ”most polluted city in the world” or as the ”death valley” due to problems related to pollution and environmental degradation caused by the disorganized and heavy occupation of the industries and population linked to them. Alonso (1992) states that the Cubatão development, before the environmental limit capacity was surpassed, was the clearest example of the unique prevalence of the economic criteria leading the decisions in the organization of spaces and in the management of the productive activities. As a consequence of the great loads of pollutants discharged into the system, in 1983 the São Paulo State government and the local industries elaborated a program aiming at gradually reducing the atmospheric pollutants emissions (Ab’ Saber 1982 in Milaré and Magri 1992). This program was executed by the government environmental protection agency, the CETESB (Lamparelli et al. 2001).

In a macro-regional analysis the proximity to the São Paulo metropolis, the good circulation axis previously built, e.g. Santos-Jundiaí Railway, Anchieta road and Santos Harbor constituted important factors in the potential of industrialization in Cubatão (Afonso 2006). Several other factors were not considered, such as:

- Limited space with strait stripes of soft soil, sandy and swampy lands with low support capacity, which are receivers of enormous freshwater sources mainly from the Serra do Mar slopes forming a complex net of channels with the hydraulic system which is very difficult to understand and control;
- The mangrove area rich in meandering rivers next to the Serra do Mar highlands interferes in the moist and warm climate, subject to atmospheric currents favoring the maintenance of the atmospheric thermal inversions which have a drastic effect against the dispersion of pollutants;
- The proximity of the Serra do Mar with its steep slopes in association with the level of air pollution increases the risk of soil slides, as the pollutants destroy the natural vegetation destabilizing the soil (Goldenstein 1992, Lineu 1992).

Cubatão city is different from other suburban centers once its organization was based on the necessity of a link between Santos Harbor and São Paulo city. In the absence of one of these elements: harbor and metropolis, Cubatão would not exist. The binomial São Paulo-Baixada Santista can be understood as a complex. Cubatão settlement occurred in a very dynamic phase of the Brazilian economy when the average growth rate of industrial production was 10.5% (between 1955 and 1960). In this period, the government assistance favored the development of industries with focus on the production of goods that used to be imported (Goldenstein 1992).

3 MAJOR INDUSTRIAL DRIVERS

Based on Alonso (1992) and Goldstein (1992), the unguided settlement process of the industries in Cubatão had as main participants the following industries.

3.1 Henry Borden Hydroeletric power plant

The Riacho Grande retention and the forced reversion of its waters towards the Serra do Mar, where the power plant was built, occurred independent of any geomorphologic evaluation. The enormous elevation gradient of the land in Santos side and the low gradient of the Rio Grande tributaries called the attention of the engineers of the Light and Power Company, which decided the water pumping downhill would occur in a single jet up to Cubatão (Mortonne 1935).

3.2 Refinery

The idea of building a refinery in the Southern part of the country appeared around 1948 and its inauguration was in 1955. The process of choosing the place for its settlement is not well known: Rio de Janeiro and São Paulo States were competitors. The decision was political, in the name of the national security, and was also based on the proximity of the Santos Harbor to be the petroleum receiver and on the energy availability from Henry Borden. Pipelines to transport the oil from Santos Harbor to the refinery were built. Another set of pipelines would be built to transport the oil products uphill to São Paulo, in the plateau. If the refinery had been built in the plateau only a set of ducts to transport the crude oil would have been enough. The waste of money did not stop: in a short time Santos Harbor became an oil terminal and the neighboring area turned into an industrial center, changing also the course of the urban development. A refinery represents one of the most powerful manifestations of modern capitalism. The great petroleum companies in the world- and among them is PETROBRÁS - constitute an enormous capital concentration, mostly based on the commercial and industrial integration, including research, exploitation, transport, refinement, and the distribution of the final products.

3.3 Petrochemistry

An example of the choices made about the location of the industries in Cubatão is the Ultrafértil Company, which was constituted by mixed funds of which the majority belonged to Philips Petroleum Company, the greatest ammonium distributor in the world. The Ultrafértil project was the production of nitrogen nutrients from naphtha and phosphoric nutrients. Such products were imported and the government gave guarantees to keep their supply. PETROBRÁS was created in 1953, as a mixed economy society, with the function of ruling the national petroleum politics and defining the attributions of the National Petroleum. The Presidente Bernardes Refinery in Cubatão was a new component of its patrimony. This refinery had a multiplier effect: several petrochemical industries became interested in using the advantages of such location. Generally these industries were subsidiaries of big North-American companies as Copebrás, Union Carbide and Estireno, among others. These companies remained relatively small for ten years, waiting for solutions of the political problems related to petroleum derivates. In 1965, when the access of the private companies to this sector was opened, the petrochemical industry and the PETROBRÁS (through its subsidiary, the PETROQUISA) had a great increase and new petrochemical industries were settled in Baixada Santista and in the plateau as well.

The ”Companhia Brasileira de Estireno” began to operate in 1957, using gaseous ethylene from PETROBRÁ?S. The Union Carbide of Brasil S/A (polyethylene producer) started its production in 1958, also using the ethylene produced in the Arthur Bernardes refinery. The Brazilian Petrochemical Company (Copebrás) started its operations in Baixada Santista in 1958. Another company, the Ultrafértil, was created in 1967 by the merger of the Brazilian group

Ultra Gás with the great American petrochemical Philips Petroleum Company. The Ultrafértil was settled in this region because 58% of the fertilizers consumption in the country was from the Central and Southern regions.

3.4 Siderurgic

The COSIPA is the great private siderurgical complex created in Baixada Santista in 1951. This company was settled in a creek linked to Santos estuary, in a swampy ground. The preparation of the ground for its purposes was expensive and time-consuming, requiring many landfills. The area occupied by COSIPA is $\approx 7.9\,km^2$ (Gutberlet 1996). The state government entered as a minority partner in 1956 to guarantee the continuation of COSIPA activities. This industry had an important role in the regional development, and also for the country development. Nevertheless, it generated many environmental problems. The cement company CIMERITA (Cimento Santa Maria) was created to use the coal furnace slag generated by COSIPA, since this material represents 60% of the bulk matter to produce this kind of cement. Nowadays, Cubatão (Figure 3) is among the first state municipalities in production values and occupies the 3rd position in the contribution to state per number of inhabitants. Its industries, with few exceptions, are big and modern enterprises, with the participation of foreign wealth attracted by the opportunity of participation in this pioneer activity in the country.

FIGURE 3: *Cubatão siderurgy park (source: ECOMANAGE photo by Renan B. Ribeiro).*

4 ECONOMIC AND ENVIRONMENTAL CONFLICTS

The growth of the global economy during the decade of 1990 led Cubatão to a big industrial production representing up to 54% of the petrochemical industry, 21% of fertilizers industry, 15% of the Metallurgy and 9% of the chemistry industry of Brazil. According to Ab Saber

(1982) the case of Cubatão can be taken as a desolate example of an imperfect organization of human occupation of the land in the third world. The excellent macro location (between Santos Harbor and São Paulo) was not able to surpass the negative aspects imposed by its micro location (in a swampy area). The city has available old and modern structures built along the time in the region, still represents the pathway linking the biggest and most important Brazilian harbor to the biggest and most important city in the country. The estuary is also used as private harbors to the local industries. However, there is little available space in the margins of the steep Serra do Mar slopes, covered by tropical rain forest. Thus, the industries, commercial and residential areas have to occupy the small parts of flat ground surrounded by the slopes and the labyrinthine tidal flat dominated by mangroves (Gutberlet 1996, Ab Saber 1982). It is important to note that since the beginning of the industrial pole settlement in Cubatão environmental legislation is not effective, until the present days (Afonso 2006).

> *"Fora de seus próprios limites, o município era notório apenas para os taumaturgos do "milagre brasileiro", cujo brilho efêmero então deslumbrava os espíritos acríticos. Ali se instalavam - diriam - um pólo industrial de base, que bem poderia simbolizar e resumir a arrancada brasileira para o desenvolvimento: o lugarejo estagnado de há pouco transmutava-se em pujante complexo sídero-químico; os manguezais bochornosos e improdutivos cediam lugar aos altos-fornos e às torres de craking, fornalhas de dólares que em breve metamorfozeariam o trópico indolente em potência do Primeiro Mundo...."* Melo Filho (1972)

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CLIMATOLOGY AND HYDROGRAPHY OF SANTOS ESTUARY

J. HARARI, C.A.S. FRANÇA AND R. CAMARGO

The atmospheric conditions at the surface have great influence on the dynamic and biological characteristics of the ocean. In this chapter, initially meteorological data produced by the global atmospheric model of the National Center for Environmental Prediction / National Center for Atmospheric Research (NCEP/NCAR) (Kalnay et al. 1996) were used to analyze the climatology of the South Eastern Brazilian coastal area, in which the estuary of Santos is enclosed. Although some local time series are available in the coastal region of Sao Paulo State, NCEP / NCAR data are very suitable for climatological and meteorological studies, due to their homogeneity, quality and number of variables, besides the fact that the atmospheric model already incorporates local observations.

Data from the period 1980 to 2004 were considered in the climatology analysis of the oceanic region, where the mean surface winds are mainly from NE, weakening and turning from E at the platform and continental area. Normal monthly mean values of the most important meteorological variables at the position 22°30'S 45°W, the nearest grid point of NCEP / NCAR model to Santos Estuary, are presented in Figure 1, with their respective standard deviations (above and below the central lines), as computed for the 25 years period (normal monthly means are the mean values of all months of January, February, etc). Near to the estuary, the annual variations are significant, as for example for the air temperature, which ranges from 17.33 ± 3.89 °C in July to 21.77 ± 1.89 °C in February, having a mean value over the whole period 1980-2004 of 19.72 ± 3.39 °C; the rates of precipitation and evaporation also display large annual variations (Figure 1).

Precipitation computations agree fairly well with data from the rain gauge stations on the coast; for example, the annual mean given by NCEP / NCAR model is 0.55 ± 0.96 e-07 m s^{-1}, which corresponds to 1734.45 ± 3027.50 mm yr^{-1}, while the rain gauge located at Guaruja, 24°00' S 46°17' W, measured 2131.22 mm yr^{-1} (SIGRHI - Information System for Water Resources Management of Sao Paulo State - DAEE site, access in 2006 - `http://www.sigrh.sp.gov.br`). The atmospheric circulation at the surface, in Baixada Santista region, depends on the Subtropical High Pressure Center of the South Atlantic and its interactions with the Sub Polar Low Pressures (Moscati et al. 2000); considering typical conditions, winds from the East are dominant in all seasons, having a mean value of 1.49 ± 3.77 m s^{-1}, with mean sea level pressure of 1015.59 ± 4.33 hPa (Figure 1). However, in the limit of the sub tropical and sub polar regions, strong western winds occur, developing instabilities with scales of several days and thousands of kilometers; as a consequence, in the Brazilian southeastern area cold fronts modify the typical atmospheric conditions, leading to a rotating wind from East to North and West, with atmospheric pressure falling about 10 hPa, followed by winds from the South, air temperature decreasing about 5 °C and atmospheric pressure rising; in the sequence, the winds turn East again, and the temperature and pressure increase up to their typical values. The intensity and duration of the instabilities vary over the year, being more frequent and stronger in winter than in summer; in general, eastern winds are weak (less than 3 $m\,s^{-1}$) of long duration (5 to 10 days) while the wind gyre (from East to North and West) takes a few hours, producing strong wind intensities (up to 10 $m\,s^{-1}$), characterizing the cold front's arrival, which is followed by strong Southern winds (between 5 and 10 $m\,s^{-1}$) during 1 to 3 days.

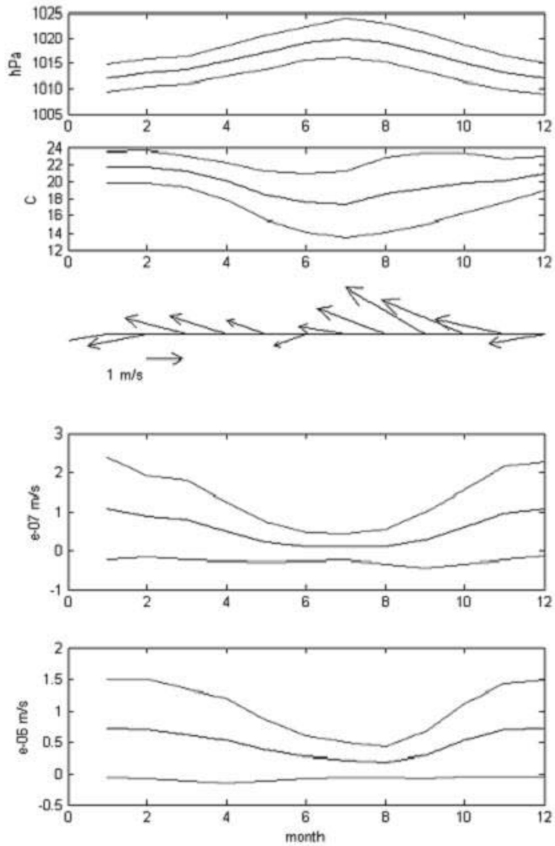

FIGURE 1: *Normal monthly mean sea level pressure, air temperature, surface winds, precipitation and evaporation (upper to lower), with standard deviations, as computed by NCEP / NCAR model for the period 1980 - 2004, at the position 22°30'S 45°W.*

The influence of cold fronts in generating Southern winds may be observed in the normal monthly means, which display a shift of the mean winds to NW, especially in the months of August to October (Figure 1). The meteorological conditions at the surface for 22°30' S 45°W, in the months of the physical measurements in Santos estuarine region, September 2005 and March 2006, are presented in Figures 2 and 3 (winter) and 4 and 5 (summer), as time series of sea level pressure, air temperature, winds, precipitation and evaporation rates, as extracted from the NCEP/NCAR global atmospheric model (Kalnay et al. 1996). Figures 2 and 3 show that during the month of September 2005 six cold fronts reached the Brazilian Southeastern region (Moscati et al. 2000), causing several periods of strong Southern winds, but only one significantly rainy period (days 26-28); however, the hydrographic and current measurements in Santos estuarine region were done under good meteorological conditions, between two successive cold fronts, in September 16th (winds from the East). On the other hand, in the month of March 2006, only three weak cold fronts were observed with several rainy days (Figures 4 and 5), and the first front occurred during the winter hydrographic and current meter survey, on days 13th and 14th of that month, causing significant winds from South.

Tides within Santos Estuary are regularly recorded in Torre Grande tidal station, 23°56.95' S 46°18.50' W, and analyzed through the harmonic method (Franco 1988, Franco and Harari 1987). The main tidal components amplitudes and phases are presented in Table 1 (Harari and Camargo 1995), showing a semi-diurnal tide with daily inequalities; the ratio of the sum of the amplitudes of the two main diurnal components to the sum of the amplitudes of the two main semi-diurnal components is F = 0.30. The tides in Santos have amplitudes of 0.60 m and 0.14 m, for spring and neap tides, respectively. Considering the tidal records of 2005-2006, daily, monthly, seasonal and annual mean sea levels were computed by applying a low-pass filter based on running means: denoting A24 as the mean value of 24 hourly observations, the sea level heights were submitted to the A24 A25 A25 filter, in order to remove tidal oscillations (Godin 1972). Values of the normal mean sea level for the analyzed period are presented in Figure 6. The normal monthly mean sea levels have a maximum in April - May and a secondary maximum in August - September, which are related to steric effects in summer (volume increase) and cold fronts in winter, respectively (Harari and Camargo 1995). Observed sea surface levels and computed mean sea levels in Torre Grande, in September 2005 and March 2006, are displayed in Figures 7 and 8.

TABLE 1: *Principal tidal components in Santos Estuary: components, angular velocity (degrees/h), amplitude and standard deviation (cm), Greenwich phase and standard deviation (degrees).*

Components	Angular Velocity (°/h)	H (cm)	± (cm)	Gw (°)	± (°)
Q1	13.3986609	3.19	0.31	101.94	6.55
O1	13.9430356	11.69	0.31	124.22	1.78
P1	14.9589314	2.03	0.31	182.34	8.68
K1	15.0410686	6.44	0.31	187.63	3.01
N2	28.4397295	5.23	0.53	231.09	5.69
M2	28.9841042	37.32	0.53	173.36	0.79
S2	30.0000000	23.91	0.53	179.27	1.27
K2	30.0821373	8.01	0.53	170.26	4.75
M3	43.4761560	6.11	0.33	349.96	2.99

The most important features of the sea level and mean sea level plots (Figures 7 and 8) are the tidal amplitude variations between spring and neap tides (0.60 and 0.14 m, respectively) and the mean sea level oscillations associated to cold front incursions (Harari and Camargo 1995); measurements on 16th September 2005 were performed after the passage of a strong meteorological system that raised the mean sea level from -0.4 to +0.5 m, while the 13-14th March 2006 observations were done under the influence of a weak cold front, which raised the mean sea level to about 0.3 m above the annual mean. Besides sea level, another important characterization of the estuary is related to currents and temperature - salinity fields. A representative sampling of the estuary was done by FUNDESPA (2002), in a section located at 23°53.99' S 46°22.64' W, near to the port of Companhia Siderúrgica Paulista (COSIPA), in front of Cubatão, sampled in the period from 21 June to 11 July 2001. In this period, a small temperature interval was observed, 20.0 to 22.9 °C, and a larger one for salinity, 23.7 to 30.8; as a consequence, density varied with salinity, presenting values between 1015.4 and 1021.4 kg m^{-3}. Smaller salinities were estimated at the surface, due to river discharges, and larger values near to the bottom, related to the intrusion of haline waters coming from the adjacent coast, transported by flooding tides and gravitational circulation. Maximum ebbing and flooding currents of the longitudinal (N-S) component in COSIPA section reached -0.63 and 0.50 m s^{-1}, respectively. In the complete analyzed tidal cycles, extreme ebbing currents were stronger than the correspondent flooding ones, due to river flows; the transversal (E-W) current component is much smaller in COSIPA area. Based on these data, time series of vertical profiles of currents, salinity and density are presented on Figure 9, representing four tidal cycles, starting on 05 July, 2001 - during spring tides. The configuration of isohalines and isopycnals indicate that, in the estuary, temperature and pressure have smaller effect in density while salinity is the dominant factor. According to the vertical stratification of salinity, the estuary, near to its head, is partially mixed and weakly stratified, indicating that part of the tidal energy is responsible for the vertical mixing in the water column.

Comparing neap and spring tidal periods, in the former surface currents reached extreme values of -0.27 and 0.32 m s^{-1} at the surface, -0.22 and 0.26 m s^{-1} at 5 m depth, finally -0.15 and 0.25 m s^{-1} at 10 m; the bigger asymmetry at the bottom (flooding currents towards the continent, >0) indicates the prevailing density currents (baroclinic effect) and the possibility of substances transport from the adjacent coast. On the other hand, during spring tides, stronger currents occur, reaching -0.45 m s^{-1} and 0.35 m s^{-1} at the surface, -0.35 m s^{-1} and 0.30 m s^{-1} at 5 m depth, -0.26 m s^{-1} and 0.25 m s^{-1} near to the bottom (10 m). The layered Richardson Number and the Stratification - Circulation Diagram (Miranda et al. 2002) show that 95% of the salt transport within the estuary is due to eddy diffusion and only 5% by advection; the weak vertical stability allows the transport of nutrients to the euphotic zone, provided their availability in deeper layers.

The measurements in ECOMANAGE Project of physical and hydrodynamic properties representing winter and summer conditions were taken across sections in the estuary channels axis with 2 or 3 stations each (Figure 10); on each station, 1 to 6 levels were sampled, depending on the maximum depth; the surveys comprised a total of 9 sections, with 25 stations in the winter campaign and 28 in the summer one.

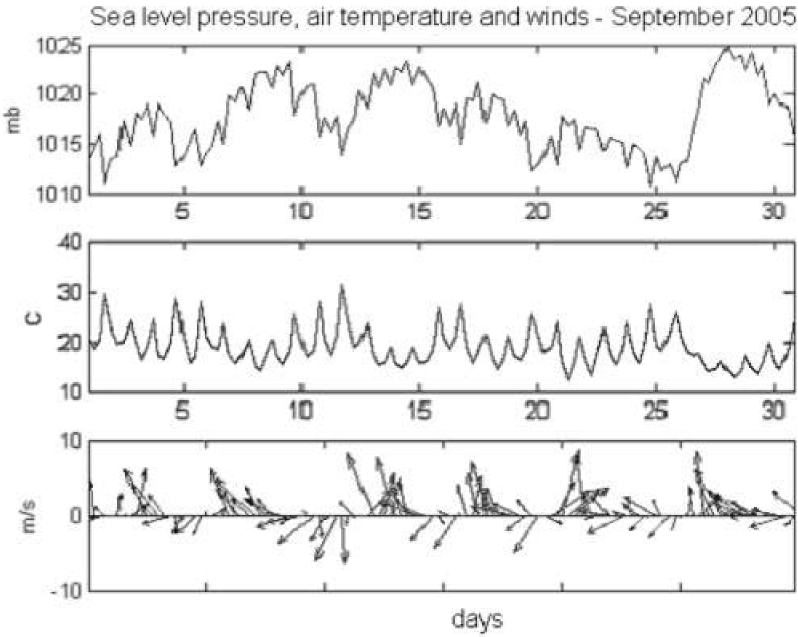

FIGURE 2: *Surface sea level pressure (upper), air temperature (middle) and winds (lower) computed by the NCEP / NCAR model in the position 22°30' S 45°W, for September 2005.*

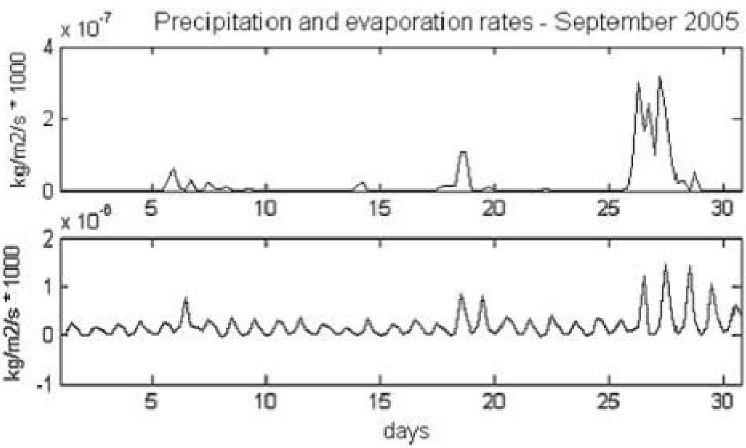

FIGURE 3: *Precipitation (upper) and evaporation (lower) computed by the NCEP / NCAR model in the position 22°30' S 45°W, for September 2005.*

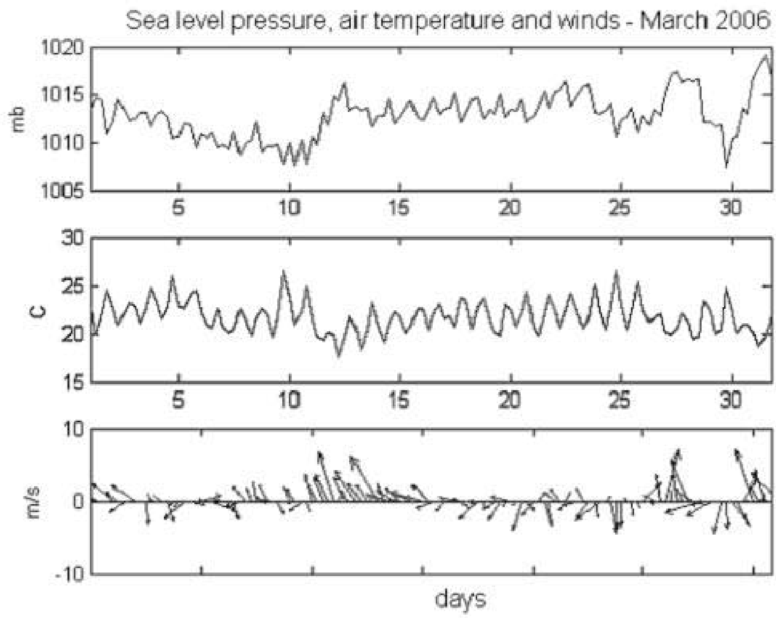

FIGURE 4: *Surface sea level pressure (upper), air temperature (middle) and winds (lower) computed by the NCEP / NCAR model in the position 22°30' S 45°W, for March 2006.*

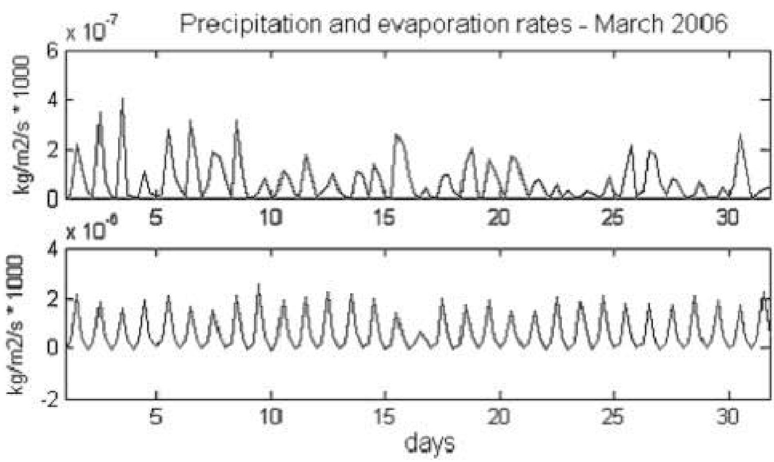

FIGURE 5: *Precipitation (upper) and evaporation (lower) computed by the NCEP/NCAR model in the position 22°30' S 45°W, for March 2006.*

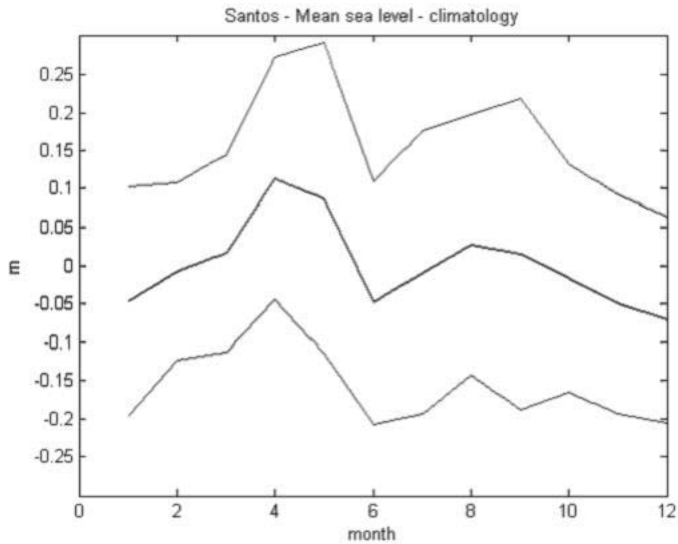

FIGURE 6: *Normal monthly mean sea levels in Santos (m), with standard deviations, considering the period 2005-2006.*

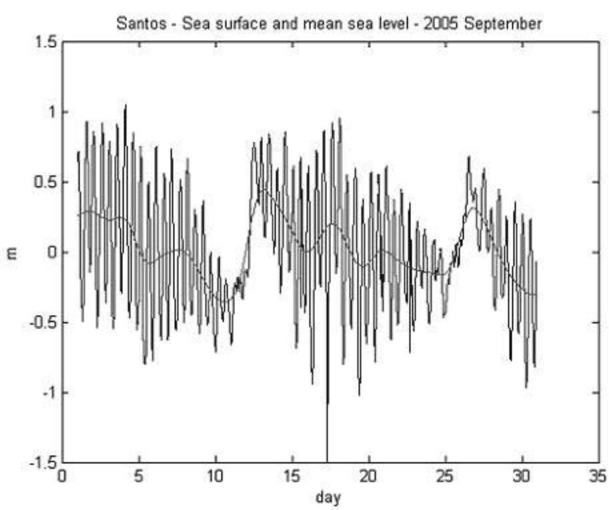

FIGURE 7: *Sea surface and mean sea level in Santos Estuary - Torre Grande tidal station, in September 2005.*

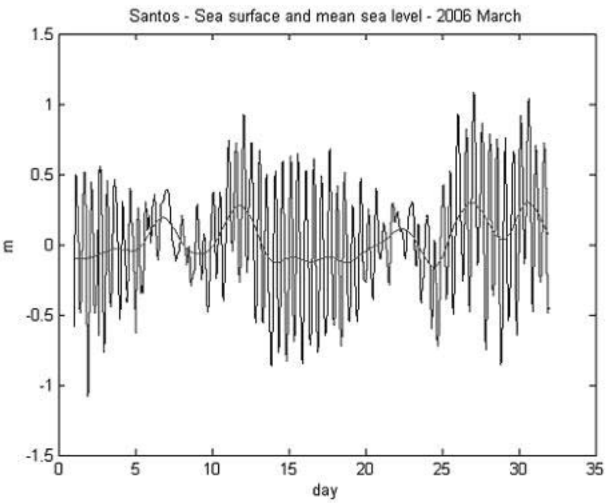

FIGURE 8: *Sea surface and mean sea level in Santos Estuary - Torre Grande tidal station, in March 2006.*

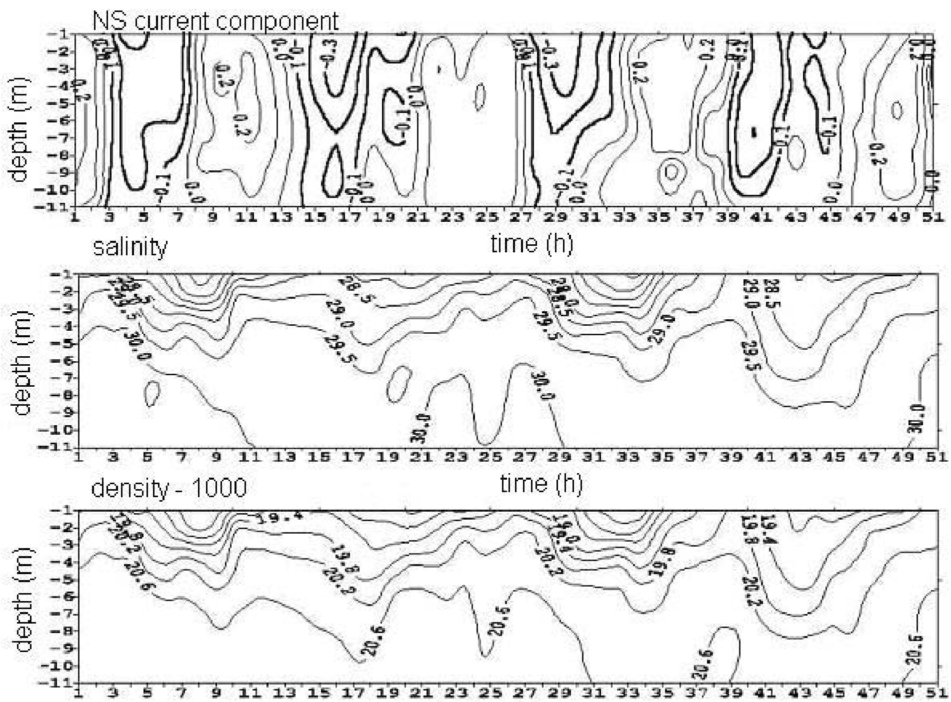

FIGURE 9: *Time series of vertical profiles of NS current component (upper), salinity (middle) and density (below, subtracting 1000 kg m^{-3}) during four complete tidal cycles, starting on 00:00 GMT 05 July 2001.*

On September 2005, stations were taken in a clockwise sense in the estuary, starting at section 8 in the western entrance, from 12:30 UTC to 17:40 UTC; this period corresponds, approximately, to flood-tide (from low-water at 10:30 UTC to high-water at 17:53 UTC), in spring tide conditions. Mean temperatures were, everywhere in the estuary, close to 21 ℃, with very small regional variability, within a range of 0.3 ℃, close to the measurement precision. Surface and bottom temperatures show the same behavior, having very small stratification (warm water overlying cold water), but again within the measurement errors.

Salinity varies from 15 (fresh) to 35 psu (haline ocean water) in the estuary. Lowest values are found in the area close to where tidal waves meet, in the inner section of the estuary, also due to the sampling schedule. Differences between west (fresh) and east (saltier) estuary entrances should be attributed to the tidal phase. Although smaller than the horizontal variability, there is a noticeable increase in salinity with the depth everywhere in the estuary, up to 4 psu.

Dissolved oxygen concentrations displayed larger values at the channels entrances (especially at the surface) and smaller ones inside the estuary (at the bottom), with maximum value at the surface of station 1A (7.68 $ml\ l^{-1}$) and minimum at the bottom of station 5A (4.58 $ml\ l^{-1}$). Figure 11 presents selected vertical profiles of temperature, salinity and dissolved oxygen, at the entrance of the Port Channel (section 1) in the innermost station (5) and at the entrance of Sao Vicente Channel (section 8). As stated before, the temperature variations are very small, while salinity and dissolved oxygen exhibit significant gradients along the Channels and strong vertical stratification in the inner region.

The observed current vectors for each station in the winter survey produced Figures 12 and 13, for surface and depth-mean currents. Mean currents reach up to 0.50 $m\ s^{-1}$ in the estuary; the tidal flooding during the observation period implied sea-to-estuary currents everywhere; these measurements show that the tidal waves, coming from both estuarine entrances, meet somewhere between sections 6 and 4 (west of Casqueiro Island). Surface currents follow quite close to the mean currents, eventually reaching more than 0.50 $m\ s^{-1}$.

On 2006, the summer samplings were taken in two consecutive days, 13th and 14th March, for the Port of Santos and Sao Vicente Channel, again with flooding currents in spring tide conditions; on 13th March low water occurred at 11:08 UTC and high water at 17:08 UTC, with measurements between 12:24 UTC and 18:13 UTC; and on 14th March, the times of low and high waters were 11:39 UTC and 17:38 UTC, with samplings between 14:50 UTC and 17:29 UTC.

In March the temperatures were significantly higher than in September, ranging between 26.92 ℃ and 29.12 ℃, but again displaying small vertical variations; on the other hand, salinity values range over a larger interval, from 16.7 to 28.8 psu, with significant stratification in some stations, of about 4 psu; dissolved oxygen also has significant gradients along the channels, with a maximum at the surface of station 1 (8.81 $ml\ l^{-1}$) and a minimum at the bottom of station 5 (3.56 $ml\ l^{-1}$). These characteristics may be observed in the selected profiles shown on Figure 14, in the same sections of Figure 11.

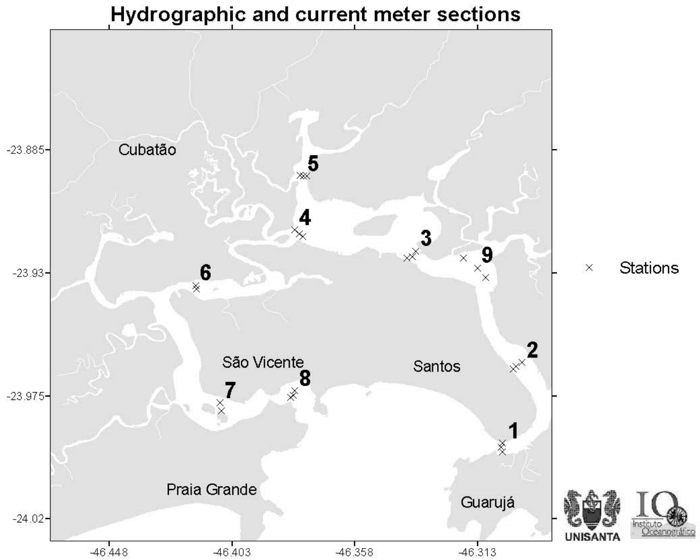

FIGURE 10: *Positions of the hydrographic and current meter sections (1 - 9).*

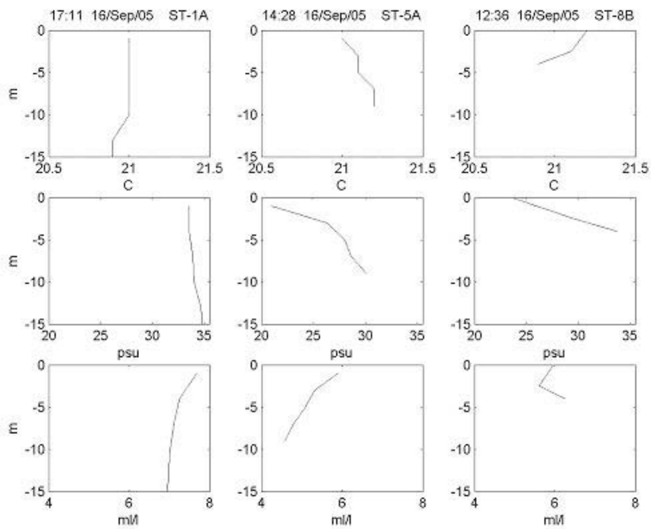

FIGURE 11: *Vertical profiles of physical properties (temperature above, salinity middle and dissolved oxygen below) in selected stations of September 2005 sampling - stations on sections 1 (left), 5 (middle) and 8 (right).*

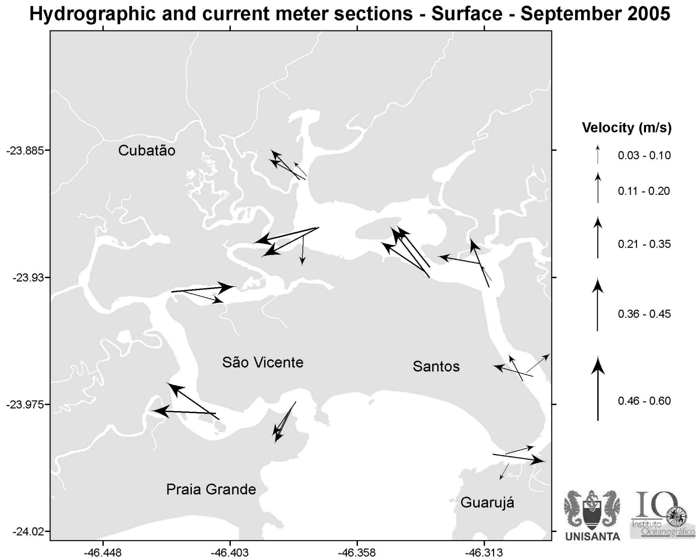

FIGURE 12: *Surface currents measured on September 2005.*

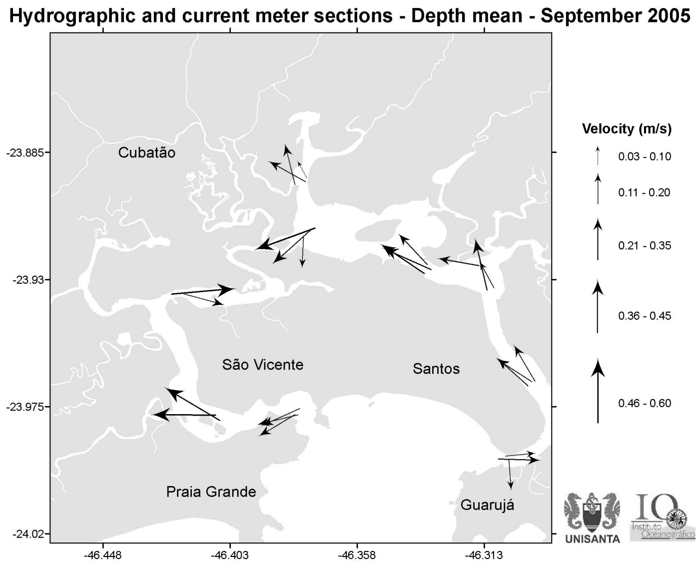

FIGURE 13: *Depth mean currents based on measurements at several depths on September 2005.*

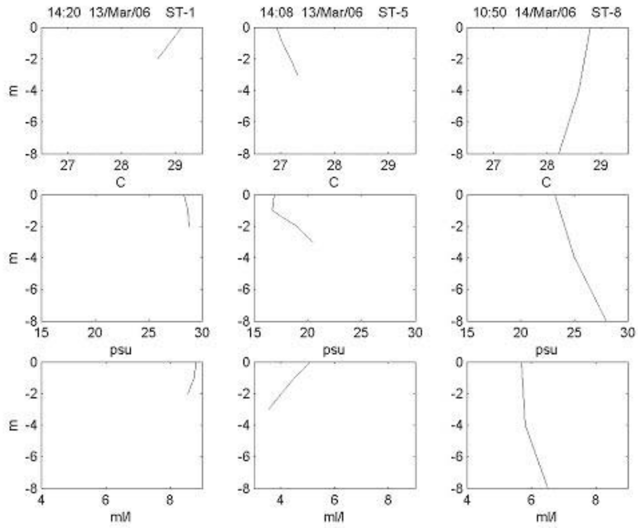

FIGURE 14: *Vertical profiles of physical properties (temperature above, salinity middle and dissolved oxygen below) in selected stations of March 2006 sampling - stations on sections 1 (left), 5 (middle) and 8 (right).*

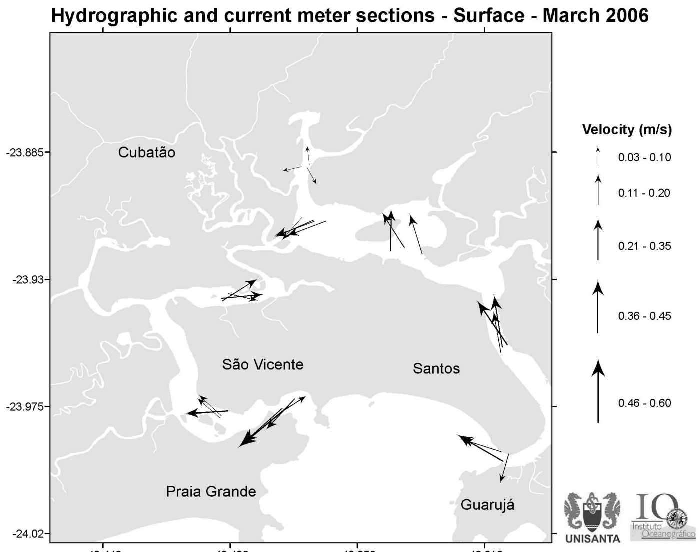

FIGURE 15: *Surface currents measured on March 2006.*

The current patterns in summer were not much different from winter (Figures 15 and 16), due to the large influence of the tidal currents, considering that both campaigns were done during spring tides with flooding currents (Harari et al. 1999, Harari and Gordon 2001). Again maximum currents are about 0.50 m s^{-1}, with small variations between the surface and deeper values.

Temperature and salinity fields measured in the ECOMANAGE Project have the same features of observations of previous campaigns; the current measurements also have the same standards, although the characteristics of residual surface currents towards the sea and bottom currents towards the continent, in the inner part of the estuary, may only be detected in longer time series.

Physical data presented here, a systematic observation of Santos Estuary waters, appear to be reliable and representative of the winter and summer conditions. These data, together with other previous observations, are used to validate the hydrodynamic numerical models.

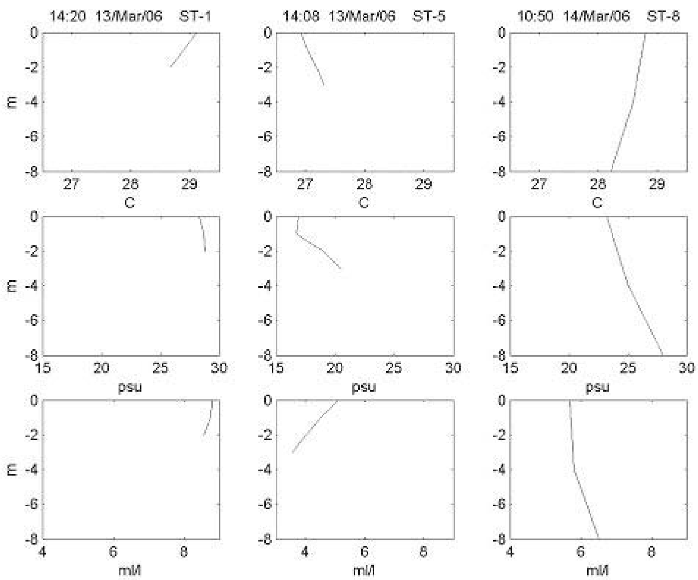

FIGURE 16: *Depth mean currents based on measurements at several depths on March 2006.*

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PRIMARY PRODUCERS IN SANTOS ESTUARINE SYSTEM

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1 ATLANTIC RAIN FOREST

Baixada Santista lies in the Atlantic Rain Forest area, which originally comprised a complex of broad-leaved evergreen forests that once dominated the Brazilian Eastern border, along an almost continuous belt from the state of Rio Grande do Norte (6°S) to Rio Grande do Sul (30°S). In the South-eastern and Southern parts, the Atlantic forest is characterized by a remarkable altitudinal gradient and its most typical physiognomy is associated with the Serra do Mar. The distinct physical conditions combined with the past influence of different floras resulted in an exceptionally high floristic and physiognomic diversity, in all the three levels: within-habitats (alfa), along environmental gradients or among different communities (beta), and diversity of the landscape (gamma). Nowadays, however, only little remains of the forests' original complexity. As the main setting of Brazilian history, the Atlantic rain forest has been drastically reduced since the discovery. The remnants of the forest currently cover no more than 7% of its original extent. Continuous remnants are presently restricted to two separate regions: Southern Bahia state, up in the Northeast, and Southeastern Brazil.

In the study area, as in other parts of the Southeast, the dense evergreen forest still covers most of the scarps of the Serra do Mar, where it was preserved to a large extent due to the natural limits of the environment - the steeply dissected relief, the dense drainage, the high atmospheric humidity and rainfall rates, and the pronounced surface processes. Figure 1 presents the distribution of the Atlantic forests' remnants. Most of the remaining natural vegetation is or has been somehow disturbed by the anthropogenic occupation, intense use of natural resources and, at least during the last half-century, by high levels of air, water and soil pollution, which was not controlled until 1983. In 1962 the forests of the Serra do Mar were not severely degraded by air pollution. Primary and secondary forests covered 53% and 30% of the Cubatão area, respectively. In the following years, the unfavorable conditions for the atmospheric dispersion of industrial emissions and the increasing levels of atmospheric pollutants promoted drastic changes in the composition and structure of the natural forests, which soon lost most of their emergent and canopy trees, for they were not able to survive in such a highly polluted environment. By the end of the seventies, virtually all the natural forests of the Serra do Mar surrounding the Baixada Santista were strongly affected by air pollution (Pompéia 1997).

The conditions have been slowly changing since the mid eighties, when the State Environment department started to control the industrial emissions. In spite of some recovery of the canopy, the forest diversity, however, is not being restored, since even the lower levels of atmospheric pollution still restrict the establishment or hamper the development of most native species,

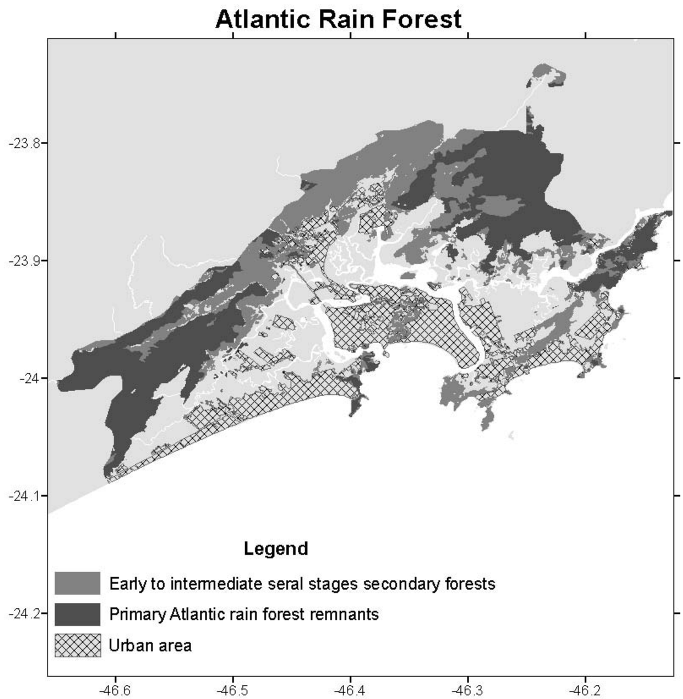

FIGURE 1: *Distribution of the forests' remnants in Baixada Santista.*

favoring the proliferation of a few tolerant pioneer species (Pompéia 1997). Intermediate seral stage secondary forests along slightly polluted areas have closed canopies, but lack most hardwood species and adult forms of *Euterpe edulis*, a palm tree intensively exploited due to its commercial value, a clear evidence of disturbance. Myrtaceae, Rubiaceae, Euphorbiaceae, Sapotaceae, and Melastomataceae are usually the richest and most abundant families. Values of alpha diversity vary from 3.4 to 3.9, similar to those of other developed secondary forests (e.g. Silva and Leitão Filho 1982, Mantovani et al. 1990, Mantovani 1993, Melo and Mantovani 1994), and densities vary from 2300 to 2500 individuals per hectare (Leitão Filho et al. 1989, Leitão Filho et al. 1993, Pompéia 1997).

In the heavily polluted areas, forests have open canopies and lack bromeliads, orchids, gesneriaceans and other epiphytes, which are remarkably diverse and abundant in undisturbed forests. Under storey tree ferns, caespitose palms and several Piperaceae are fairly common, especially in forest gaps. Myrtaceae, Melastomataceae, Palmae, Rubiaceae, and Fabaceae are the richest and most abundant families. Pioneer species, such as *Tibouchina pulchra*, *T. mutabilis* and *Miconia cinnamomifolia* (Melastomataceae), *Bactris setosa* and *Syagrus romanzoffiana* (Arecaceae), and *Cecropia glaziouii* (Cecropiaceae), are abundant. Alpha diversity, otherwise, is low, varying from 2.14 to 3.09 (Leitão Filho et al. 1993, Pompéia 1997). A considerable number of trees belonging to the most common species develop an abnormal number of additional trunks, suggesting the premature death of the main trunk and basal branching. These species are able to grow relatively well until they reach the canopy, when they become more vulnerable to the phytotoxic effects of the atmospheric pollutants. At this stage, most of the typical canopy and emergent trees die and are replaced by tolerant and fast-growing pioneer species (Pompéia 1997).

2 MANGROVE FOREST

Besides the rain forest remnants, the area still preserves quite extensive mangrove wetlands (Figure 2). Mangrove forests are the usual vegetation in large parts of tropical and equatorial sheltered areas. It is well known that mangrove forests represent an important detritus source to adjacent estuarine and coastal systems (Odum and Heald 1975). They hold great intrinsic ecological value providing various kinds of goods and services.

In Brazil, mangroves are usual structures along its coastal areas, from the Northern frontier (4°30' N) to Laguna, in Santa Catarina state (28°30' S). The total area of Brazilian mangroves is estimated from 10,000 km^2 (Herz 1987) to 25,000 km^2 (Saenger et al. 1983), depending on the methodology employed for the quantification. The Brazilian mangroves present a typical vegetal formation constituted by six species [*Rhizophora mangle* L., *R. racemosa* G. F. Mayer, *R. harrisonii* Leechman, *Avicennia schaueriana* Stapf. and Leech, *A. germinans* (L.) and *Laguncularia racemosa* (L.)]. Only three of these species occur in São Paulo state: *R. mangle*, *A. schaueriana* and L. racemosa (Lamparelli and Moura 1998). According to Herz (1991), the mangrove area in the coast of São Paulo State encompasses 231 km^2 and 42.5% is located in the Baixada Santista. However, according to Lamparelli and Moura (1998), this area is slightly larger, with 120.2 km^2 or 52% of São Paulo state mangroves forests. Santos, São Vicente and Cubatão are the main municipalities that shelter the Santos estuarine system and are covered by 69.7 km^2 of mangrove forests (Lamparelli and Moura 1998). According to CETESB (1991), 16% of the original mangrove areas in the Baixada Santista have been converted to urban and industrial uses, 40% of the forests are well preserved, mainly those located in the Bertioga region and some in the São Vicente region; the degraded mangroves total 44%, mainly in the Santos/Cubatão region, which is clearly related to industrial pollution.

The main sources of degradation in this region have been oil spills and the industrial and urban effluents that enter the system through the Cubatão and Mogi Rivers, including the waters of the Billings Reservoir (São Paulo municipality), discharged through the Henry Borden power plant (CETESB 1990) into the Cubatão river. In spite of its ecological importance, there is little information about the mangroves structure and production in the Santos estuarine system, and most of it was obtained in studies developed in small distinct areas like those by Comelli et al. (1994), Menezes (2000), Pozzi-Neto (2000), Menguini (2004) and Fruehauf (2005).

Comelli et al. (1994) studied the phytosociological characteristics of a small mangrove area in Cubatão. An attempt to restore degraded mangrove areas was performed by Menezes (2000) through the planting of mangrove species in a bare area previously occupied by mangrove in

FIGURE 2: *A typical mangrove forest in the Santos estuarine system (photo ECOMANAGE by João A. P. dos Santos).*

Cubatão. The author also made an attempt to gain the commitment of the local fishing community in projects of environmental restoration/conservation and concluded that the restoration of degraded mangroves is feasible in the area, provided some care is exercised and the local communities are involved in the establishment of conservation policies. Pozzi-Neto (2000) studied some mangrove sites on the Santos estuarine system aiming at applying economic assessment techniques to obtain the partial economic value of those mangroves forests. Menguini (2004) studied the mangrove area at Barnabé Island (Santos Estuarine System), a site impacted in 1998 by a DCPD (dicyclopentadiene) spill that was followed by a fire, in order to assess residual impacts and to follow the natural restoration processes. Fruehauf (2005) assessed the contamination by heavy metals in the soil, water, vegetation and organisms in three areas in the Santos estuarine system. The worst environmental conditions were found at the Cosipa channel (= Piaçaguera channel), followed by the Cascalho River mangrove and Mariana River. It was verified that metal contamination was the cause of the decreasing establishment of *R. mangle*.

There was one large study in the region done by CETESB (1988), that analyzed 33 sites in the mangrove system, including Bertioga channel which used to receive less contaminants. This study described the forest height ranging from 4.5 m to 13.2 m (average of about 8 m), density ranging from 600 to 3800 trees ha^{-1} and with 900 to 5800 stems ha^{-1}. The diameter at breast-height (DBH) ranged from 3.6 to 12.8 cm with the basal area between 3.59 to 31.13 m^2 ha^{-1}. The seedling and sapling density ranged from 0 to 23200 ha^{-1} and from 0 to 5200 ha^{-1}, respectively. With regard to the leaf size, the average in the different sites ranged from 20.7 to 36.5 cm^2 for *R. mangle*, 12.5 to 27.9 cm^2 for *L. racemosa* and 10.5 to 18.8 cm^2 for *A. schaueriana*.

In order to obtain recent information about the mangrove system, a new estimate of the mangrove area was performed using the satellite image mosaic of Google Earth Plus® (obtained in October and November, 2005) and the aerial photos from the Santos municipality cartographic basis (year of 2002). Field confirmation was done using GPS, and a new map of the mangrove forest area was developed comprising the area inside the limits of ECOMANAGE Project, that is São Vicente, Cubatão and parts of those from Guarujá, Praia Grande and Santos (the eastern limit was the Largo do Candinho - Bertioga channel).

To obtain information about the structure and litter production of the mangrove forests, monthly data were obtained based on 20 belt-transects in the estuarine area named Bertioga Channel (BERT), Diana River (RDIN), Jurubatuba River (JURB), Barnabé Island (BARB), Nossa Senhora das Neves Stream (NEVE), Quilombo River (QULM), Piaçaguera Channel (CPCG), Pedreira River (RPDR), Duas Barras Island (IDBS), Cubatão River (RCUB), Cascalho de Cima River (RCSC), Cascalho de Baixo River (RCSB), Mariana River (RMRN), Piaçabuçu River (RPCB), Queirozes River (RQRZ), Maranhos River (RMRH), Santana River (RSTN), Branco River (RBRC), Largo da Pompeba (LPPB) and Barreiros Channel (CBRR) (Figure 3). The belt-transects were placed randomly in each area and were divided in four 10x10 m parcels located at the fringe portion, the end portion and two others located at equitable distance between the extreme points.

The phytosociological characteristics were evaluated according to Schaeffer-Novelli and Cintrón (1986) using the tree density, height and diameter at breast-height (DBH) as structural measurements from the mangrove trees according to the formulae given by Curtis and McIntosh (1950). From these data the Importance Value Index of Curtis (IVI) (Curtis 1959) was obtained, which is the sum of the relative density, relative dominance (based on the basal area) and relative frequency for each one of the mangrove tree species. This index expresses the relative importance of each species in the system.

The biomass production was evaluated at all the sudy sites according to the Brown (1984) methodology. For this purpose, we placed 240 baskets (2 mm mesh size) under the mangrove trees positioned at random, three baskets in each stratum, and using 3 m intervals as units. The litter production was collected monthly. The material was dried in stove and weighed. The dry weight was converted to biomass m^2 from the area of basket aperture and the daily production was calculated from the number of days between sampling.

Using Arcview® software the total mangrove coverage in the study area was calculated as 71.3 km^2, distributed over the municipalities as follows: Santos, 31%, São Vicente, 23%, Cubatão, 22%, Guarujá, 14% and Praia Grande, 10%. These values are very similar to those obtained by Lamparelli and Moura (1998) - in their case the value was a bit lower because they did not include the end portion of the Bertioga channel, which was done in the present study. Among the 20 studied sites, 10 of them presented *A. schaueriana* with the highest IVI (*R. mangle* presented a small difference, with 9 sites); and *L. racemosa* presented the highest IVI only in one of the sites (Piaçaguera channel) (Figure 4).

The litter production ranged from 432.9 $g\ m^{-2}\ y^{-1}$ (Piaçabuçu River) to 945.7 $g\ m^{-2}\ y^{-1}$ (Cubatão River) (Figure 5). The highest litter production was observed for R. mangle in Queirozes River (628.7 $g\ m^{-2}\ y^{-1}$), followed by *A. schaueriana* (541.7 $g\ m^{-2}\ y^{-1}$ in Mariana river) and *L. racemosa* (284.0 $g\ m^{-2}\ y^{-1}$ in Barreiros channel. The mean litter production for the study sites was equal to 675.8 $g\ m^{-2}\ y^{-1}$. The leaves represented 80% of the total litter production. The seasonal litter production was lowest at the end of the winter (September with 35.5 $g\ m^{-2}$) and the highest in the summer (February, with 84.3 $g\ m^{-2}$) (Figure 6). Maximum production for *R. mangle* (40.7 $g\ m^{-2}$) and *L. racemosa* (17.8 $g\ m^{-2}$) were also observed in September, but for *A. schaueriana* the largest production was observed in November (30.1 $g\ m^{-2}$).

2.1 Saltmarsh

The Baixada Santista estuarine area also comprises several saltmarsh areas (Figure 7). Saltmarshes provide large amounts of organic matter to estuarine and coastal food chains. This organic matter usually cannot be used directly by estuarine consumers. The large amounts of cellulose and lignin in its plant material make the digestion by most of the organisms difficult, except for microorganisms that possess special enzymes for that. Therefore, a large part of the saltmarsh primary production is consumed in the form of debris generated by the slow microbial decomposition. Lima et al. (1987) showed that 50% of the organic matter of the estuarine sediment in Alagado de Pedra de Guaratiba (Rio de Janeiro State, Brazil) consisted of salt marsh debris.

The salt marsh community is considered essential to coastal environment (Wiegert and Pomeroy 1981) for the creation of refuge area to mating, nursery, home and food for many birds, rodents, reptiles, fishes and crustaceans (Day et al. 1989, Darby 2006). One important group of saltmarsh plants belongs to the Family Poaceae, genus *Spartina*. In Brazil, Adaime (1976) observed the excellent adaptation of this genus to estuarine conditions, which allows it to tolerate climatic variations, salinity fluctuations as well as long exposition and flooding periods, forming dense banks in the intertidal region.

Many aspects of saltmarsh plants in São Paulo state were studied in the Cananéia area by Adaime (1976), Takeda (1988), Tararam et al. (1991), Flynn (1994), Tararam (1994), Attolini et al. (1997), Cunha-Lignon (2001, 2005), Cunha-Lignon et al. (2005), Esposito et al. (2006) and Oliveira et al. (2006), and in Itanhaém by Biudes and Camargo (2006). However, despite the importance of the Santos estuarine system on the São Paulo coast, no data was reported on its saltmarsh banks. The saltmarsh banks from the Santos estuarine system area mostly comprise *Spartina* from the species *S. alterniflora* and *S. densiflora*. Both the settlement area of the saltmarsh and its density were measured in situ for the whole estuarine system except in a small area nearby Cubatão, where its characteristics were evaluated by random samples. The total area occupied by *S. alterniflora* banks was 167,742 m^2 with a mean density of 77.4 individuals m^{-2} (Table 1; Figure 8). *S. densiflora*, occurred in an area of 14,786 m^2 with a mean density of 568.5 individuals m^{-2} (Table 1).

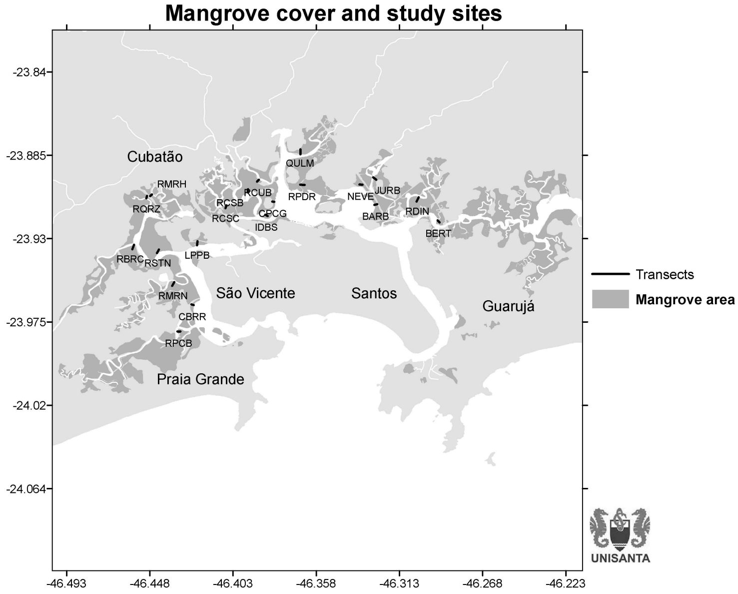

FIGURE 3: *Mangrove covered area and the location of the 20 transects for phytosociology and litter studies. See text for explanation of the abbreviations.*

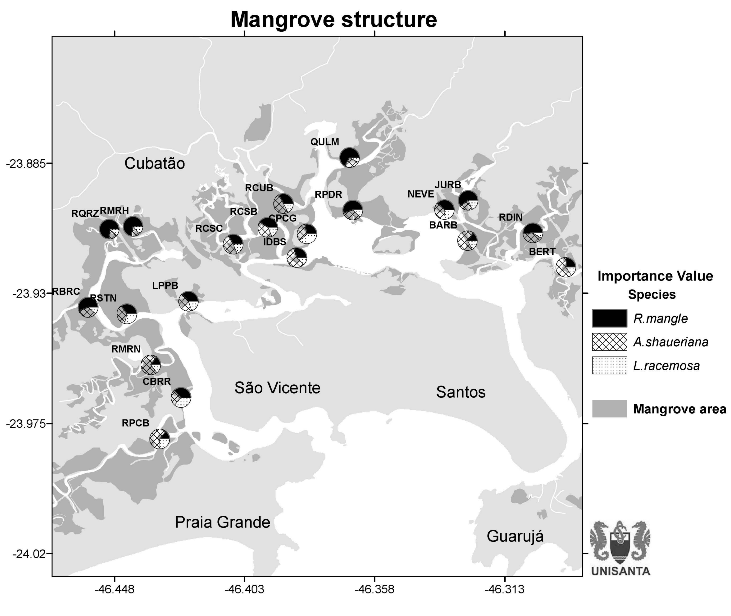

FIGURE 4: *Importance Value Index of the mangrove trees in the studied places.*

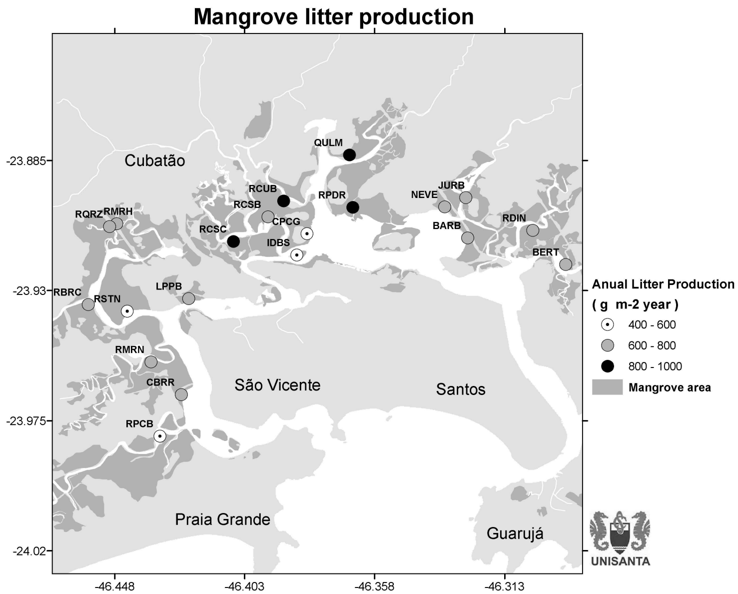

FIGURE 5: *Mean weight of the mangrove litter in the studied places.*

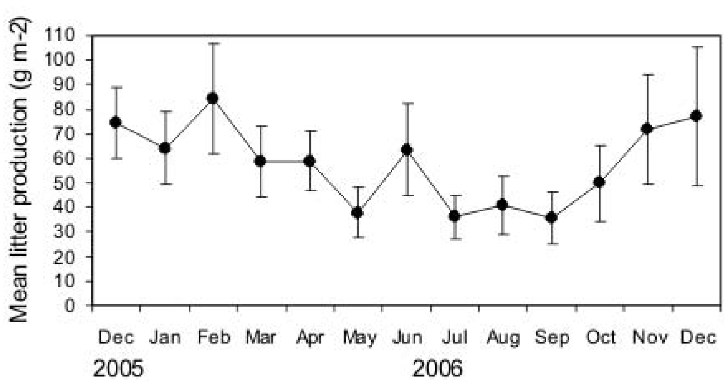

FIGURE 6: *Mangrove mean monthly litter production.*

TABLE 1: *Total area covered by Spartina ssp).*

	Spartina alterniflora	***Spartina densiflora***
Area (m^2)	167,742	14,786
Mean density (individuals m^{-2})	77.4 ± 59.3	568.5 ± 259.9

Three *S. alterniflora* banks were sampled to estimate live and dead biomass and above-ground net primary production (ANPP). Five random samples from each bank were obtained during eight months (March to November, 2007) using a 35x35 cm quadrat. The individuals were submitted to a close-cut to the soil, separated in alive and dead components by coloration and the living ones were counted. The plants were sorted by color difference into alive and dead components and dried in a stove for biomass evaluation. There was a tendency for a biomass reduction of both living and dead fractions from March to November with a small increase in August and September (Figure 9), which may be due to the small seasonal variation found in subtropical regions. It was also verified that the mean dead biomass weight was always larger than the live one and that the standard deviation was much higher for the dead biomass. The above-ground net primary production (ANPP) of the *Spartina alterniflora* banks was estimated during a period of eight months by the Smalley method (Long and Mason 1983). The mean value of the three banks for this period was 1481.7 g m^{-2}.

3 PHYTOPLANKTON

The phytoplankton community may be an important contributor to primary production even in estuarine systems where other producers are quite relevant. The phytoplankton community in Santos estuarine system is dominated by phytoflagellates although diatom blooms are also a common feature. Gianesella-Galvão (1978) reported *Skeletonema costatum* blooms reaching up to 99% of the bulk phytoplankton density during the summer in the Bay of Santos. This author considers other phytoplankton genera also important: *Cyclotella, Thalassiosira, Chaetoceros, Coscinodiscus, Asterionella, Leptocylindrus* and *Nitzschia*. At that time, at the end of the 1970´s, high mercury levels were detected in the Santos estuarine system, and several species of phytoplankton presented morphological abnormalities.

FIGURE 7: *A typical Spartina alterniflora bank in the Santos estuarine system (photo ECOMANAGE by Mauricio P. dos Santos).*

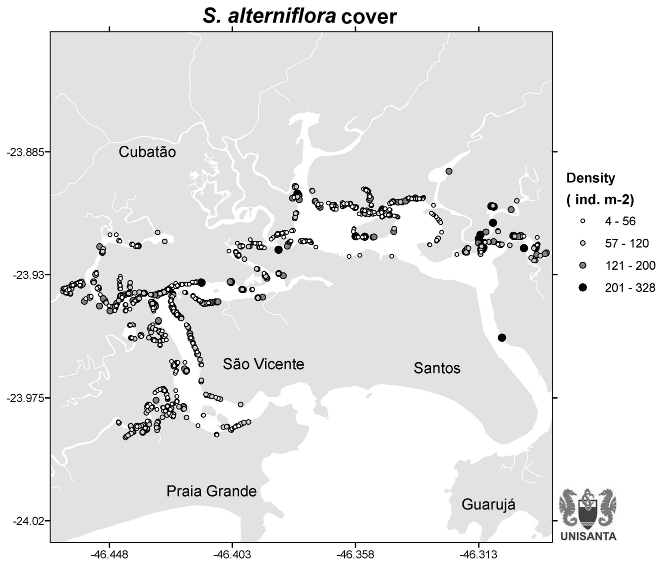

FIGURE 8: *Spatial distribution of S. alterniflora banks in the Santos estuarine system. Circles indicate occurrence of the banks.*

FUNDESPA (1998) reported nanophytoplankton densities ranging between 2.88 to 5.87 x10^6 cells l^{-1} in Santos bay. The community was composed mainly by phytoflagellates smaller than 6μm, *Thalassiosira* sp, pennate diatoms and coccolithophorids. The microplankton fraction was dominated by *Chaetoceros compressus, C. curvisetus, C. socialis, Guinardia* sp, *Pseudo-nitzschia* sp and *Leptocylindrus* sp, with densities ranging from 1.89 to 3.73 x10^5 cells l^{-1}. Moser (2002) identified the species *Skeletonema costatum* and *Skeletonema* sp as the main components of the nanoplankton in Santos and São Vicente channels. Other important genera were *Chaetoceros, Thalassiosira* and *Cyclotella*. Among the microplankton forms the more abundant genera found by this author were *Navicula, Thalassionema, Pseudo-nitzschia, Nitzschia, Rhizosolenia, Coscinodiscus, Asterionellopsis, Guinardia, Leptocylindrus, Hemiaulus* and *Rhaphoneis*. Cyanobacteria were also important in the summer, in the inner portions of São Vicente Channel and Largo do Canéu. In this place the dinoflagelates *Prorocentrum gracile* and *P. scutellum* were also important.

More recently, Ancona (2007) observed the nanoplankton (largely represented by phytoflagellates and some cyanobacteria) as the dominating phytoplankton community. The following diatom genera were the main components of the microplankton of Santos Channel and Bay: *Actinoptychus, Chaetoceros, Corethron, Coscinodiscus, Cyclotella, Dactyliosolen, Guinardia, Hemiaulus, Leptocylindrus, Melosira, Meuniera, Odontella, Skeletonema, Stephanopyxis, Thalassiosira, Triceratium, Asterionellopsis, Cymbela, Cylindroteca, Diploneis, Fragilariopsis, Navicula, Nitzschia, Pleurosigma, Pseudo-nitzschia, Rhizososolenia* and *Thalassionema*. Among the dinoflagellates the main genera found were: *Ceratium, Gonyaulax, Oxytoxum, Peridinium, Prorocentrum* and *Protoperidinium*.

The occurrence of harmful algal blooms in coastal areas under intense anthropogenic pressures has been an increasing concern especially due to its consequences for the public health. According to Hallegraeff et al. (2003), phytoplankton blooms have become more frequent since the 70´s. In Brazil there are few studies focusing on this problem (Owen et al. 1992, Gianesella-Galvão et al. 1996, Proença 2004, Carvalho et al. 2007). In the Santos estuarine system, *Skeletonema costatum* blooms were reported by Gianesella-Galvão (1978) and Moser (2002). Despite the fact that these species do not produce toxins, Hallegraeff et al. (2003) warned about their potential harmful effect, whatever that might be. Another matter of concern is the relative increase of dinoflagellates in the phytoplankton community recently observed in Baixada Santista (Moser 2002, CETESB 2004), since a great number of species in this group are potential toxin-producers. Gianesella and Saldanha-Corrêa (2003) observed the occurrence of the genus *Alexandrium* near the sewage outfall in Guarujá, besides the cyanobacteria of the genus *Anabaena*, typical of polluted waters. Carvalho et al. (2007) reported a *Trichodesmium erythraea* bloom along a broad area of the Baixada Santista off shore.

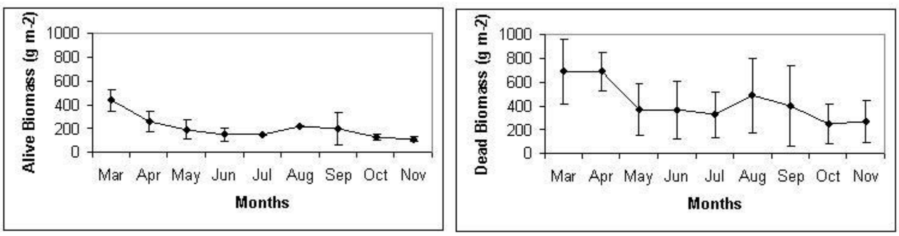

FIGURE 9: *Alive (left) and dead (right) biomass variation of the S. alterniflora for the eight studied months.*

Gianesella-Galvão (1982) reported primary production rates as high as 488.4 mg C m^{-3} h^{-1}, pointing out this system as one of the most productive tropical estuaries indicating the effect of the urban and industrial effluents on phytoplankton biomass development in Santos estuary. The Chla concentration was also high (maximum of 55.32 mg m^{-3}). Further studies (FUNDESPA 1998, CETESB 2001) corroborate the high eutrophication level in this area. Recently, Moser (2002) computed the instantaneous transport rates from São Vicente and Santos Channels inlets to Santos Bay and observed that the nutrient inputs via these channels were the main sources of the eutrophication of such water body, and the inputs from the submarine outfall were responsible for a small fraction of this total input.

Several studies since 1975 (Gianesella-Galvão 1982, Moser 2002, CETESB 2005) have demonstrated an increase in Chla concentrations in the estuary through the last 30 years. Moser et al. (2005) recorded a maximum of 97.4 mg Chla m^{-3}. Ancona (2007), based on monthly observations during a year in Santos Channel and Bay, found phytoplankton biomass ranging from 0.85 to 28.08 mg m^{-3} in the bay and from 1.93 to 65.28 mg m^{-3} in Santos Channel. Primary productivity in channel waters ranged between 1.34 to 48.75 mg C m^{-3} h^{-1} and from 0.09 to 24.91 mg C m^{-3} h^{-1} in bay waters. The lowest biomass and production rates were detected in the winter period. According to this author, light and water column stability are the main factors for phytoplankton development in Santos estuary. The highest Chla values reported for Santos Estuary by the several authors cited above, are in the range of hypertrophic environments as stated by Smith et al. (1999).

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ZOOBENTHOS OF THE SANTOS ESTUARINE SYSTEM

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1 INTRODUCTION

ECOMANAGE, which applies the DPSIR methodology, has identified a number of human activities in the Santos Estuarine system which lead to a high level of pressures on this ecosystem. Benthic fauna has been shown to be a very good indicator of such pressures, since it is directly exposed to the constant changes of state suffered by the estuary. Although the objectives of the project did not include a zoological survey of the area, the ECOMANAGE data field campaigns on benthic fauna and its respective detailed measures of diversity and biomass, were very useful to indicate the state of both water and sediment.

2 SOFT-SEDIMENT SUBLITTORAL ZOOBENTHOS

The study of soft-sediment benthos communities within the Santos Estuarine System was developed owing to the scarcity of available data. For these reasons, it was decided that the collection and analysis of the macro-benthic organisms inhabiting the soft sediments in the estuarine channels should be included, looking for determining communities' basic characteristics and their specific composition along the estuary. It was made with samples from eight stations regarded as strategic in terms of impacts, water flow and nutrient drainage (figure 1). One expects, also, to identify the possibly most sensitive areas to human activities. In this work soft-sediment macrobenthos on eight stations in the Santos Estuarine System were sampled, both in winter (August, 2005) and summer (February, 2006) campaigns.

Twenty-eight taxonomic groups were registered in all of the samples (Table 1). These groups comprised 11 phyla or subphyla of benthic animals: Cnidaria, Platyhelminthes, Nemertea, Nematoda, Mollusca, Annelida, Sipuncula, Echiura, Arthropoda (Subphyla Cheliceriformes and Crustacea) and Chordata (Subphyla Cephalochordata). Among those groups, some individual organisms could not be identified to a lower level owing to fragmentation of the specimens during the sampling procedure; The relative abundance of major groups in all samples pooled can be seen in Figure 2.

The faunistic composition was similar to that observed in other surveys in estuarine regions, particularly comparing with the results obtained by Abessa (2002) in the Santos Estuarine System, and by Hostin (2007) in the Paranaguá Estuarine Complex (Paraná state, BR), which also is an important port terminal complex. Polychaeta was the main group in general, followed by Gastropoda (Figure 2). A predominance of annelids was reported before by Heitor (2000) in the Santos Bay and by Abessa (2002) in the Santos estuary. Along the port of Santos and Cubatão channel sites (i.e. 2 and 5) the number of individuals in the main groups was much higher in winter than in summer, particularly Polychaeta, Gastropoda and Crustacea in site 2, and Polychaeta in site 5. However, the opposite was observed in the sites deeper in the estuarine system like the São Vicente internal area (on site 6 of Pompeba for instance, Polychaeta was more than three times more abundant in the summer than in the winter) and in Santos site 4 (Caneu), Polychaeta was almost four times more abundant in the summer than in the winter.

The two sites that showed the highest contrast in species richness were those located along the port of Santos and Cubatão channel (sites 2 and 5, respectively). Site 2 was the richest in both seasons and site 5 was the poorest in species in both seasons (Figures 3 and 4, Table 2). The Shannon-Wienner diversity index (H' in bits) follows the richness, with site 2 the most diverse and site 5 the least diverse (Table 2). It is remarkable that the number of groups was higher in site 2 than in the other sites studied. Other works reported high toxicity near this site in the recent past (Abessa 2002); the same was detected in site 5 area (Sousa et al. 2007). The other sites suffer more impact of domestic sewage.

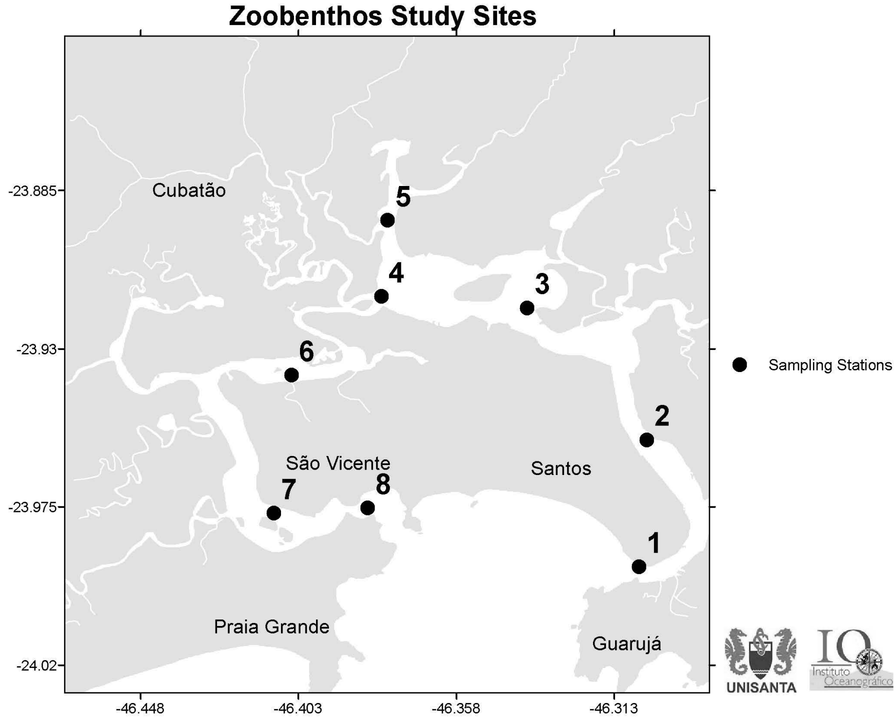

FIGURE 1: *Sampling points of the estuarine Benthos.*

In only one site Polychaeta was not the most abundant group: the site 8 (Gonzaguinha), where Gastropoda was many times as abundant as the other groups, probably related with sediment (sand with grain size larger than in other sites) and relative position in the estuary (Figure 5).

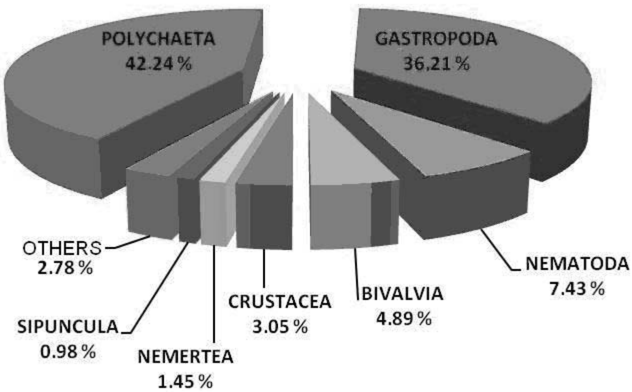

FIGURE 2: *Relative abundance of the main groups; data of both seasons and all sites is pooled.*

TABLE 1: *Taxonomic groups and total abundance in both seasons.*

Groups	Abundance	Groups	Abundance
Annelida 1	12	N2*	4
Polychaeta	1080	N7*	1
Oligochaeta	25	N8*	2
Gastropoda	926	N11*	1
Bivalvia	125	N13*	5
Nematoda	190	N14*	8
Crustacea	78	N15*	1
Nemertea	37	N16*	2
Sipuncula	25	N20*	1
Anthozoa	9	N21*	2
Cephalochordata	11	N23*	3
Echiura	4	N24*	1
Hidrozoa	2	N25*	0
Hirudinea	1		
Platyhelminthes	1		

* Groups which were not identified but safely isolated from the identified ones.

TABLE 2: *Extreme values of species richness and diversity (Shannon-Wienner H' - bits) found along the estuary's studied sites: 2 (TECON) and 5 (Piaçaguera Channel).*

Index	Site 2 (TECON)		Site 5 (Piaçaguera)	
	Winter	Summer	Winter	Summer
Richness	10	18	4	2
Shannon (H')	2.082	2.861	0.666	0.890

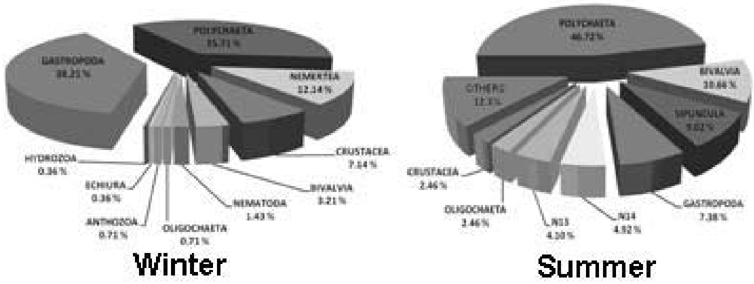

FIGURE 3: *Relative abundance of the main groups in site 2 (TECON) in the winter (left) and in the summer (right).*

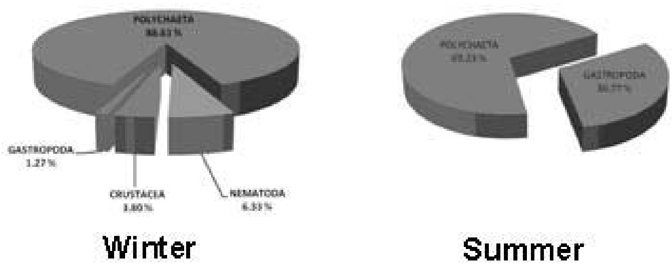

FIGURE 4: *Relative abundance of the main groups in site 5 (Piaçaguera Channel) in the winter (left) and in the summer (right).*

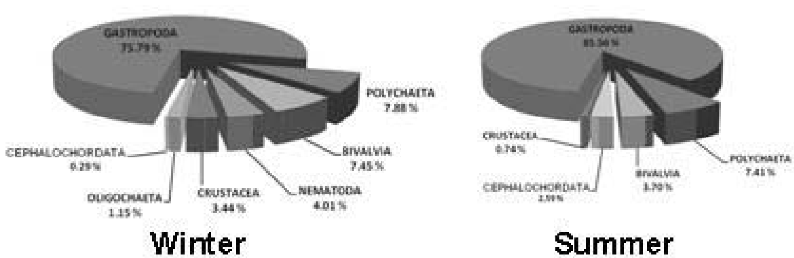

FIGURE 5: *Relative abundance of the main groups in site 8 (Gonzaguinha) in the winter (left) and in the summer (right).*

The number of major groups didn't show a clear correlation to either season or organic matter content of the sediment. The same was observed in relation to sediment grain size, because in almost all sites the sediment was sand-silt (70% fine sand), typically mud. These results were not surprising and were reported in other works performed in estuaries around the world. Sediment analyses showed no direct relation between grain size and organic matter content. It's established that, for coastal waters, this kind of relation is commonly direct (Levinton 2001, Nybakken 2001). However, in estuarine systems, particularly those with mangrove forests, the large amount of organic matter that comes to the bottom may be more significant than other factors to determine the content of this matter in the sediment.

The Santos Estuarine System is affected by human occupation and it, perhaps, determines the amount of mineral particles that comes to the water mass. An area that receives a large amount of sewage, as revealed by socio-economic survey and also observed by Bonetti (2000) was Largo da Pompeba (6), which contained a high number of major groups and a great abundance of organisms. The large number of factors affecting soft benthos communities in the Santos Estuarine System, from natural complexity of the system to human interferences, makes it very difficult to reveal and understand how the main forces work. These factors change in kind and intensity along the estuary and perhaps only with integrated studies, like ECOMANAGE program, a general understanding of this kind of system will be possible.

3 HARD SUBSTRATE SUBLITTORAL ZOOBENTHOS (FOULING COMMUNITIES)

The ECOMANAGE project conducted several experimental studies in order to find ways to characterize Santos' estuarine ecosystems in an integrated way, according to the DPSIR (Drive, Pressure, State, Impact and Response) methodology. The Fouling Project, developed as a sub-item of the estuarine biological community fauna studies, evaluates the community of encrusting marine sessile organisms in the indicative search of pressures to the functioning state of the estuarine ecosystem.

The community of these encrusting invertebrates has been regarded as a pressure to the system in the whole world for its importance to shipping, mainly due to the economic problems that are caused by encrustation in boats and ships, piers, floating buoys and submarine cables. The structure of "fouling" in the estuarine region of Santos and São Vicente was observed during summer and winter seasons at four distinct points in the estuary (Figure 6). In each one of them, four ceramic plates were used as fouling collectors during a period of three months. The analyses were carried out through the analysis of diversity, biomass and volume of the collected organisms, in order to obtain a qualitative and quantitative set of data for hard substrate sessile organism distribution along the different sites according to Giordano (2001) and Borges (2002).

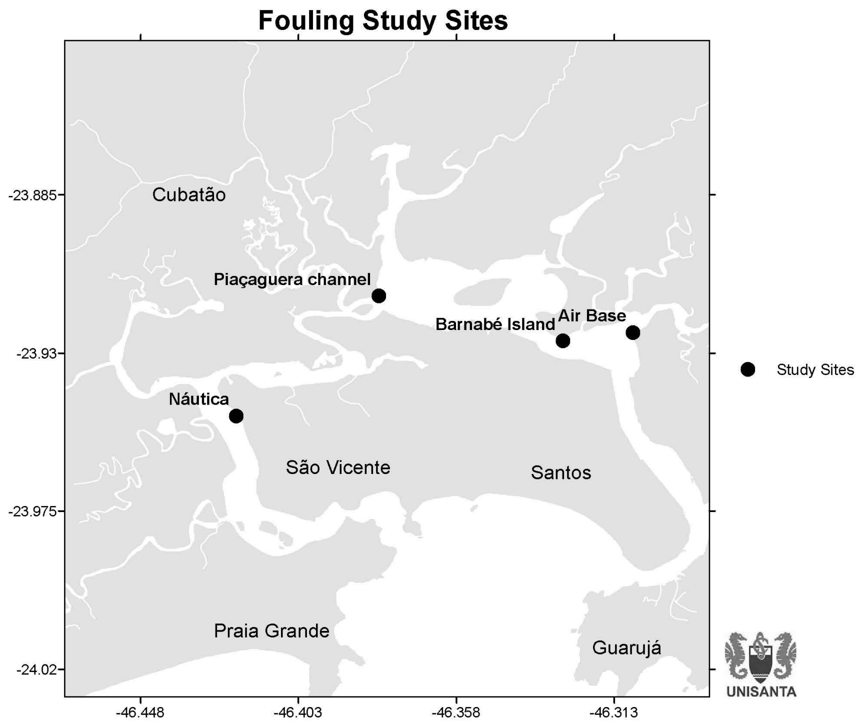

FIGURE 6: *Site studies where ceramic tiles were used as fouling fauna samplers.*

The results show values of the diversity indexes to be higher during the winter season and values of biomass and volume to be higher during the summer season. It was also verified that approximately 65% of the taxonomic groups found had been observed in both seasons of the year (Figure 7 showing the plates with different taxonomic groups among them sea-squirt, bryozoans, barnacles, sea worms and mussels were the most abundant groups). The results of richness and diversity indexes obtained in São Vicente were very low, probably due to a great ousting of sewers, coming from the palafittes, sub-houses from which sewage directly pours into the estuary and prevents the settlement of more sensitive members of the fouling community. However, in this same region, the biggest growth rates of the fouling community's biomass occurred, mostly due to the incredible rates of bivalves' filter feeders growth (up to 11 g of dry weight of organic matter a day per plate!). Fouling, as well as other bio-indicators, demonstrates that the greatest pressures in the estuarine system come from sewage and sub-normal residences at the edge of the estuary whereas along the port channel the shipping activities played the important role of new species introduction by ballast water discharges.

FIGURE 7: *Ceramic tile used for fouling community recruitment (source: ECOMANAGE photos by Leandro Kodama).*

4 FINAL CONSIDERATIONS ON BENTHOS

As the fouling community fauna depends much more on water column conditions while soft-sediment fauna is more frequently related to sediment quality we can conclude that:

- In Cubatão and Santos Estuarine areas (Northern and Eastern part of the estuary) the growth of the fouling fauna was not as high, but instead, higher diversity indexes seemed to be explained by the relative lower levels of sewage discharges in these waters;
- Instead, it seemed to be related to a high growth of fouling fauna, the negative effects of waters contaminated with sewage discharges, noticed in São Vicente estuarine area (western area of the estuary) also supported by the results of the water column quality tests obtained by ecotoxicology procedures;
- Soft-sediment Benthic Fauna biodiversity reflects the high levels of sediment contamination, historically reported in the estuary as a major pressure for the ecosystem. This could be confirmed with the ECOMANAGE field data analysis, by the very poor indexes obtained on diversity and the high dominance obtained of these values whenever the site studied was on the Santos Cubatão side of the estuary (places subjected to former dredging activities) such as the Piaçaguera Channel;
- The benthos survey was an important tool in order to evaluate the environmental ”status” as affected by two factors: the suspended organic load and the sediment organic matter content. The abundance of the first factor showed a close relation to high levels of fouling communities’ biomass increase; nevertheless, the abundance of the second factor showed no close relation to the number of individuals and taxa in all sites.

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ECOLOGICAL STATUS OF THE SANTOS ESTUARY WATER COLUMN

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1 INTRODUCTION

The Santos Estuary, in São Paulo State, Brazil, is known for its high pollution levels related to port, industrial and urban uses. Despite its importance and strategic location in the national context, few systematic studies have been conducted aiming to determine the ecological status of the system. There is no available data on the Santos Estuary's pristine conditions, i.e. before the occupational boom occurring in this area at the end of the 1960s and early 1970s, with the development of the Cubatão industrial park and Santos port expansion, besides the urban center.

The first extensive survey in Santos Bay was carried out between 1974-1977 to establish the original environmental conditions before the construction of the Santos submarine outfall (CETESB 1978, Gianesella-Galvão 1978, 1982), regarding the future management of the area. The findings indicated that the environment was heavily impacted: phosphate values varied from 1.44 to 4.22 μM and mean total inorganic nitrogen varied from 10.84 to 21.53 μM, with nitrate concentrations attaining 19.28 μM. However N:P ratio never exceeded 10. At that time, Hg was also detected in high concentrations, with values attaining 3.64 μM, 10 times the maximum detected by Fitzgerald and Lions (1973) in the seawater. The highest nutrient concentrations were detected in the inner estuary portions, decreasing towards Santos Bay. Such gradient were consistent with the circulation model suggested by Garcia-Ochipinti (1972). According to this author, during high tide, the seawater enters the estuary through the bottom layers, whereas brackish water flows downwards through surface layers (0-1 m). This is the predominant pattern about 85% of the time.

The Cananéia Estuary (25°04 'S), south of the Santos Estuary, was chosen as a comparative model since it is relatively near the Santos Estuary (located at neighboring latitudes and geographically at the same embayment) and displayed similar climate, soil and mangrove vegetation, despite that each estuary presents their particular characteristics. Kato (1966) reported 3.0 μM nitrate and 0.7 μM phosphate as maximum concentrations in Cananéia Estuary. Such values are a clear indication of the eutrophication process that the Santos Estuary underwent from 1974-1977. By the mid 1970s, domestic and industrial effluents had begun to be governmentally regulated. In the 1980s, the state environmental protection agency (CETESB) initiated a program for controlling the main sources of pollution in the Santos harbor and industrial park of Cubatão. An effort was also made by the authorities to control clandestine sewage disposal. However, subsequent studies (FUNDESPA 1998, Braga et al. 2000, Frazão 2001, CETESB 2001, Moser 2002, Ancona 2007) indicate that eutrophic conditions still persist. Braga et al. (2000) verified nitrate and phosphate concentrations higher than 90 and 24 μM, respectively, near Cubatão city. These authors affirm that the concentration declines towards Santos Bay; nevertheless, low concentrations are not attained for both elements in bay waters. Frazão (2001), comparing the trophic state of waters of Ubatuba, Santos and Praia Grande, concluded that Santos is the most eutrophic of the three. CETESB (2001) verified high levels of heavy metals in the sediments and biota from Santos and São Vicente estuaries. Moser (2002) verified nitrate and phosphate concentrations higher than 12 and 4.0 μM, respectively, in Santos and São Vicente Channels pointing out that the nutrient input from these channels into Santos Bay are much more relevant than the discharge of the sewage submarine outfall (located at the middle of Santos Bay).

Ancona (2007) studied the time-space variability of nutrients, chlorophyll-a (Chl-a) and other related variables in Santos Channel and Bay, based on monthly samplings from November 2004 to December 2005. This work represents the best data set available to characterize this region, also notably referring to a very recent period. The author reports dissolved inorganic nitrogen attaining values up to 41.1 μM in Santos Channel and up to 35.1 μM in Santos Bay. Ammonium was the main inorganic nitrogen form (maximum of 32.5 μM in Santos Channel and 21.5 μM in the Bay). Ammonium availability was negatively correlated to salinity, indicating that its source is the brackish waters, being more important in rainy periods. On the other hand, nitrate concentrations were higher in the driest periods. Phosphate maximum concentrations were 11.31 μM in the bay and 8.38 μM in the channel waters. The source of phosphate for the system is also the inner channel waters, as attested by negative correlations between salinity and phosphate. The low N:P ratio is still observed in the area, due to the phosphate overload, and not to low nitrogen availability. In terms of silicate, the mean value in channel waters was nearly 15 μM, while in the bay waters the mean was lower, approximately 10 μM. Such silicate levels are high denoting an environment under the influence of great land drainage.

The current status of the estuarine system was evaluated by samplings performed in 8 stations distributed throughout the estuarine region (Figure 1) in winter (August, 2005) and summer (March, 2006).

2 MATERIAL AND METHODS

In situ vertical profiles of physical properties (T, Salinity, pH) were performed using a HORIBA multi-parameter sensor. Secchi disk readings were used to estimate the euphotic zone thickness (Zeu), according to Poole and Atkins (1929). Discrete water samples for chemical variables were taken with van Dorn bottles at three depths: surface, mid-water and 1 m above the bottom. Dissolved oxygen (DO) and alkalinity analyses were processed in the field laboratory, within 6 h after sampling. DO samples were analyzed by Winkler's method (Grasshoff et al. 1983) and oxygen saturation (DOS) was computed according to UNESCO (1973). Alkalinity determinations were performed by titration using HCl 0.05 M (Gran 1952) and the data were used to estimate the dissolved CO_2 concentrations based on Carmouze (1994) computations. Aliquots of water were filtered through AP-40 Millipore® glass fiber filters for photosynthetic pigment analyses (Saldanha-Corrêa et al. 2004) and seston determination. The filtered water was frozen for the dissolved inorganic nutrient analysis.

Inorganic nutrients were analyzed by spectrophotometric methods: nitrite, nitrate and ammonium according to Aminot and Chaussepied (1983) and phosphate and silicate following the methods described in Grasshoff et al. (1983). Chlorophyll-a (Chl-a) was determined according to Jeffrey and Humphrey (1975) equations, active chlorophyll-a (Act Chl-a) and pheopigments-a by Lorenzen (1967) and carotenes according to Parsons et al. (1984). Seston analyses were performed by the gravimetric method discriminating inorganic and organic fractions (APHA 1985) using pre-combusted and pre-weighed AP-40 Millipore® filters. The phytoplankton internal nutrient status was also evaluated using the ratio between the optical density at 480 and 665 nm of the samples used for pigment determinations (Heath et al. 1990).

Primary productivity rates (PP) were determined in surface and bottom samples of stations 1, 2, 3, 5, 6 and 8 in winter and 1, 3, 5, 6 and 8 in summer, through incubations using ^{14}C (Steemann-Nielsen 1952) in photosyntetron-like chambers. The ^{14}C assimilated was converted to units of mg C m^{-3} h^{-1}, according to the equations described in Teixeira (1973). Aliquots of the same samples submitted to primary productivity experiments from stations 1, 2 and 3 in winter and 1, 3, 5, 6 and 8 in summer were taken to determine the bacterial secondary productivity (BSP) following the method proposed by Smith and Azam (1992), using $^{3}H - Thy$ instead of ^{3}H – Leucine. The number of disintegrations per minute of each sample was converted to moles of incorporated $^{3}H - Thy\ L^{-1}\ h^{-1}$ according to the equation proposed by Bell (1993). The conversion of moles incorporated to cells produced per unit time was done by applying the thymidine conversion factor (TCF) of 0.52×10^{18} cells per mol of thymidine determined by Barrera-Alba et al. (2004) in Cananéia Estuary. A second conversion, from cell production to productivity in units of carbon was done assuming the factor of 20 fg C $cell^{-1}$ (Lee and Fuhrman 1987).

Considering the complex composition of the industrial and urban sewage discharged in Santos Estuary due to several types of pollutants, ecotoxicologic tests were also performed in order to gain complementary information. These bioassays are useful for evaluating the environmental degradation process since they determine the synergistic effect of the chemical action of the toxic agents, the physiology of the test organism and the environmental conditions (FUNDESPA 2007, Rachid 2002).

Toxicity tests with surface water were conducted using non-diluted water samples. The embryolarval development test with *L.variegatus* followed the ABNT (2006a). Chronic toxicity tests were conducted for 24 h, after this period the percentage of normal pluteus larvae was evaluated. For the acute toxicity tests with the surface water, the bacteria Vibrio fischeri was used according to ABNT (2006b). This bioassay consisted of determination of the inhibitory effect of water samples on the light emission of these luminescent bacteria, for 5, 15 and 30 min. The ecotoxicity data were compared statistically using the MSD (Minimum Significant Difference) method and analysis of variance followed by the Dunnett's test for detection of significant differences and determination of the toxic samples (ASTM 1992, Swartz et al. 1985, USEPA 1991).

3 RESULTS AND DISCUSSION

The Santos Estuary is a partially stratified system, with sharper stratification in São Vicente Channel and the inner estuarine areas. During winter samplings, water temperature was near 22 °C and salinity ranged from 25.7 in station 6 up to 33.7 in the channel inlets. During summer samplings, water temperature was nearly 6 °C higher than during winter, with a mean value of 28 °C. Due to the higher summer rainfall, which increases the freshwater contribution in the estuarine head, salinity mean value was 23.07, i.e. about 8 units lower than in winter. Water density in summer was much lower than in winter as a result of the high temperatures and low salinities (Table 1).

TABLE 1: *Mean ($\pm$SD), maximum and minimum values of the physical and chemical variables obtained in winter and summer sampling cruises in the 8 stations distributed throughout the Santos estuarine system. (Sigma-t in kg m^{-3} and dissolved inorganic nutrients in μM).*

Aug/05	T (ºC)	Sal	Sigma-t	pH	OD (ml l^{-1})	DOS (%)	CO2 (%)	CO2/ DO	NH_4^+	NO_3^-	NO_2^-	DIN (μM)	PO4-3	N:P	Si(OH)4
Mean	22.04	31.25	21.33	8.63	4.45	87.32	307.81	3.47	12.85	6.15	1.45	20.44	4.89	5.07	15.07
SD	0.46	2.63	2.18	0.12	0.68	14.07	158.47	3.07	11.28	8.27	1.03	15.13	4.39	2.69	5.04
max.	23.10	34.90	25.00	8.70	5.77	114.90	621.40	10.17	51.31	41.84	4.93	58.07	22.40	12.56	28.54
min.	21.40	23.70	16.00	8.30	2.69	52.00	135.74	0.00	0.66	1.20	0.15	6.96	0.55	1.65	7.41

Mar/06	T (ºC)	Sal	Sigma-t	pH	DO (ml l^{-1})	DOS (%)	CO2 (%)	CO2/ DO	NH_4^+	NO_3^-	NO_2^-	DIN	PO4-3	N:P	Si(OH)4
Mean	27.99	23.07	13.61	7.29	3.85	80.38	721.13	9.35	20.12	5.18	1.12	26.42	3.81	8.51	24.21
SD	0.60	3.71	2.79	0.28	1.06	24.49	446.16	7.52	16.40	5.37	0.77	19.33	2.27	8.96	11.84
max.	29.12	28.80	18.00	7.80	6.17	134.28	1835.33	26.52	69.76	23.93	3.29	78.04	8.33	44.35	46.65
min.	27.03	16.70	8.00	6.91	2.52	51.66	189.09	0.00	0.33	0.71	0.21	3.72	0.81	2.45	5.99

Dissolved oxygen (DO) mean concentrations and DO saturation percentage (DOS) were higher during the winter than in summer (Table 1). DO levels were always lower in the inner brackish waters (with the minimum at station 6) tending to increase towards the channel mouths. DO levels were above the minimum recommended limit by Brazilian environmental legislation (CONAMA 2005) but in stations 6 and 7, the averages were very close to this limit. The DO saturation levels were mostly below 100% indicating a deficit in O_2 production related to its consumption, mainly in summer.

Mean pH value in winter was more than one unit higher than the summer mean. The average concentrations of dissolved CO_2 in the water column were also higher in summer than in winter but in terms of percentage of CO_2 saturation, the estuarine waters always presented CO_2 over saturation (Table 1), indicating an intense heterotrophic activity. Summer values were twice as high as winter ones, and attained more than 1500% saturation at stations 4 and 6. An estimate of the balance between heterotrophic and autotrophic processes was obtained using the ratio CO_2/O_2 saturation. Results indicate strong heterotrophic behavior in this system with a significant increase during summer.

The inorganic nutrient concentrations detected throughout the estuary in summer and winter confirmed the eutrophic characteristics (Table 1). Some points, such as station 6 site, displayed worse conditions: in summer, the maximum ammonium concentration obtained (69.7 μM) was within the limit of acceptance for brackish waters according to the environmental legislation (CONAMA 2005). The region's nearby station 6 is shallow and receives the runoff of a garbage disposal on the island side with high loads of ammonium, phosphate and many other toxic components, not quantified in this study. Ammonium was the main inorganic nitrogen form in both sampling periods but mainly in winter, corresponding to almost 90% of the total dissolved inorganic nitrogen. Ammonium and silicate were clearly associated with waters of continental origin since the highest values were detected at the inner stations (4 to 7) where ammonium ranged from 21.57 to 51.52 μM in summer, whereas the maximum silicate concentration (Table 1) was attained at the innermost station (st. 5). Ammonium is also produced by organic matter decomposition and the high summer temperatures certainly contributed to the increase of this activity.

Inorganic phosphate, otherwise, seems to have an inner local source, relatively independent of rain contribution, since higher concentrations were detected during winter (Table 1). Currently, there are several possible sources of nutrients in the inner areas, including domestic and industrial effluents, and these sources are relevant, as indicated by the absolute nutrient concentrations. The silicate and phosphate concentrations observed were a bit higher than those observed by Braga et al. (2004) in the inner estuarine regions. On the other hand, the nitrate values were much lower than those reported by these authors, who did not evaluate the ammonium contribution to the system. Despite the high concentrations of total inorganic nitrogen verified in both periods, the N:P ratios (Redfield 1934) were always below 10, due to the high phosphate loads.

Seston distribution presented the same pattern in summer and winter, with higher concentrations in Santos Channel, almost 5 times higher than the mean values obtained in São Vicente and inner estuary stations. The organic fraction (OS) corresponded to around 30% of the bulk (Table 2). The high seston load is an important factor in reducing the euphotic zone thickness. Indeed, light penetration was usually low, between 2 and 5m, and may be limiting phytoplankton growth in the subsurface layers, as Santos Channel can reach up to 15 m in depth.

The estuary is characterized by its high phytoplankton biomass. Chl-a levels are compatible to the observed in eutrophic system as stated by Smith et al. (1999), with mean concentration around 10 mg m^{-3}. Phytoplankton biomass was slightly more elevated during winter than summer (Table 2). During summer, the spatial Chl-a distribution tended to be more homogeneous. In winter, Chl-a concentration tended to increase from Santos Channel towards the inner estuary with a maximum value at station 7. Conversely, station 7 presented the lowest biomass in summer. The lower Chl-a values observed in the upper reaches may be ascribed to the smaller retention time of the waters at this period, due to the larger contribution of freshwater into the system. The main vertical differences were observed in the inner stations, principally in station 5, where the highest values occurred in surface waters.

TABLE 2: *Mean ($\pm$SD), maximum and minimum values of the variables: euphotic zone thickness (Zeu), total seston (TS), percentage of organic seston (% OS), photosynthetic pigments (in mg m^{-3}) PP and BSP (in mg C m^{-3} h^{-1}), PP/Chl-a (mg C (mg Chl)$^{-1}$ h^{-1}) and the ratio BSP/PP on a daily basis, obtained in winter and summer sampling cruises in Santos Estuary.*

Aug/05	Zeu	TS	OS	Chl-a	Chl-a act	Chl-c	Carotene	480/665	PP	PP/Cl-a	BSP	BSP/PP
Mean	2.80	179.02	31.65	9.73	6.55	2.15	5.57	1.21	21.72	2.68	0.31	0.06
SD	0.80	250.07	32.44	6.96	5.66	1.25	4.93	0.14	18.22	1.06	0.09	0.03
max.	4.20	766.20	94.07	30.30	27.07	5.76	22.42	1.59	60.80	4.43	0.41	0.10
min.	2.10	36.60	9.84	2.95	0.00	0.75	0.43	1.02	2.36	1.03	0.10	0.04

March/06	Zeu	TS	OS	Chl-a	Chl-a act	Chl-c	Carotene	480/665	PP	PP/Cl-a	BSP	BSP/PP
Mean	3.40	179.10	26.20	8.09	4.98	1.48	5.36	1.22	60.00	7.98	1.17	0.12
SD	1.50	258.65	35.31	2.82	2.12	0.56	1.97	0.23	75.00	12.39	0.95	0.14
max.	6.10	889.40	90.35	12.16	9.11	2.58	9.49	1.72	230.31	42.10	3.45	0.49
min.	1.90	32.50	0.00	2.78	1.09	0.53	2.60	0.95	3.87	0.55	0.37	0.01

Total carotene concentrations followed the same temporal distribution pattern of Chl-a, with mean concentration values slightly higher during winter than summer. The active Chl-a generally represented more than 50% of total Chl-a, which indicates a healthy state for the phytoplankton. The ratio 480/665 (Table 2) corroborates this assumption since all the values for this ratio were below 2, indicating a good intracellular nutritional status (Heath et al. 1990) in both studied periods.

The mean primary productivity ratio (PP) obtained in summer was higher than winter one (Table 2). PP rates obtained from surface waters were generally higher than those from bottom samples, except in st. 8 and 1 in summer. The highest rate was attained from the surface water of st. 6 (230.31 mg C m^{-3} h^{-1}) and the PP observed values in Santos Channel were always below those from the other stations, indicating that Santos Channel is the less productive portion of the system. This feature can be attributed to the higher depths verified in this portion of the estuary which restricts phytoplankton cells to remain light limited in this course throughout the mixing layer. Also, this channel shelters the Santos port, experiencing intense traffic of large ships, which increases the water instability and turbidity, negatively affecting the phytoplankton growth.

In terms of PP/Chl-a ratios, Santos inlet station presented the lowest value in both periods. In winter, mean ratio was 2.68 mg C mg Chl^{-1} h^{-1}, whereas the mean ratio in summer was almost four times this value (Table 2). These results may be ascribed to the high water temperatures observed in summer favoring the phytoplankton development in nutrient rich waters and at more stable conditions than those verified at the channel mouths. The highest PP/Chl-a ratio considering both sampling periods was 42.2 mg C mg Chl^{-1} h^{-1}, verified at surface of station 6 in summer.

The BSP rates observed in summer were almost twice those observed in winter (Table 2). A similar spatial pattern of PP/Chl-a ratios was observed for BSP in summer, with the highest value in station 6 (3.45 mg C m^{-3} h^{-1}) and the higher ratios near the surface. However, in terms of BSP, the differences between surface and bottom samples were usually small. Considering the results on a daily basis, the values observed in the Santos system ranged from 2.35 to 82.76 mg C m^{-3} d^{-1}, or 14.4 to 114.2 mg C m^{-2} d^{-1} in area terms. The BSP rates were higher than those obtained by De Souza et al. (2003) in the Zuari estuary, India, where the authors report mean rates of 70 mg C m^{-3} d^{-1} in the surface and 35 mg C m^{-3} d^{-1} in the bottom waters.

This environment shows physical similarities with Santos estuary as the temperature ranges, depth, mixing conditions, and mangroves in the surroundings. Another example of a tropical environment was the lowland rivers of Orinoco basin (Castillo et al. 2004) that presented values between 4.8 to 6.24 mg C m^{-3} d^{-1}, rates significantly lower than those observed in Santos estuary. If one considers the bacterial productivity rates in terms of area, the results obtained were extremely low compared to the ranges reported by Barrera-Alba (2004) in Cananéia-Iguape estuary (in summer: 459.0 to 12730.0 and in winter: 182.5 to 1354.5 mg C m^{-2} h^{-1}). However, the BSP rates obtained in Santos estuary are in the same range as those obtained by Hock and Kirshman (1993) in the Delaware estuary, with also similar values for Chl-a and PP rates, by Shiah et al. (2003) in the continental shelf of East China Sea BSP (6 to 179 mg C m^{-2} h^{-1}) and higher than those of Almeida et al. (2002) in the Ria de Aveiro system (1.5 to 36.8 mg C m^{-2} h^{-1}). Both PP and BSP rates were highest at station 6, demonstrating that this is an important place for the evaluation of the estuarine dynamics, which is producing and probably exporting materials toward the bay through São Vicente Channel (stations 7 and 8) favoring eutrophication processes in the bay.

The BSP/PP results indicate the dominance of phytoplankton autotrophic processes over the bacterial heterotrophic ones. According to these results, the ratio CO_2/O_2 was expected to be quite low. However, the CO_2/O_2 ratios clearly indicate the predominance of heterotrophic processes over autotrophic ones, mainly in the inner estuarine portions. Such differences may be ascribed to the heterotrophic activities not related to water column process. One of particular importance, not analyzed in the present study, is the bacterial decomposition activity in the mangrove sediments: certainly a great part of the CO_2 produced is dissolved in the water as the high tides invade the mangrove. Additionally, the heterotrophic activity in the estuarine sediment is possibly high, removing O_2 from and introducing CO_2 into the water column.

Seston analysis showed phytoplankton as a less important component of the organic fraction. This means that a great load of the organic material comes from the terrestrial or aquatic vegetation and also from domestic and industrial effluents discharged throughout the estuary, which decompose on the way to the open sea. Consequently, the BSP/PP ratios indicate the phytoplankton primary productivity is able to support the water column heterotrophic demand; nevertheless, other CO_2 sources determine the heterotrophic characteristics of the Santos Estuary.

The maintenance of high productivity rates and the dominant water column mixed conditions observed in the inner areas of the system during the summer prevented anoxic conditions. Acute toxicologic tests with *Vibrio fischeri* presented negative results for both sampling periods. However, the *Lytechinus variegatus* tests indicated chronic toxic effect on the embryolarval development in the sample of station 6 (at Largo da Pompeba) for the winter sample. In summer, samples from stations 2, 3 and 4 were also considered toxic. The sample from station 3 (Largo Santa Rita) showed the lowest embryolarval development rate, about 7%. Samples from stations 2 and 4 presented rates of approximately 60 and 70%, respectively (Figure 2). The larval development in winter samples was generally higher than those observed in summer tests.

4 CONCLUSIONS

The comparison of these recent data with those of thirty years ago clearly shows the escalating of nutrient input into the area, despite the environmental programs applied, increasing scrutiny and reduction of the legal limit of contaminants discharge in industrial effluents. The verified increase in concentrations was due to population growth in the metropolitan Santos area beyond the limit of the support capacity of the estuarine system, revealing both the enormous pressure to which the area has been submitted over the last 3-4 decades and the necessity of integrated management of the area.

The huge loads of nutrient and seston received by Santos and São Vicente Estuaries significantly impact the phytoplankton biomass and productivity and also the balance between autotrophic and heterotrophic processes. Nutrients are not limiting phytoplankton growth, and the main forcing agents related to phytoplankton productivity were temperature and light. Water column depth and dynamic stability seem to be the main forces causing the differences between São Vicente and Santos Channels concerning phytoplankton biomass and production rates.

The biomass concentration allows for characterizing the inner region as eutrophic (based on Smith et al. 1999). The high percentage of active Chl-a and the 480/665 ratio indicated good physiological conditions of the phytoplankton cells. DO concentrations in the system were never under the minimum legally accepted limit, due to the high phytoplankton productivity and prevailing mixed conditions of the environment, despite the strong heterotrophic activity indicated by CO_2/O_2 ratios.

Ecotoxicity tests with water samples did not indicate acute toxicity for the entire area. Nevertheless, the results of chronic toxicity tests revealed critical areas in the estuary. The results observed in terms of chronic toxicity provide significant insight into the ecological impact on the area, even though the advective processes in the water column tend to dilute the toxic agents.

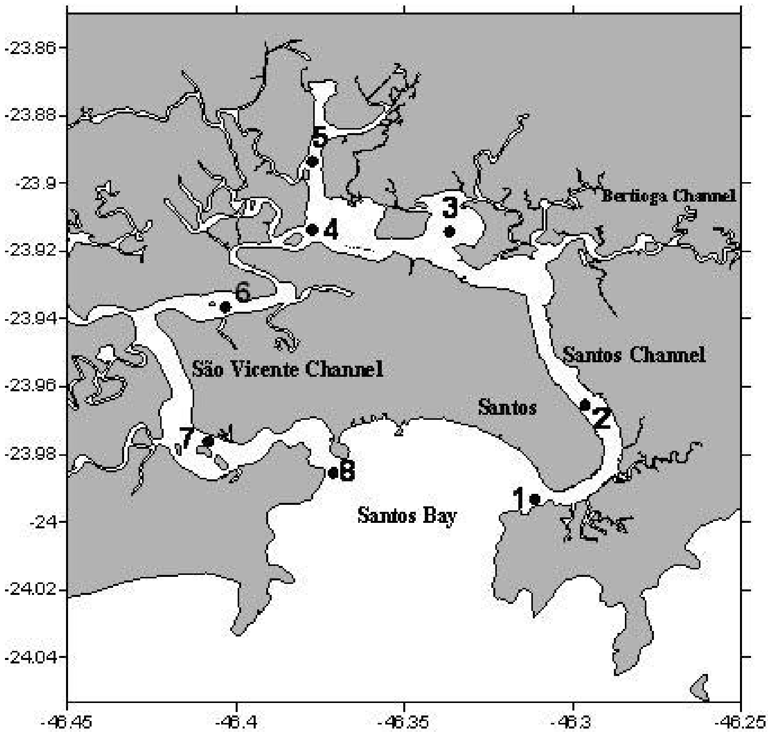

FIGURE 1: *Map of the Santos estuarine system (Southeast Brazil) and the location of the 8 sampling stations visited in August 2005 and March 2006.*

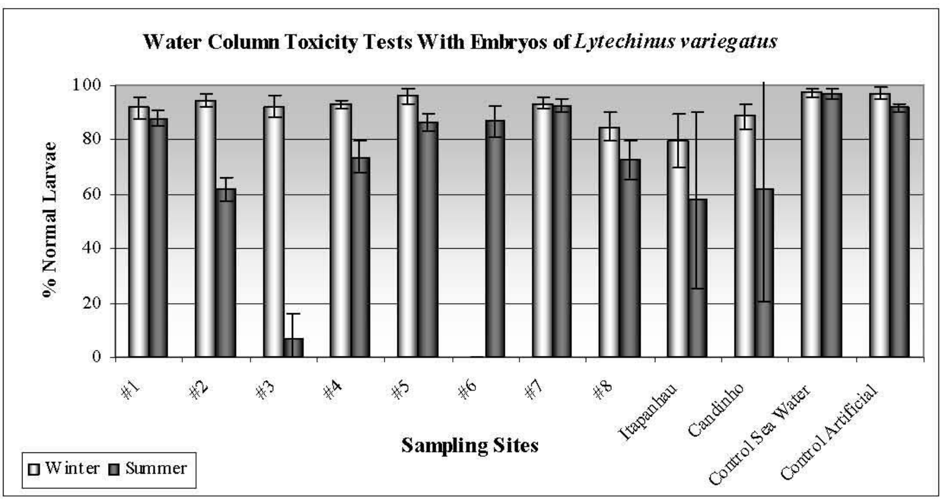

FIGURE 2: *Medium percentage of normal larvae and standard deviation obtained at the test with water column samples collected in winter and summer.*

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SEDIMENT QUALITY OF THE SANTOS ESTUARINE SYSTEM

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1 INTRODUCTION

Besides water column conditions already described in Chapter Ecological status of the Santos Estuary water column, especial attention should be given to the estuarine sediment quality, considering that sediments accumulate contaminants in concentrations higher than those observed in the water column, producing harmful effects on the benthic biota and the organisms that feed on the benthos or the sediment. Due to benthos' ecological importance and the persistence of pollutants in this environmental compartment, the sediment is adequate for environmental evaluations (Swartz et al. 1982).

Santos Estuary is regarded as a highly disturbed and polluted environment. The high inhabitant density and the intense industrial and port activities have been induced environmental perturbations in the area. The physical and chemical characteristics of the sediment in this region have been extensively studied (Boldrini and Navas Pereira 1987, Bonetti 2000, CETESB 1981, 1985, Fulfaro et al. 1983, Lamparelli et al. 2001, Medeiros 2000, Montone 1987, Prosperi et al. 1998, Tommasi 1979, 1985, Weber 1981) and pollutants in concentrations able to produce biological adverse effects have been observed.

Ecotoxicological studies were recently applied as another tool for the estuary's environmental assessment. Such approach includes biological information about the effects of xenobiotics in aquatic organisms (Abessa et al. 1998, 2001, Cesar et al. 2006, Rachid 2002, Sousa et al. 2007) and a new advance has focused on the "weight of evidence" (Chapman 1992), integrating physical, chemical and biological data to perform a synoptic view of the environmental quality of this estuary (Abessa 2002, Abessa et al. 2005, Cesar et al. 2007). The aim of this study was to verify the sediment quality in the Santos Estuarine System affected by different sources of contamination through ecotoxicological assays.

2 MATERIALS AND METHODS

The sampling area (Figure 1) is comprised on Santos Bay and Estuary. The east side of the estuary comprises stations 1, 2, 3 and 5. That area receives high influence of domestic waste, industrial effluents and dredging interference from port activity. The west side of the estuary is influenced particularly by domestic waste from slums at the border of the São Vicente Channel (stations 6, 7 and 8). Reference samples were collected from Ilhabela City, situated in São Sebastião Island, São Paulo (23º48' S, 45º23' W) and in Itapanhã and Candinho, both situated on the Bertioga Channel. Ecotoxicological analyses consisted of toxicity tests with water-sediment interface, elutriate, porewater and whole sediment, according to Table 1.

Toxicity of the sediment-water interface was determined following the procedures described by Cesar et al. (2004) and in accordance with accepted guidelines (ABNT 2006b, CETESB 1999, Environment Canada 1992, USEPA 1995, 2002).

Elutriate can be defined as a liquid phase extracted from fine particles, obtained after re-suspension of sediments in water (Oxford Reference 1990). Toxicity tests with elutriates were developed specifically to evaluate the effects caused by disposal of dredged material on water quality, simulating, in short-scale, the transference of the toxic capacity from sediments to water as a consequence of such disposal. Elutriate samples were prepared according to Lamberson et al. (1992).

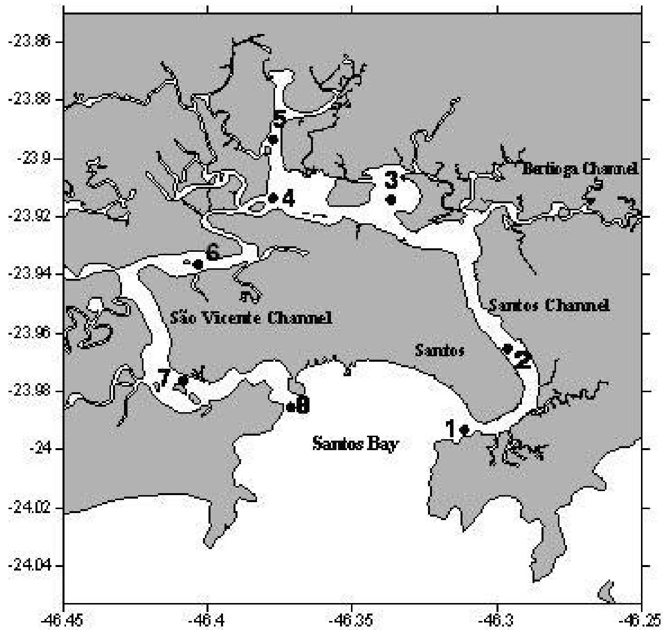

FIGURE 1: *Map of the Santos Estuarine System (Southeastern Brazil) and the location of the 8 sampling stations in the visited area in August 2005 and March 2006.*

TABLE 1: *Ecotoxicological tests.*

Sample	Endpoint	Organism
Water –sediment Interface	Embryolarval development	*Lytechinus variegatus*
Elutriate	Embryolarval development	*Lytechinus variegatus*
Porewater	Embryolarval development Offspring	*Lytechinus variegates* *Nitocra* sp
Whole sediment	Survival Offspring	*Tiburonella viscana* *Nitocra* sp

Porewater is the liquid phase of the sediment and was extracted by suction using the methodology proposed by Winger and Lasier (1991). The following porewater dilutions were prepared in filtered seawater: 25%, 50% and 100%. For liquid samples (water-sediment interface, elutriate and porewater), an embryolarval development test with *Lytechinus variegatus* following ABNT (2006b) methodology was used.

According to this methodology, chronic toxicity tests were conducted for 24hours and, after this period, the percentage of normal pluteus larvae was evaluated. *Nitocra* sp was also used to test porewater toxicity through the methodology described by Lotufo and Abessa (2002). A static 96 hours test was conducted. The chronic effect was measured through reproduction rate, considering the amount of live offspring (nauplii and copepodites) in each concentration.

Toxicity of whole sediment was assessed using the method described by Swartz et al. (1985) and adapted by Melo and Abessa (2002) for *Tiburonella viscana*. A static acute 10-day test was conducted and the mortality rate was evaluated. The chronic test with whole sediment was performed with *Nitocra* sp as well during 96 hours. The offspring rates were evaluated (Lotufo and Abessa 2002). All the data was analyzed by one-way analysis of variance (ANOVA) followed by Dunnett's test (ASTM 1992, Swartz et al. 1985, USEPA 1991). The results were compared statistically using the MSD (Minimum Significant Difference) method and analysis of variance followed by the Dunnet's test for detection of significant differences and determination of the toxic samples.

3 RESULTS AND DISCUSSION

3.1 Interface water-sediment *L. variegatus*

In relation to interface water-sediment tests conducted in the winter, only the samples of stations 7 and 8 (situated at Garganta do Diabo and Portinho) showed toxic effect to the embryolarval development of *Lytechinus variegatus*. During the summer, the stations considered toxic were 1, 2, 3, 4, 5, 6, 7 and 8. During this period the stations 1, 2, 6, 7 and 8 exhibited the low development of *L. variegatus* embryos, as observed in Figure 2.

It is possible to observe that interface water-sediment samples, collected in summer, presented lower development than samples collected in winter. This pattern is not observed at stations 5 and 8.

3.2 Elutriate *L. variegatus*

Taking into account the low development of *L. variegatus* embryos exposed to elutriate, all samples collected in summer showed adverse effects at the concentrations 100%, while in winter the toxic samples were 2, 4, in Itapanhau and Candinho. The results are presented in Figure 3.

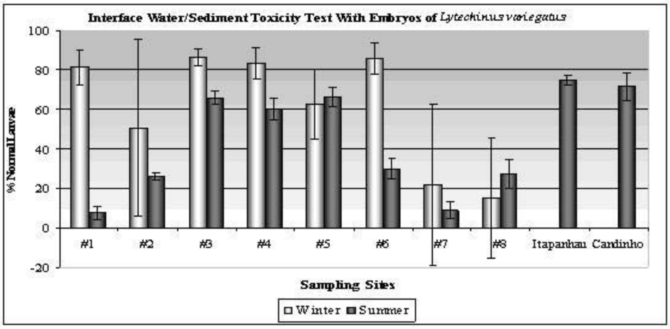

FIGURE 2: *Medium percentages of normal larvae and standard deviation obtained from the test with interface water/sediment samples collected in winter and summer.*

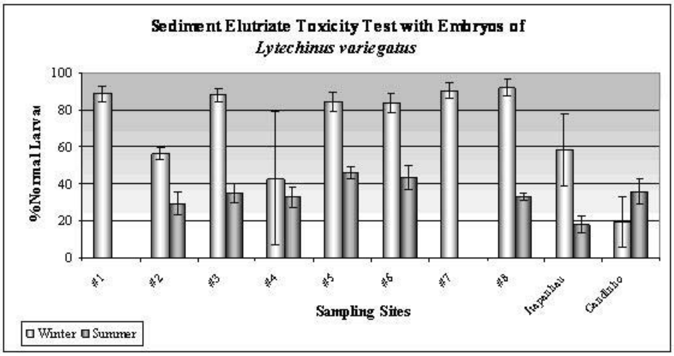

FIGURE 3: *Average percentages of normal larvae and standard deviation obtained from the test with elutriate samples collected in winter and summer.*

3.3 Porewater *L. variegatus*

Considering the low development of *L. variegatus* embryos, the samples collected in winter that exhibited adverse effects at the concentrations 100% were: 1, 2, 3, 5, 7, Itapanhau and Candinho. During the summer, the samples considered toxic were the ones from stations 2, 3, 5, 6, 7, 8, Itapanhau and Candinho. The results are presented in Figures 4 and 5.

3.4 Porewater *Nitocra* sp

Considering *Nitocra* sp offspring rates, the sample of station 5 collected in winter exhibited reduced offspring rate, as observed in Figure 6. In summer, no significant difference was observed. For offspring rates, the sample from station 7 exhibited an offspring rate more reduced than other samples, as observed in Figure 6. It is possible to observe that porewater samples collected in summer presented offspring rates lower than samples collected in winter, but this pattern is not observed at stations 1 and 5.

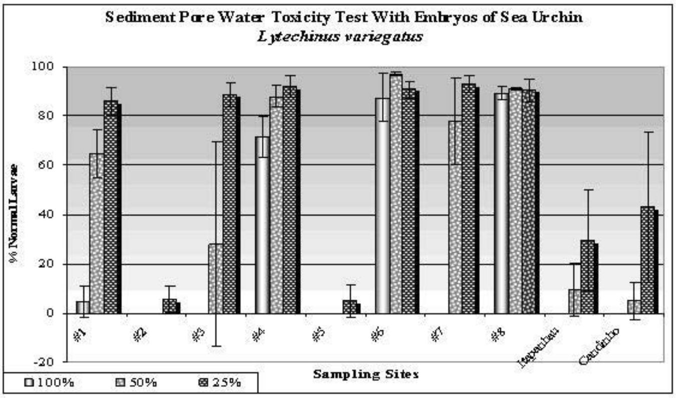

FIGURE 4: *Average percentages of normal larvae and standard deviation obtained from the test with porewater samples collected in winter.*

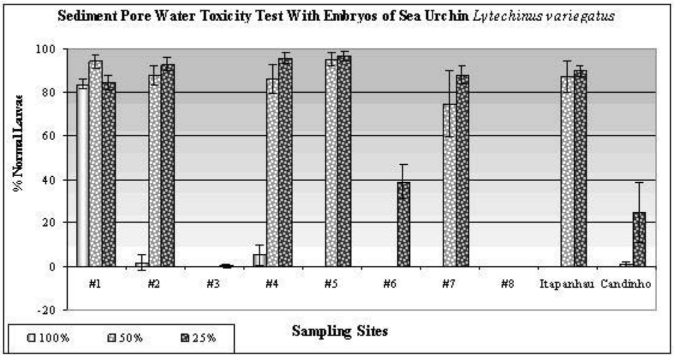

FIGURE 5: *Average percentages of normal larvae and standard deviation obtained from the test with porewater samples collected in summer.*

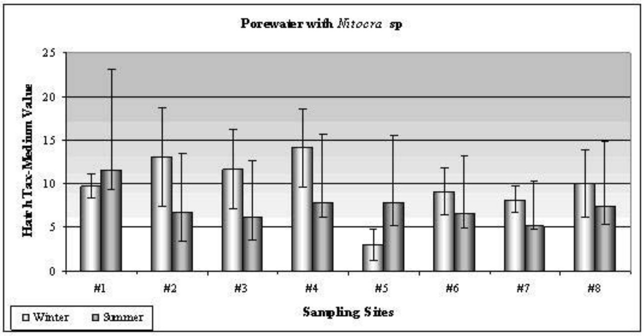

FIGURE 6: *Medium tax of offspring and respective pattern deviations obtained from the test with porewater samplesin winter and summer.*

3.5 Whole sediment *Tiburonella viscana*

The acute test with the amphipod *T. viscana* exhibited toxicity at station 7 and reduced survival at station 4, collected in winter, and stations 3 and 5, collected in summer. The results obtained in both seasons are presented in Figure 7. According to these results, the survival percentage of adults amphipod in relation to the sediment evaluated was higher in samples collected in winter than samples collected in summer.

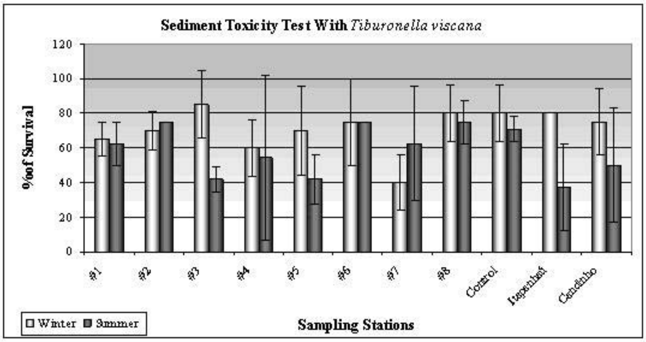

FIGURE 7: *Medium survivor for amphipod* T. viscana *and standard deviation in sampling stations collected in winter and summer.*

3.6 Whole sediment *Nitocra sp*

The chronic test with *Nitocra* sp exhibited reduced offspring rates in samples of stations 2, 3 and 6 in relation to other samples collected in winter, although no significant difference was observed between samples collected in summer. The results obtained in this test are presented in Figure 8. Regarding the adverse effects presented for *Nitocra* sp test, it is possible to verify toxicity in all stations, with variations on the exposure routes, end points and organisms tested.

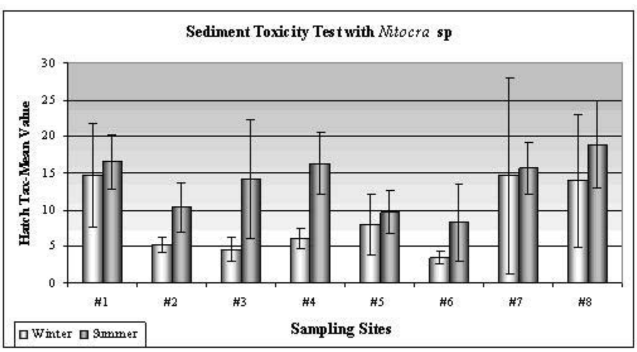

FIGURE 8: *Medium taxes of offspring and respective standard deviations obtained in whole sediment test, winter and summer.*

3.7 Toxicity frequencies

Figure 9 shows the frequency of each studied area. Toxicity was more often verified at stations 2, 3, and 5. These stations are situated near the container terminal at Porto Channel; at Largo Santa Rita, between Bagre and Barnabé Islands and COSIPA Channel, respectively (Figure 9). According to Abessa (2002), higher concentrations of Cd, Hg and Zn were detected in samples of sediment from station in region 3. The region of station 2, according to Nishigima 2001, presents PAHs concentrations between 1.38 and 2.91 μg g^{-1} and some aromatic compounds presenting values higher than PEL (Probably Effect Level), such as Criseno, Fluoranteno and Pireno.

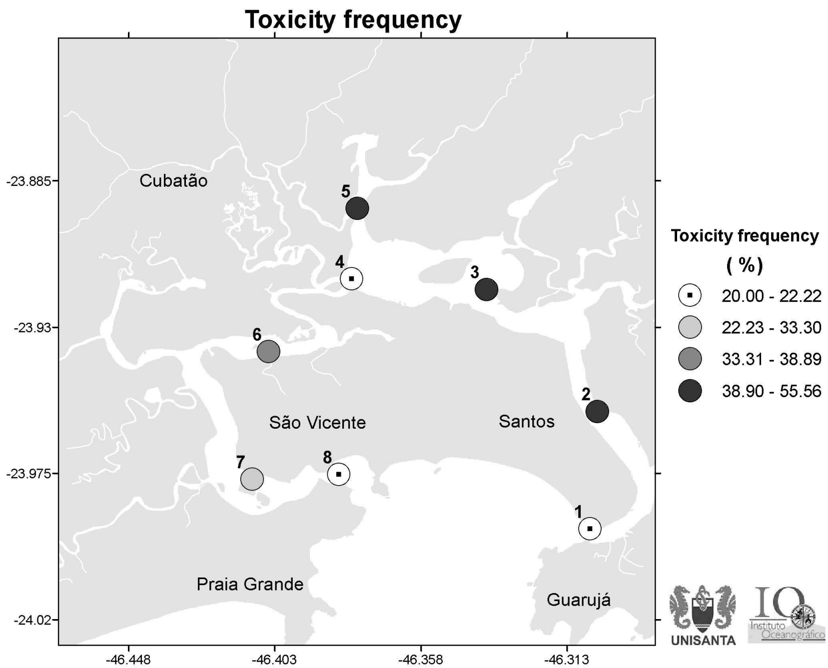

FIGURE 9: *Toxicity frequencies in samples from studied areas.*

According to Lamparelli et al. 2001, in the region of station 3, Largo de Santa Rita, higher levels of Hg were detected in samples of sediment and the total PCBs incorporated to fish were above the established limit for human consumption. The regions of stations 4 and 5 are areas influenced by industrial discharges, and are highly contaminated by PAHs and metals (Cesar et al. 2006). Station 6, located in Largo da Pompeba, presented moderate toxicity frequency. However, the chemical contaminant level, except for Cd, is not high. However, according to Lamparelli et al. 2001, this area receives a significant amount of drainage water contaminated by sewage, which could explain the higher frequency of toxicity.

According to Medeiros (2000), Medeiros and Bicego (2004), who quantified biochemical indicators at sewage in estuary and Santos bay, the Largo do Pompeba region presents LABs

(Linear aquilbenzeno) values similar to the values observed at the end of the outfall in Santos Bay. This value was attributed to the introduction of domestic sewage *in natura* from slum and wood stacks at the margin of the entire region and from the garbage deposit of Alemoa. Although that compounds might not be to blame, many others could be responsible for the toxicity found in that region. The samples from stations 1, 7 and 8 in external areas of the estuarine system present lower frequency of toxicity and, according to Abessa and Sousa (2001), these areas presented moderate or inexistent levels of degradation.

Our results were in accordance to Abessa (2002), Cesar et al. (2006) and Sousa et al. (2007) that considered the superior portion of the estuarine system critical. In the present study, the area with higher toxicity was found at stations 2, 3 and 5. Those areas were considered degraded while stations 6 and 7 presented moderate degradation. The seasonal variation on toxicity results could be explained in part by the terrestrial drainage runoff in summer, which caused the increasing uptake of contaminants. In the estuary, the salinity is another factor that controls the availability of contaminants between sediment, interstitial water and overlying water, and the time of balance between these compartments varies due to the type of sediment and the time of overlying water remain. The rise of salinity intensifies the removal of organic matter dissolved in the sediment, forming the particulate organic matter that probably intensifies the sorption of hydrophobic chemicals from water to sediment.

Metals like Cd and Zn could diminish the sorption process with the organic particulate matter in higher salinities, increasing the availability in the water phase (Chapman and Wang 2001). The distribution of chemicals in sediment can also be related to the type of grain. Local hydrodynamics conducts the distribution of superficial particles in sediment, and, therefore, the distribution of contaminants and the bioavailability of them inside the estuary.

4 CONCLUSIONS

The higher incidence of toxicity was observed in the tests carried out with sediment samples from Santos and Piaçaguera Channels due to the accumulated contaminants in concentrations higher than those observed in the water column, which produced negative acute and chronic effects as mortality and reduced offspring.

Considering the previous pollution in this compartment and the variations on terrestrial drainage runoff, industrial and domestic sewage inputs, hydrodynamics and dredged process, the bioavailability and effects of contaminants in the sediment of this estuary could be considered a dynamic process, that should be monitored constantly as part of an integrated costal zone management.

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SOCIO-ECONOMIC ISSUES IN THE SANTOS ESTUARY

A.F.P. SAMPAIO AND J.M.S. FERREIRA

1 INTRODUCTION

This chapter presents the main aspects of the socioeconomic issues in the Santos and São Vicente estuarine systems which have been addressed in the Ecomanage project. In close cooperation with the stakeholders two socioeconomic issues were selected for implementation with the analysis methodologies developed in this project. In light of the fact that high population density and "urban sprawl" in metropolitan areas are usually the consequence of numerous drivers, this work sought to identify indicators that could mirror the basis of these metropolitan socio-economic conditions and identify the pressures associated with them. Thus, information was sought on population growth rate and density, economic activities in the region and the characteristics of its sanitation infrastructure for water and sewage.

2 STUDY REGION

The study area encompasses five municipalities that comprise the Baixada Santista metropolitan region, formed by nine coastal municipalities, whose socioeconomic activities take place in the Santos and São Vicente estuarine systems (Figure 1). The municipalities of Santos, São Vicente, Cubatão, Praia Grande and Guarujá all fall in the study area; the two latter municipalities have only part of their territory in this drainage basin. At the continental rise of Serra do Mar, the cities of Santo André and São Bernardo do Campo, two cities of the metropolitan region of São Paulo, also fall within this drainage basin. However, as this area lies in the Serra do Mar State Park, where any development activity is forbidden, these cities were excluded from the socioeconomic analyses presented here.

3 METHODOLOGY

A wide-ranging literature search was carried out and an analysis of secondary data from census surveys was completed by an extensive information and data gathering campaign with the municipal administrations of all five municipalities. All the information obtained was converted into a GIS format at a scale of 1:50,000. A year-long preliminary study comprised successive field visits, along with participation in countless regional and state discussion fora, such as environmental councils, public hearings, regional and health development seminars and participation in meetings with representatives of the various sectors active in the region, among them industry representatives, hydrographic basin committees, port management, Agenda 21, coastal management, among others. This work helped in the identification of stakeholders and in the holding of consultative meetings by the project's staff with these stakeholders to disseminate the DPSIR approach. These meetings, described by Belchior et al. (this volume) from the first turned out to be effective for making an informed choice from the many socioeconomic issues existing in the region and in obtaining clear answers from the stakeholders concerning their management needs and -weaknesses in tackling these issues. This approach provided the project with the much needed support to focus on two important socioeconomic drivers associated with the lack of urban infrastructure which were chosen for the implementation of the Ecomanage methodologies of integrated coastal analysis and the management tools developed in the project.

The second step was to perform a prospective analysis of socio-economic development for these two drivers based on historical trends and on an assessment of the current trend in the private as well as in the public sector. To that effect, demographic projections were made for the five cities. Also an analysis was carried out of the main projects foreseen by private and governmental organisations.

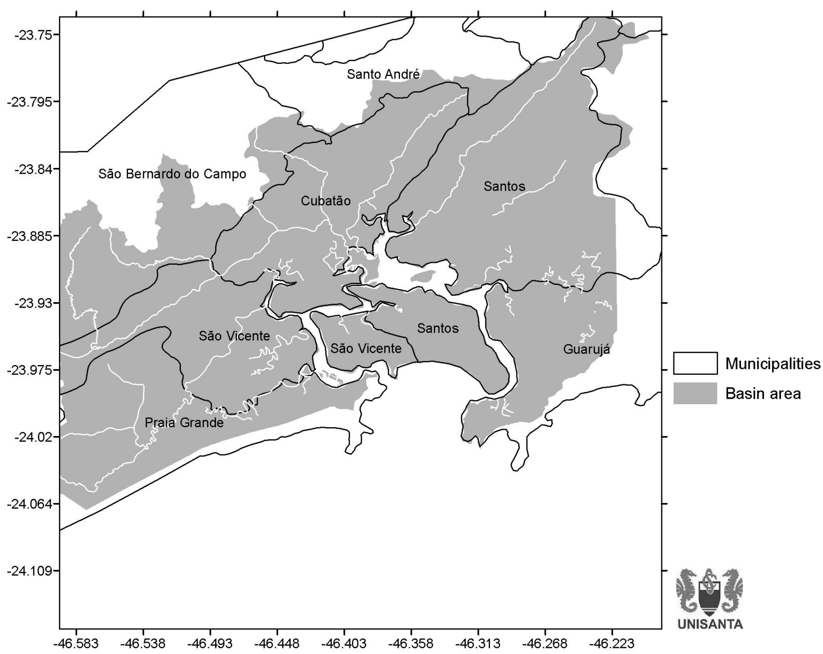

FIGURE 1: *The Santos study area.*

4 SOCIO-ECONOMIC DRIVERS OF CHANGE

Due to the presence of a large industrial agglomeration around the largest Brazilian port and the existing high rate of urbanization, there is a huge variety of issues with environmental aspects that could be studied in the region; however, the preliminary studies and the existing secondary data, reinforced by the needs of the stakeholders, expressed during the forums in the second year of the project, indicated two high-priority study subjects: (1) **Housing issues** related to the existence of a large number of illegal and ramshackle dwellings on river banks and along estuarine channels, and (2) **sanitation issues** related to the diffuse discharge of domestic sewage.

In order to correctly forecast the development of these problem areas, other drivers of change such as population growth and urban, industrial and port development were taken into account so that local actors could eventually be provided with comprehensive information on the future development of these scenarios. The following sections describe the history of urban development in the region up to the present, the current demographic profile as population growth rate, demographic density, urban infrastructure and prospective analysis of the selected scenarios.

4.1 Historic evolution of urban development in Santos

Marked socioeconomic development of the region started in the late nineteenth century, driven by the growth of the coffee economy linked to the Port of Santos. The expansion of the coffee trade - in the early twentieth century - made the Port of Santos the largest port exporting this product. Thus, the city of Santos had an important role as broker of this operation's financial and commercial activities. This greatly stimulated the urbanization and development of Santos early in the twentieth century and the city needed improvements in infrastructure. Immigration at this time was important too, because a good number of the immigrants that arrived in the country via the Port of Santos settled there, enhancing population growth (Figure 2), but also worsening the environmental conditions on account of the scarcity of sanitation infrastructure and -services (Silveira 1994).

The historical landmark of sanitary investment at that time was Saturnino de Brito's project with the construction of drainage channels all over the insular area of Santos, started in 1910. This system, still in working condition today, was projected to cater for a resident population of 150,000 inhabitants (AGEM 2001), that suffered from the lack of basic sanitation as well as from the health and death problems associated with it.

In 1947, with the construction of a highway connecting the coast with the highlands of São Paulo, followed by the implementation of the car industry in the sixties, Santos started being strongly pressed by the leisure and tourism industry. This momentum was a change for the entire region, making it a tourist center for the capital and the whole state of São Paulo. The tourism industry accounted for changing dramatically - and in a very short period of time - the whole spatial and urban configuration of the region. The city started experiencing increasing demands for vacation housing, services and associated commerce. This occurred with such intensity in Santos that in the late sixties the city lived through urban saturation phenomena that would make for a change in the demographic, economic and social patterns in the seventies (AGEM/EMPLASA 2002). At that time the degree of urbanization of Santos was already 95% (Young and Fusco 2006) and its population over 260,000 people (Jakob 2003).

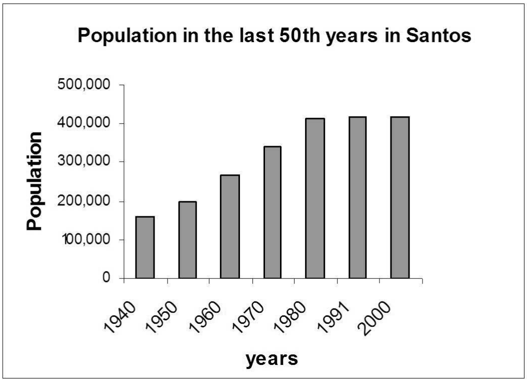

FIGURE 2: *Population growth in Santos (1940 to 2000).*

Simultaneously, between 1953 and 1954, the city of Cubatão, the only city in the region not having beaches and therefore not counting on the benefits of the tourism industry, started attracting industry thanks to the proximity of the estuary, of the Port of Santos but mainly thanks to the abundance of water and electricity provided by the Henry Borden hydroelectric plant opened in 1933 and expanded in 1955. With the decision of building the Presidente Bernardes refinery, the supply of raw materials produced by that company induced the settlement of a series of other industries, making the region a strongly integrated petrochemical complex, followed later on by the construction of Companhia Siderúrgica Paulista (Cosipa) and industries of the chemical sector, including fertilizers. Thus, the development and the expansion of industrial activities accounted for economic growth and increasing job supply, attracting a large number of migrants in search of job opportunities from other regions of the country, chiefly Bahia, Minas Gerais and São Paulo's countryside. Therefore, Cubatão's urban population which was only 1,887 inhabitants strong in 1940, grew to 78,327 inhabitants in 1980. However, after some time, the serious effects of air and water pollution by these heavy industries, unencumbered by pollution control policies at the time - worsened by the topographical and climate characteristics of Cubatão - turned this scenario of economic development into one of the most serious environmental problems of the Southern Hemisphere. Cubatão came to be regarded as the most polluted city in the world in the eighties.

From 1950 - with the beginning of industrialization in Cubatão and the increase of tourism, made possible by the construction of the Anchieta highway - to the eighties, population growth strongly changed the region. Housing went high-rise, following in the footsteps of the demographic growth in the insular part of Santos and São Vicente and consequently the saturation of available areas, causing a great urban agglomeration to emerge, mainly in the up-market areas close by the sea. Due to the city of Santos' physical limits in its insular portion, to the high costs close to the beach avenue and the weakness of the land on the city's hills, the low-income population migrated to the peripheral zones of the city and to the neighboring municipalities (Jakob 2004). This characteristic of the urban consolidation of Santos, driven by the scarcity and consequent increase in the cost of land, made young families look elsewhere for places more within their means. This natural flow, characteristic of many big cities in development produced three migratory vectors surging from Santos, still going on today.

The first is toward insular and continental São Vicente and its closest neighbor, Praia Grande, the second is toward Cubatão and the third toward the district of Vicente de Carvalho in Guarujá. These migratory vectors were identified by Jakob (2003) as having occurred since 1980. However, the urban expansion in the region was also accompanied by the development of countless low-grade (slums) and clandestine lot division. The places chosen for this kind of occupation were the environmentally weakest areas such as mangrove areas and river banks, difficult to access and completely lacking infrastructure. The growth of these clandestine settlements occurred in a haphazard way and worsened even more in the following decades, during periods of economic recession and unemployment that occurred all over the Baixada Santista metropolitan region.

From the second half of the seventies a new process of expanding demand for both port and port-associated services - concentrated in Baixada Santista - was stimulated by the economic expansion occurring in the state of São Paulo, making the Port of Santos undergo deep changes and become the largest port for commercial cargo in the country. However, into the eighties, despite the improvement in road access with the construction of a new highway, industrial development lagged behind. The negative economic impacts of the crisis and the economic policy of that period caused a remarkable reduction in economic growth. The high degree of environmental pollution - forcing environmental protection agencies to tighten control over the industries - made the petrochemical complex lose new investments, (shifted to other parts of the country) despite its excellent location. These events directly affected the whole of the region, causing unemployment and underemployment with strong impacts in the regional urban space (AGEM/EMPLASA 2002).

The conjunction of all of these factors at the time made the cities suffer a tremendous process of environmental degradation and loss of quality of life owing to industrial and domestic pollution, the latter caused by the great urban agglomeration in the insular area and the appearance of several low-grade settlements in the peripheral zones. Consequently, a great number of tourists went on to look for other coastal regions as holiday destinations. This forced the state government and the city of Santos to start an improvement program aimed at recovering the tourist potential that the city was losing. Following local political pressure, a state sanitation program was started in the peripheral zones that host the largest number of low-grade settlements; however this action was not enough to serve all of the existing low-income nuclei that were settled between the seventies and nineties (Young and Fusco 2006), and that still subsist today.

Studies carried out by Jakob (2003) show that currently the region goes through a period of great change in its population dynamics, marked by a deceleration in the growth rate and by the emergence of an urban metropolitan structure, in a phase of consolidation on the insular portion of the cities of Santos and São Vicente (Young and Fusco 2006). These trends however can be slightly altered or accelerated since there is evidence of a resurgence in economic growth, both country-wide and regional, marked by significant investments in real estate, harbour development and industry, all known to be urban growth and development fosterers. However, it is also extremely difficult to predict with any degree of certainty what the impacts of this new age of investments will be. Therefore, what has been done was to make some considerations and inferences based upon historical trends that have affected this region.

From this point of view, based on the process of occupation and development marked by the insular area's urban consolidation, peripherization of the region and this new economic momentum, qualitative and quantitative analyses of the agents, current and future socio-economic pressures on the environment were carried out. Besides, a drive to design prospective scenarios on regional development took place including demographic projections for the five cities, along with an analysis of public and private projects related to existing low-grade settlements and the expansion of basic sanitation services in the region. Both analyses were performed for a time horizon up to 2015, the year in which some of these cities prepare to present the results of the actions towards reaching the goals set out in the ”Millennium Development Goals”, a UN program for sustainable development of cities (SEADE 2005).

4.2 Demographic Profile

Among the five towns of the estuary region, Santos is the town with the largest number of inhabitants and with the largest territorial area, followed by São Vicente, that holds the greatest population density per square kilometer, according to the last Census in 2000 (Fig.3). These five cities, according to the demographic census, house 72% of total about 1,500,000 people from the entire Baixada Santista metropolitan region and all of them have over 100,000 and fewer than 500,000 inhabitants (Table 1).

The fact that these cities are completely urbanized, only separated by natural barriers, such as rivers, hills, mangrove areas and estuarine channels indicates that the insular area of Santos and São Vicente, characterized by high demographic density (Figure 3) due to the huge existing population agglomeration, cannot grow significantly any more. Nevertheless, in the case of São Vicente, while in the insular part the population only went from 221,296 inhabitants in 1991 to 225,873 inhabitants in 2000, in the continental area, the population had an extraordinary growth, jumping from 47,322 inhabitants in 1991 to 77,678 inhabitants in 2000, thus growing at a 5.66% annual rate. This rate far surpasses the region's average growth rate, explaining the great expansion observed in the urban area of Vicente, located on the continental side and showing the region's heterogeneity.

4.3 Slum-housing

A great number of illegal low-grade human settlements (slums) exist in the region. There is a significant portion of people living on palafittes and many of them have settled on the river banks and in extensive mangrove areas (PRIMAHD 2005, IBGE 2000) of the coast plain without any control from the public office. According to census data (IBGE 2000) along with the most recent municipal surveys, 20% ofthe population in the Santos - São Vicente estuary drainage basin lives in illegal housing (Table 2). Most of these illegal human settlements have over 1,000 inhabitants (Figure 4). In the city of Santos, 60% of low-grade settlements have this characteristic and in São Vicente, 100% can be found like this (Young and Fusco 2006).

Due to the fact that they are illegal, and thus officially do not exist these settlements have no proper water and sewerage servicing and due to the fact that they are found in mangrove areas and river banks, considerable volumes of domestic effluents are being drained directly into the estuary (Table 3). Thus, the *in-natura* dumping of this domestic sewage into channels, rivers and the sea of the region is a (potential) source of estuarine pollution (CETESB 2001) and consequently very likely of health problems for people, especially for children from 1 to 10 years old who use the estuarine water for leisure and recreation (Figure 5).

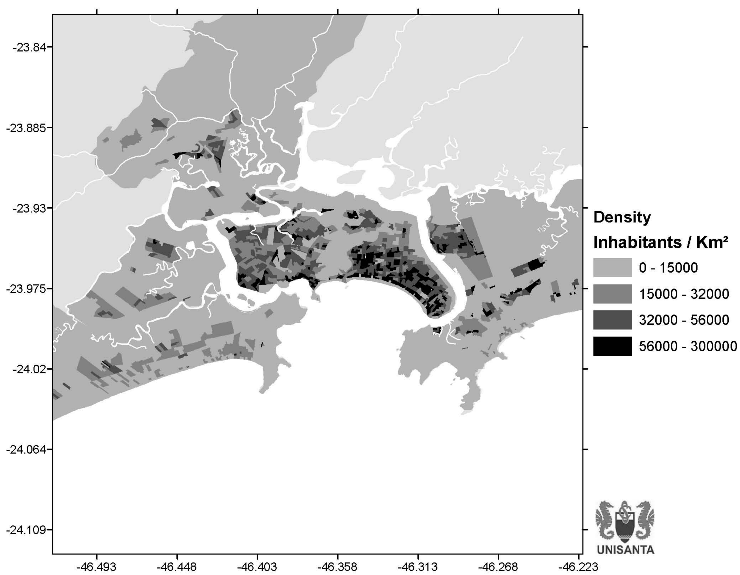

FIGURE 3: *Demographic density of Santos - São Vicente estuary (Source: IBGE 2000)*

4.4 Health

Child mortality is one of the most used indices to evaluate the health conditions of a population. The five towns lowered their child mortality rate between 1995 and 2000 (Figure 6); however, all the towns with the exception of Santos had higher child mortality rates than the average rate in the whole state of São Paulo (RMBS 2007). It was also observed that the child mortality rates of the five cities showed large fluctuations which deserve special attention in the treatment of this issue. Although the figures fluctuate without showing a clear trend, they do show that the situation has worsened visibly in Santos and São Vicente in CETEC/CETAM (2000). Adding to this all, the results of microbiological water analyses carried out twice a year by CETESB corroborate a contamination scenario, since the quantity of coliforms found in these samples has not only exceeded the maximum limits but has also progressively increased in recent years, making the prevalence of domestic sewage discharges evident (CETESB 2006, 2005). Besides, two analyses of colimetric data performed all over the estuary in 2006/2007, summer and winter have also confirmed this type of estuarine contamination.

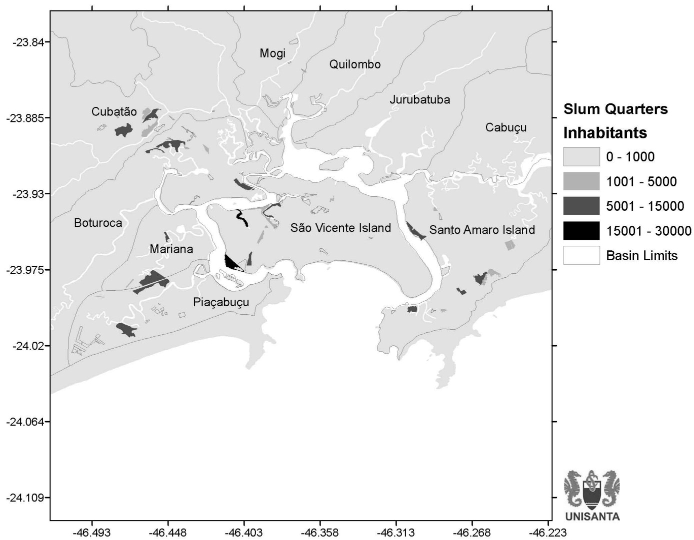

FIGURE 4: *Inhabitants of illegal settlements slums/low-grade dwellings. Source: municipal data.*

FIGURE 5: *Children playing in the Santos estuary channel.*

The abundance of faecal coliforms indicates that the estuarine waters used for leisure by a part of the population may also be contaminated by pathogens responsible for the transmission of (infectious) diseases, putting this population's health at risk. Likewise, no data from epidemiologic research in the region was found that was associated to the current scenario of microbiological contamination encountered. Consequently, it is necessary to prioritize future works of this nature to identify what the impact of this increasing deterioration of estuarine water quality is on the health of these populations that frequently use these waters for leisure and may therefore run the risk of contracting diseases.

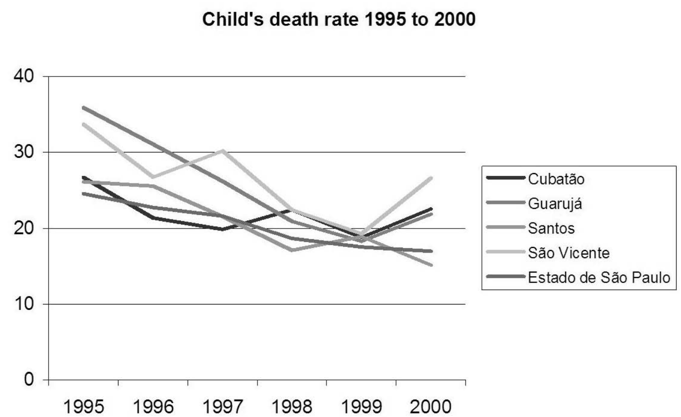

FIGURE 6: *Child mortality rate (per 1,000 born alive) from 1995 to 2000.*

TABLE 1: *Population by city (Source: IBGE 2000).*

Town	Population
Santos	417,983
São Vicente	303,551
Guarujá	264,812
Praia Grande	193,582
Cubatão	108,309
Total	**1,087,273**

TABLE 2: *People in illegal housing (slum). Source: municipal data.*

City	Inhabitants
Cubatão	52,863
Santos	30,724
São Vicente	86,909
Praia Grande	14,592
Guarujá	43,869
Total	**228.957**

TABLE 3: *People serviced by sewerage services (Source: IBGE 2000 and SABESP)*

SANITARY CHARACTERISTICS	POPULATION
NOT TREATED	457,067
TREATED (ETE)	81,451
TREATED (SANTOS OUTFALL)	520,688
TREATED (PG AND GJÁ OUTFALL)	28,037
TOTAL	**1,087,273**

4.5 Sanitary Infrastructure

As the fast process of urban expansion in these cities was not followed by the supply of basic sanitation service, there is nowadays a great deficit of sewerage collecting network (Figure 7). Through the crossing of data from the last demographic census (IBGE 2000) and from the Companhia de Saneamento Básico do Estado de São Paulo - SABESP it was possible to estimate quantitatively and with more precision the number of people yearly attended and not attended by domestic effluent drainage and treatment services. Thus, except for the city of Santos, that now has 98% serviced (CETEC/CETAM 2000), from the total population of the five cities, 42% has no sewerage collecting network. If we consider the submarine outfall system in the region (where there is no primary treatment) as a final effluent disposal system, this number will account for about only 8% of the population upon counting the number of people that have their effluents treated before dumping into the estuary. This data corroborate the scenario of water contamination by domestic sewage, which seems to be one of the major problems for the Santos Estuarine System. The subsequent hyper-eutrophication could cause great harm to the flora and fauna, besides being a permanent danger to public health, especially for those living alongside these estuarine waters. Bringing the Santos Estuarine System to good sanitary conditions should therefore be a priority for local management.

5 PROSPECTIVE ANALYSIS

Prospective analyses for 2010 and 2015 were carried out to define the scenarios for urban area expansion and for domestic effluent sanitation services. Demographic projections of the status-quo were made from the history of socioeconomic development over the last decades, also taking into account the possible influence on the urban scenarios from the implementation of large industrial and harbour projects as well as from urban infrastructure projects in this period.

In this sense, integration with the social actors was fundamental to learn about the currently existing efforts to safeguard the socioeconomic development of the region. Projects of basic sanitation service expansion, slum urbanization as well as those from the port and industrial zone were assessed. This analysis helped to associate the current demographic trends with those of economic development. The main projects that have already started or are in the planning stage were identified, even those which are still under public discussion/analysis. Large construction projects - as, for example, industrial plant expansion and real estate projects - were identified in the region and associated to demographic growth through the spontaneous stimulus of migratory phenomena, which are usually associated with such undertakings. Thus, this analysis aims to provide elements for some prognostics on the impact from these regional development trends on the future quality of hydric resources in the coastal environment.

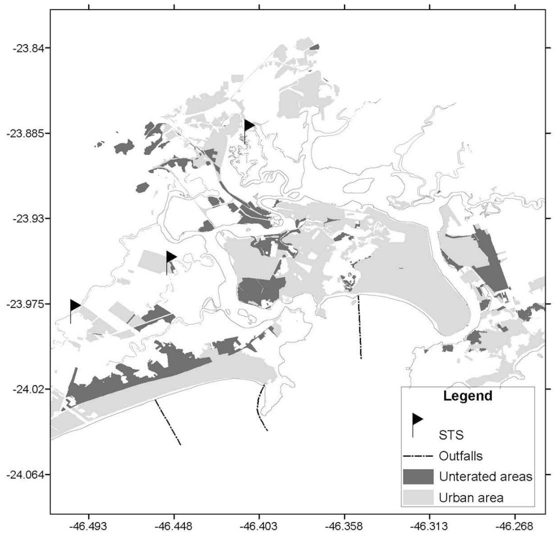

FIGURE 7: *Sewerage collecting network in the Santos area.*

5.1 Demographic Trends

Population growth (Table 4) in the region shows a decreasing trend with annual values below 2.0% and the process of urbanization driven by the intra and inter-metropolitan migratory flow also is slowing due to the urban consolidation of the central municipalities, Santos and São Vicente. Hence, we can formulate two hypotheses for the population evolution in the region:

- Decrease of population growth rate and relative control of migratory flows, higher population densities in the peripheral cities, owing to a more moderate expansion of the urban area and/or implementation of measures to restrict building on environmentally-protected areas as Serra do Mar slopes, mangrove areas and river banks.

- (Slight) increase of population growth rate due to an increase in migratory flows, followed by a process of increasing population in the peripheral cities and mainly in the continental area of Santos and São Vicente, owing to the lack of measures for controlling building in environmentally-protected areas, as exemplified by past events, stimulated by the ease of traffic access.

In the light of these hypotheses and of an infrastructure investment boom in the region fostered mostly by the growth of the national economy, some considerations could be formulated. Countless industrial and port expansion works - many of them underway now - have been altering the regional socioeconomic situation. Thus, it was found that the probability of the second hypothesis coming true is much higher than that of the first. As such, it is recommended to implement the measures proposed in the Coastal Management Program and its important management tool, the Economic Ecological Zone Division. It is also recommended to promote integrated solutions creating new urban designs for existing natural and urban areas (Afonso 2006), since these urban areas have irregularly expanded over natural areas. Finally, demographic data show that this problem is prone to occur more strongly in the continental areas of Santos and São Vicente, with São Vicente, due to the greater ease of road access and the existing urban infrastructure, being the most vulnerable.

TABLE 4: *Demographic projections for 2015.*

	Santos	**São Vicente**	**Cubatão**	**Praia Grande**	**Guarujá**	**total**
2000	415,922	303,413	110,378	105,950	151,580	1,087,243
2015	419,681	371,563	144,954	180,678	211,737	1,328,613
balance	3,759	73,120	34,576	74,728	60,157	**241,370**

5.2 Sanitary infrastructure

The 2010 and 2015 scenarios of the evolution of sanitary infrastructure (Figure 8) were projected based on the current and planned interventions of governmental projects and demographic estimates. The 2015 scenario considers a broad project of sanitary upgrading developed with the aim of eradicating the actual deficit in sewerage drainage and treatment, and improve the microbiological quality of the region's beaches. Health indices have been considered in this analysis. According to Sabesp - Companhia de Saneamento Básico do Estado de São Paulo - an increase in the treatment service is foreseen for the urban areas of the nine cities of Baixada Santista, among them, the study region, through the "Projeto de Recuperação da Qualidade Ambiental da Baixada Santista (Baixada Santista's Environmental Quality Recovery Project)" (DIARIO DO SENADO FEDERAL 2004).

The study areas of ECOMANAGE in the Santos estuary were assessed in order to estimate the percentage of the population with and without sewerage facilities, as well as the sewage treatment. However, it was found that in spite of the large investment, extension and importance of the project for the entire region of Baixada Santista, the population that irregularly occupies the banks of rivers and estuary channels is not part of the population that will be addressed in this program, but the part of the population that, while living in the urbanized areas, have no sewerage drainage network. Therefore, actions that could address these informal communities were undertaken. Thus, it was also possible to identify the urbanization processes foreseen for these illegal areas from information provided by governmental projects and programs. Conditions for the 2010 scenario were inferred from this data (Figure 8). From the population projections for the five municipalities and considering the implantation of environmental sanitation projects foreseen for the region, it was found that until 2010, the region will increase the percentage having access to sewerage drainage network service from 58% to 67% and for the 2015 scenario the level of servicing grows to be 89% of the population.

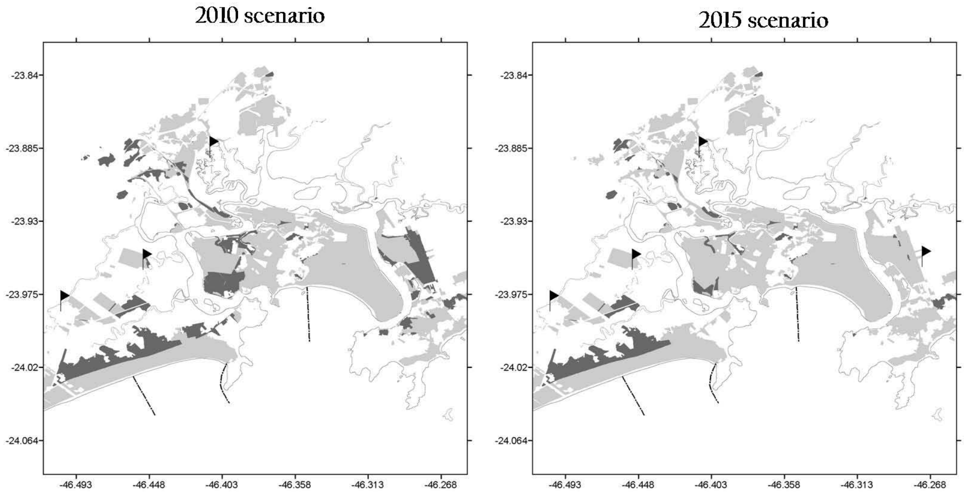

FIGURE 8: *Evolution of sanitary infrastructure scenario for 2010 and 2015.*

6 FINAL CONSIDERATIONS

Historically, the unevenness of interests created by the lack of public policies - that can set mechanisms to guarantee social welfare - associated to the region's economic development resulted in the loss of precious spaces and consequently of the opportunity of offering a significant portion of the population the possibility of integrating to urban spaces where urban service supply is established. Thus, the approach adopted in this study is justified for setting a knowledge base through spatial, demographic and sanitary characterization, capable of helping in the understanding about the estuary's contamination sources.

An existing deficit in sanitary service coverage is found, estimated in 42% of lack of sewerage drainage network service that includes the illegal houses and also the lack of drainage in regular urban areas. Therefore, considering the final realization of all interventions foreseen by governamental organs currently under way and demographic projections, the evolution of these scenarios was designed, projecting them for 2010 and 2015. It is expected that these results - along with the contaminant dispersion numerical modelling technique - are useful for the understanding and the forecast of its effects and impacts on the coastal zone of the region.

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THE BAHÍA BLANCA ESTUARY: AN INTEGRATED OVERVIEW OF ITS GEOMORPHOLOGY AND DYNAMICS

M.C. PICCOLO, G.M.E. PERILLO AND W.D. MELO

1 INTRODUCTION

Estuarine dynamics are dependent on the geomorphology, freshwater input, tides and meteorological conditions predominant in the system (Perillo 1995a,b). Based on the geomorphological features, tidal propagation into the estuary is strongly modified and can produce variations of the tidal range along the channel and in the resulting tidal currents and residual transport of salt, sediment, nutrients and contaminants. The large intertidal areas of the Bahía Blanca Estuary (Figure 1) play a major role on the physical processes that are observed in this environment. On the other hand, these same morphological features make the standard measurement methodologies inadequate to control the whole system (Perillo and Piccolo 1998). The objective of this chapter is to provide a general and integrated description of the geomorphology and the physical oceanography of the Bahía Blanca Estuary as a basis for understanding the biogeochemical and modelling efforts presented in other chapters of this book. It is based on previous reviews by Perillo and Piccolo (1999), Perillo et al. (2000, 2004), Cuadrado et al. (2004) and on data gathered since then.

2 GEOMORPHOLOGY

Argentine estuaries are concentrated along the temperate Southwestern Atlantic coast and range from microtidal (e.g., Rio de la Plata) to macrotidal (e.g., Gallegos, Santa Cruz) (Piccolo and Perillo 1999). The Bahía Blanca Estuary (BBE) together with Anegada Bay (Figure 1) are a nearly unique case as they are atypical estuaries. They are part of the partly submerged Late Pleistocene-Early Holocene delta of the Colorado River and smaller tributaries that either no longer flow into the system or have disappeared altogether (Melo et al. 2003). An example of the historical evolution during the last 20,000 years is given in Figure 2.

BBE (Figure 1) corresponds to the northern portion of the former delta with a general triangular shape. The base of the triangle is the mouth having a width of 53 km while the length along the Canal Principal (the largest tidal channel) is 60 km. The modern Bahía Blanca Estuary extends over about 2,300 km^2 andcomprises several tidal channels, extensive tidal flats (1,150 km^2) with patches of low salt marshes, and islands (410 km^2). Owing to marked differences in surface morphology, the area can be divided along the northern shore of Falsa Bay into a funnel-shaped northern sector, characterized by the Canal Principal and many small tidal channels, and a southern sector, dominated by the Falsa, Verde and Brightman bays (Perillo and Piccolo 1999).

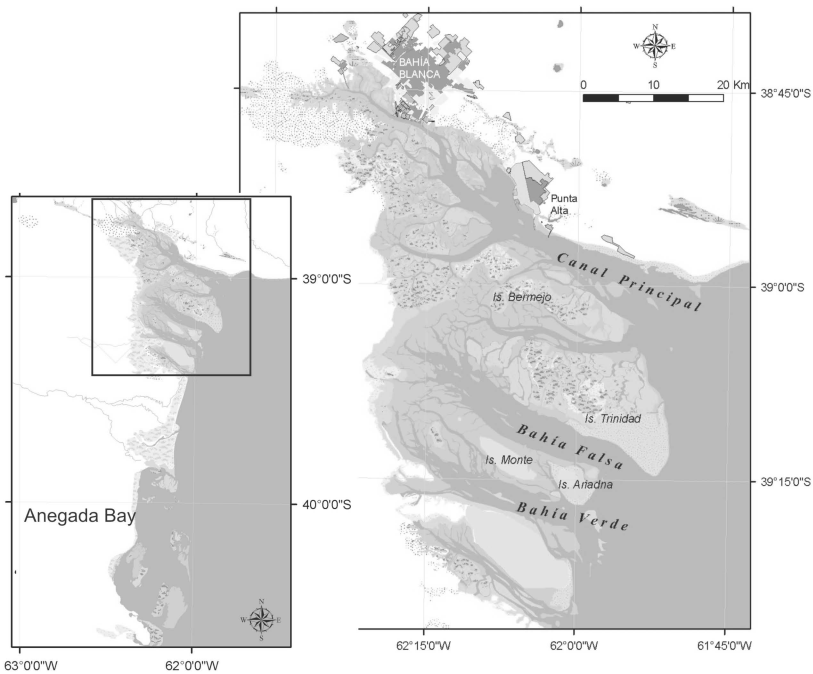

FIGURE 1: *Location map of the Bahía Blanca Estuary and its relationship with Anegada Bay.*

As indicated, the Canal Principal has a total length of 60 km and varies in width from about 3-4 km at the mouth (22 m depth) to 200 m at the head (3 m depth); both depth and width increase almost an order of magnitude from head to mouth (Perillo and Piccolo 1991). Like other major channels (bays) that flow towards the inner shelf, the Canal Principal is partly closed by a modified ebb delta (Cuadrado and Perillo 1997). The channel cross section is steep on the sides, with a U-shaped bottom having a small asymmetry to the right. Upstream of Puerto Galvan (Figure 2) the channel narrows and becomes more V-shaped with the asymmetry following the meandering pattern headward (Gómez et al. 1997). At the confluence with the Canal Principal, the funnel-shaped mouth of tributary channels is turned seawards, due to the ebb dominance, and up to 25-m-deep scour holes may develop (Ginsberg and Perillo 2004).

Most ebb deltas at the mouth of major channels have undergone changes (Gómez and Perillo 1992); however, the original delta shape and southward orientation of the ebb channel and associated shoals in the Canal Principal are still preserved despite strong marine dynamics. This is due to its stable connection to the northern shore and its position on top of a sill (Chasicó Formation), which has reduced sediment transport and served as an anchor for the ebb delta (Cuadrado and Perillo 1997). The mobility of delta shoals depends on the approach angle of tidal currents (Perillo and Cuadrado 1991), while 3D dunes in the Canal Principal and tributary channels are formed due to high current velocity and geomorphological ”traps” which favor sedimentation (Aliotta and Perillo 1987, Gómez et al. 1997). Except for bed forms in flood-dominated channels, all 3D dunes and shoals in the BBE have ebb dominance. Since sediment runoff from rivers is virtually absent and ebb delta characteristics impede sediment input from the shelf, the high concentration of suspended sediments in the estuary is due to erosion of tidal flats and island shores (Ginsberg and Perillo 1990).

The southern coast of the Canal Principal across Ingeniero White Harbour has retreated up to 50 m between 1980 and 1986 and 1.5 million m^3 were exported from an 8-km stretch in the mid-reach of the channel (Perillo and Sequeira 1989). However, the dredging of the harbour and navigation channel to a nominal depth of 45' (13.5 m) in 1989-1990 affected most of the tidal flats located along large portions of the southern coast of the Canal Principal. The tidal flats have become *Spartina* marshes in about 3 years time after the dredging. At the present time, there are indications that the *Spartina* marshes are now in an incipient erosional stage, probably due to lack of sediment input.

Because most sediment in the estuary are silts and clays (Gelós et al. 2004), strong currents and short slack water intervals impede their deposition in channels and on the tidal flats, while short, locally generated waves erode old sediments and prevent any accumulation of new ones. These conditions explain the erosional stage of most of the estuary and the prevalence of sediments from the deltaic deposition period. Furthermore, the biological and physical interactions in the system are rather important in the development of tidal creeks (Perillo and Iribarne 2003) and play a role in the erosional processes. For instance, Minkoff et al. (2005) and Minkoff (2007) have demonstrated that crabs and plants acting together were responsible for eroding over 2200 m^3 of sediments of a 270 Ha high marsh in 45 yrs.

3 PHYSICAL OCEANOGRAPHY

3.1 Freshwater Input

Two freshwater tributaries enter the estuary from the northern shore. The Sauce Chico River (drainage area of 1,600 km^2) discharges into the Canal Principal about 3 km downstream from the head of the estuary and the Napostá Grande Creek (drainage area of 1240 km^2) reaches the estuary about 1 km downstream of Ingeniero White Harbour (Figure 1). Both tributaries behave similarly in spring and summer during maximum mean rainfall but they are out of phase in autumn when the Sauce Chico River has a secondary mode (Piccolo and Perillo 1990). Although mean annual runoff flows of the Sauce Chico River and the Napostá Grande Creek are low (1.5-1.9 and 0.5-0.9 $m^3\,s^{-1}$, respectively), runoff from the Sauce Chico River may peak between 10 and 50 $m^3\,s^{-1}$, with a recorded maximum of 106 $m^3\,s^{-1}$ in 1977, although there are other sources that indicate a maximum of 200 $m^3\,s^{-1}$. Besides these major freshwater inputs, the inflows from other, smaller tributaries into the estuary are intermittent and only significant during periods of high local precipitation. However, the largest input of freshwater, nutrients and contaminants is provided by the sewage discharges from Bahía Blanca, Punta Alta and Ing. White cities.

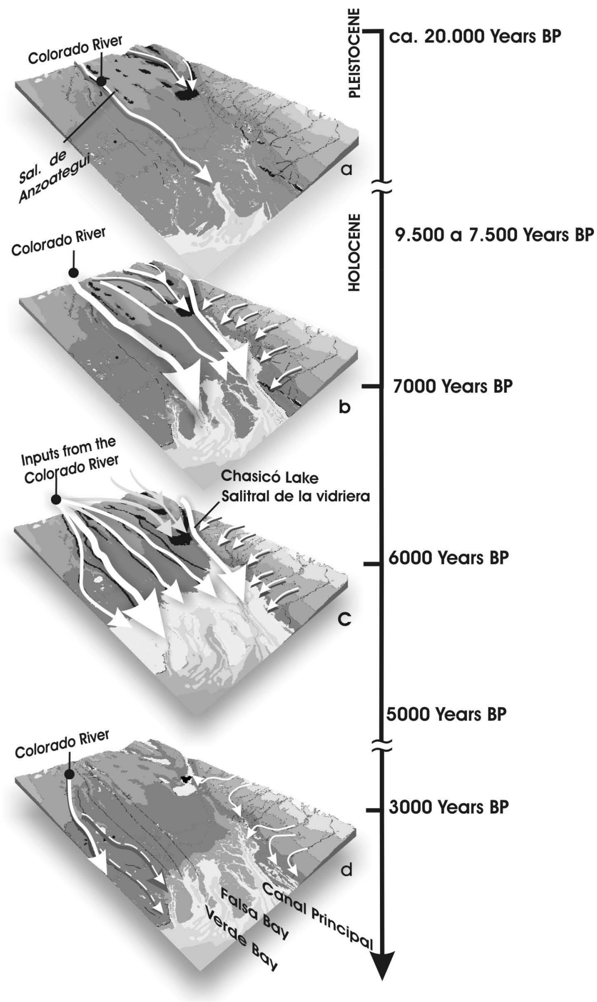

FIGURE 2: *Historical evolution of the Bahía Blanca Estuary during the last 20,000 years.*

3.2 Tides and Tidal Currents

The principal energy input into the Bahía Blanca system is produced by a quasi-stationary, semidiurnal tidal wave. The mean tidal amplitude varies between 3.5 and 2.2 m at the head and mouth of the estuary, respectively. The amplitude to depth ratio is large in the inner reach varying from 0.09 to 1.39. Although, this ratio may appear small for the outer reach stations, it is caused by a geomorphologic difference. There is a tendency for the semidiurnal constituents to increase from the mouth to the head of the estuary. M2 is the most important tidal component, followed by N2, S2, and K1, at Ingeniero White, Puerto Belgrano and Oceanographic Tower, respectively. At the Oceanographic Tower the tide is predominantly mixed semidiurnal, but along the Canal Principal it is purely semidiurnal.

The propagation of the wave is affected by the geometry of the channels and intertidal environments and islands. Reflection on the channel flanks and the head convert the original progressive form into a standing wave, while bottom and wall friction drain energy from the wave. The amplitude of the tidal wave increases with a decrease in depth and/or breadth of the channel; therefore, the actual amplitude of the tide along the estuary is directly related to the balance between friction and convergence. Thus, the Bahía Blanca is a hypersynchronous-type estuary, where the amplitude increases steadily from the mouth to the head, implying that the convergence effect on the tidal wave is larger than the friction effect.

A strong reduction occurs along the Sauce Chico River, due to a frictional effect associated with dampening produced by river flow on one side and spreading over salt flats at the other side. The average rate of tidal energy dissipation per unit mass of fluid due to friction is 0.0017 $m^2\ s^{-1}$ (Perillo and Piccolo 1991). Large negative dissipation rates in the outer reach of the estuary are due to differences in amplitude between the mouth (Bermejo Island) and the head. Differences in duration between flood and ebb indicate that depth mean longitudinal current velocities in the Canal Principal and other major channels are asymmetrical and largely caused by the extensive tidal flats bordering the Canal Principal (Piccolo and Perillo 1990). The peak ebb current is about twice the maximum flood current in the upper reaches; however, the flood usually has a longer duration.

Typical velocity profiles over a tidal cycle for two stations located on a cross-section of the Canal Principal near the Ing.White Harbor are shown in Figure 3. Those profiles clearly show examples of flow acceleration and deceleration and the reversing condition of the flow. When tidal records are compared with the predicted astronomical tide, large differences both in height and time are encountered. In general, winds blowing from the NW sector produce a set down of the water level by pushing the surface water out. SE winds generate the opposite effect. However, the predominant winds from the NW and N produce the greatest tidal variations by: (a) advancing the time of low water, (b) delaying the time of high water, and (c) reducing the predicted water levels at both high and low tide (Perillo and Piccolo 1991).

Storm surges or tidal perturbations were studied employing two years (October 1983-1985) of hourly tidal records measured at Ingeniero White and the Oceanographic Tower (Perillo and Piccolo 1991). Ingeniero White has the largest number of cases with no perturbation; however, it presents the highest deviations from the predicted tides. In 24 cases the surges were larger than 2 m, with negative and positive peaks of - 4.01 m at low tide and 2.39 m during a falling tide, respectively. At the Oceanographic Tower station, peak values were - 1.51 m (high tide) and 1.87 m (rising tide). In both stations the maximum negative surges coincide with winds blowing from the NW and maximum positive ones with SW winds. The latter are very intensive winds that act over the intertidal flats as they are covered inducing a set-up against the coast.

Northern winds produce a set down of the tide while southern winds produce a set up. When the wind blew from the N-NNW it caused negative mean deviations of -22 and -12 cm, meanwhile WSW winds generated positive deviations of 52 and 17 cm. A simple analysis relating the azimuth of the Canal Principal at Ingeniero White (129°) to the wind direction during high and low water conditions indicates that for winds blowing from the south, the storm surge should be positive, and negative for the northern winds. This result holds for all cases but the SE one. For SE winds, the storm surge was positive about half of the time during high tides and unknown circumstances may have generated a modification of the behaviour.

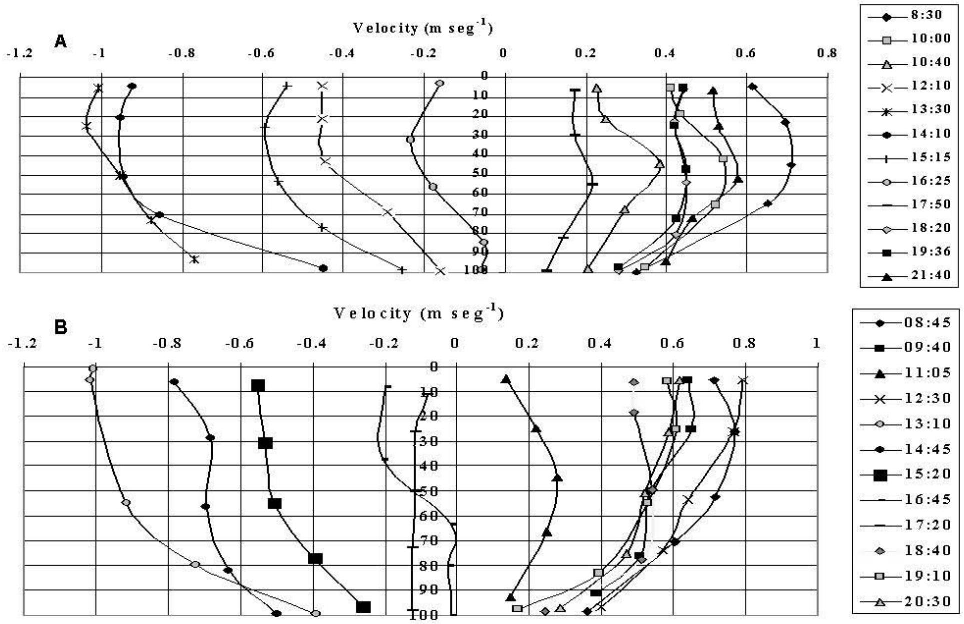

FIGURE 3: *Velocity profiles along a tidal cycle at two stations located across the Canal Principal of Bahía Blanca Estuary near Ing. White Harbor. a) northern station, b) southern station.*

A significant relationship was also found between positive and negative time (Δt) and height (ΔH) differences and the stage of the tide (Figure 4a). Considering all cases, the coincidence was over 90%. On the other hand, if the relationship between wind and channel orientation is taken into account, for northern winds positive time differences (delay) should be associated with high water and negative ones (advance) with low water. The opposite occurs for the southern winds. However, the observations show no coincidence with the previous reasoning for the first three groups (0 h $<$ time $<$ 432 h) and complete coincidence for the last one (606 h $<$ time $<$ 668 h). In the fourth group (438 h $<$ time $<$ 600 h) the agreement is relative since only one low tide value was negative.

Long-term changes (time scales $>$ 10 days) of the wind over the sea level are predominant for shorter scales, the spectra (not shown here) have common energy peaks at 6.4, 4, 2.9 and 2.1 days. The phase propagation from the inner estuary to the mouth for periods shorter than 16 days has an average time lag of 2.5 h. The sea level fluctuations, for periods between 2.2 and

5.6 days and larger than 10 days, are normally driven by the NW and W winds, respectively. Therefore, the continental winds play a major role in generating long period variations within the estuary. Although both stations have similar responses to wind forcing, the wind effect is greater in the inner estuary. The differences may be associated with their particular geomorphologic setting. The Oceanographic Tower is located in an almost open ocean situation; therefore, only when the extensive tidal flats are covered by water both stations can show similar characteristics. Nevertheless, the analysis of coherence and wind direction points out that the subtidal fluctuations at the Oceanographic Tower are associated with the regional wind pattern at periods shorter than 7 days.

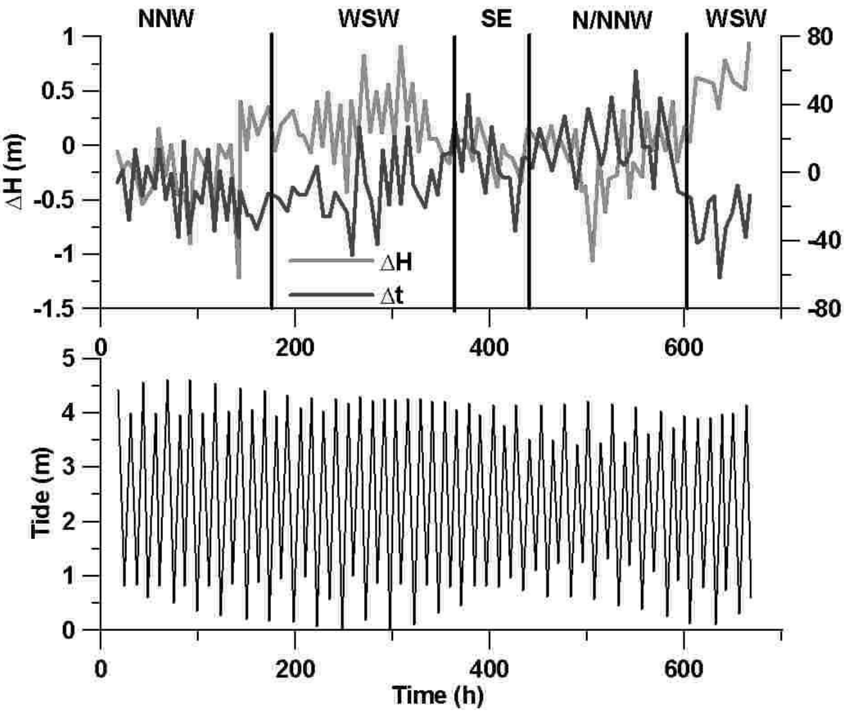

FIGURE 4: *a) relationships between wind direction and height (H) and time (t) differences between real and b) predicted tide at Ing. White Harbor.*

On the other hand, at high frequencies, the wind produces two types of waves in the estuary: (a) wind waves, and (b) waves formed by the interaction of wind and tidal currents (Perillo and Sequeira 1989). Small waves of about 5 to 10 cm in height, maximum wavelengths of 1 to 3 m and periods of 1 to 3 s characterize the Canal Principal and the intertidal flats during high tide. Oceanic waves are only important on the coast outside the estuary and on the outer banks. Waves generated within the channels by the interaction of the incoming tide with the wind blowing from the northern sector are extremely steep, up to 1.5 m high, with wavelengths of the order of 10 to 30 m.

3.3 Salinity and Temperature

Mean annual temperature based on continuous data gathered between 2000 and 2007 at Puerto Cuatreros and Ing. White stations is 15.6 ℃, with maximum and minimum registered values of 29.2 ℃ and 3.6 ℃, respectively. Longitudinal temperature distributions vary between rainy periods in spring/summer and low runoff in winter, when the vertical thermal structure of the estuary is homogeneous and longitudinal variations are less than 3 ℃. Mean surface salinity increases exponentially from the head to the middle reach of the estuary. Depending on runoff conditions, salinity differences between the mouth and head of the estuary may reach 17 and more than 4 between surface and bottom. At low water, haline stratification (halocline 1-3 m) may occur in areas of freshwater inflow, while salinity tends to be homogeneous in the outer estuary (Figure 5).

Salinity and temperature distribution therefore characterize the Bahía Blanca Estuary as type Ia (Hansen and Rattray 1966). During normal runoff, the partially mixed inner region between the mouth of the Sauce Chico River and Ingeniero White Port tends to become vertically homogeneous at low runoff, while salinity patterns in the outer homogeneous region are similar to the adjacent continental shelf. The boundary between these regions depends on river discharge (Martos and Piccolo 1988, Piccolo and Perillo 1990).

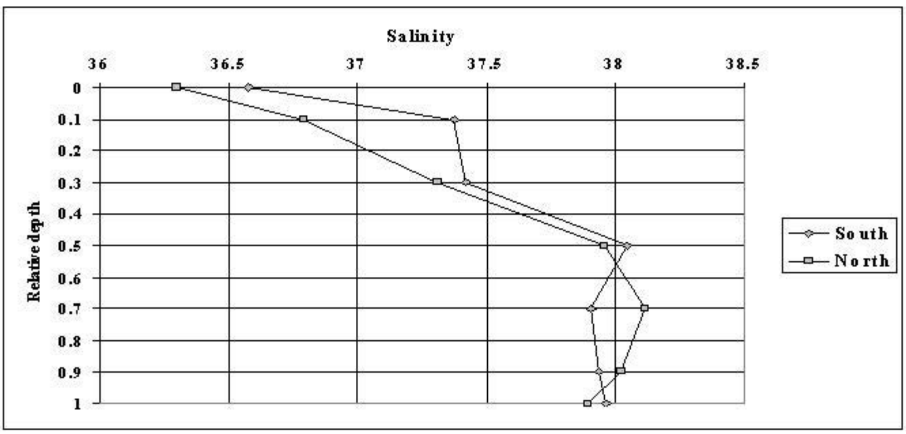

FIGURE 5: *Examples of salinity profiles at two stations of the Canal Principal of the Bahía Blanca Estuary.*

3.4 Residual Fluxes

Examples of cross-section residual fluxes at the Canal Principal near Ing. White Harbor are given in Figure 6 (based on the methodology developed by Perillo and Piccolo 1993, 1998). The new field and data reduction methodology insures an error-free calculation of the fluxes even with complex cross-section morphologies. The estuary behaves mostly as vertically homogeneous with residual flows directed in opposing directions. On the southern portion of the channel the flow is headward whereas on the northern portion the flow has two layers similar to a partly mixed estuary. However, the flows are reversed as the upper layer is headward and the lower layer seaward. Therefore, the residual circulation shows a significant difference in the direction of mass transport, causing salt concentrations in the inner portion of the estuary often exceeding those of the inner continental shelf.

Figure 6 clearly shows two layers: an upper one where the residual flux is seaward (positive values) and a lower one with headward flux (negative values). The upper layer becomes deeper from $x = 0.6$ northward indicating the influence of the center of the channel and the higher ebb velocities found at the stations measured there. Meanwhile, the effect of the tidal flats is well demonstrated by the smaller thickness of the upper layer southward of $x = 0.4$.

4 DISCUSSION AND CONCLUSIONS

Bahía Blanca Estuary displays one of the most complex sets of geomorphological and dynamic conditions to be found on the Argentine coastline. There are major differences with the typical river mouth estuary because of the geological conditions that created this environment. First of all, the estuary is only a part of what once was the largest delta in the country being almost 200 km wide at the distal point. For the sake of comparison, the Paraná Delta located within the Rio de la Plata Estuary is only 45 km wide, the Nile Delta is 250 km and the Niger Delta is 400 km wide. During its geological evolution (Figure 2), it went through full fluvial and subaerial conditions to a shallow bay-shelf with depths up to 7-10 m over the tidal flats today and depths of more than 30 m in the outer reaches of the channels. The Flandrian transgression retreated at a much slower pace than the transgression; the coast moved progressively, leaving sand bars and spits. Although there are no definitive paleontological features that can provide hard evidence, we believe that the climatic conditions during the retreating period were already as dry as the present one. Therefore, very little input of sediment was provided by the rivers coming into the receding bay.

This supposition is derived from a comparative geomorphological analysis of Anegada Bay. After its formation, this southern portion of the Colorado Delta had very few fluvial (if any) tributaries coming into it, thus its evolution was dependent only on the input of sediments from the shelf. However, both Anegada and Bahía Blanca have very similar geomorphological properties. Normally, sediment from the shelf does not enter the estuary (most probably the same is true for Anegada Bay, although there are no studies dealing with this subject there).

Based on the salinity and temperature distribution alone, the Bahía Blanca Estuary is divided in two. The inner part, from the mouth of the Sauce Chico River to Ingeniero White, is classified as a partially mixed estuary during normal runoff conditions, but with a strong tendency to become vertically and even sectionally homogeneous during low runoff. The outer part is sectionally homogeneous. The boundary between both parts is transitory and depends on the river discharge. According to the classification of Hansen and Rattray (1966), the estuary may be characterized as Type Ia.

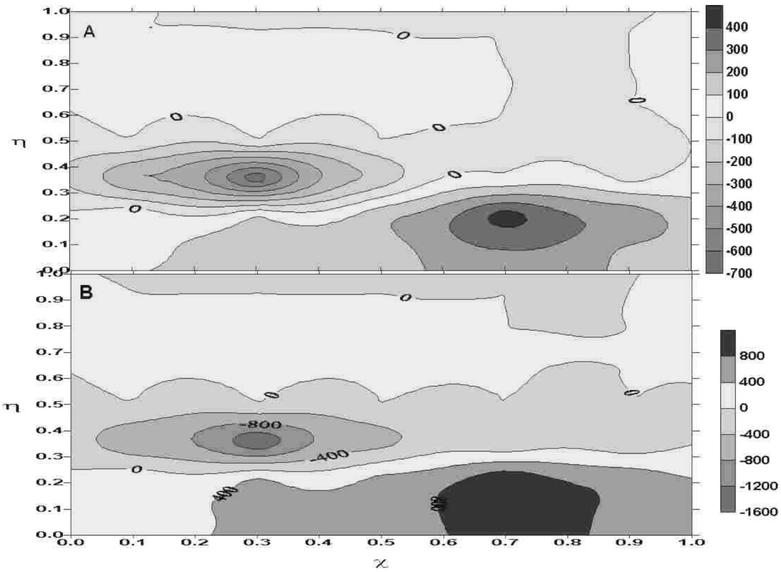

FIGURE 6: *a) Mass and b) salt residual flux distributions at a cross-section near Ing. White Harbor on spring tide.*

The mixing regime is characterized by Ri < 2 and it is associated to strong turbulent processes during maximum ebb and flood conditions, in particular on the northern flank of the estuary. Therefore, fully developed mixing is found almost throughout the tidal period. Stratification is established at the beginning of the ebb cycle, being generated by the combined outflow of the river and the ebbing tide, producing a seaward tilt of the isohalines. The process is enhanced by low turbulence levels due to the small currents. As the current velocity increases, turbulent mixing also increases. During the flooding stage, the possibilities of developing any stratification are very low since the flood is driving well-mixed water from down-estuary. Turbulent mixing at midflood is high and practically constant resulting in low Ri.

The residual circulation shows a significant difference in the direction of the mass transport. On the deeper parts of the sections (northern flank) the flow reverses with depth, being headward near the bottom. The net transport is completely landward in the shallower parts. This behaviour, added to the washing of the back estuary saline, produce a concentration of salt in the inner portion of the estuary, resulting in salinities larger than those observed in the inner continental shelf. Analysis by Piccolo and Perillo (1990) of the current stations located on the southern flank of the Canal Principal may indicate that the asymmetry of the tidal current is mainly caused by the presence of the extensive tidal flats along the channel.

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CLIMATOLOGICAL FEATURES OF THE BAHÍA BLANCA ESTUARY

M.C. PICCOLO

1 INTRODUCTION

Climate variations have significant economic and social consequences, particularly in regions with important agricultural production, such as the Buenos Aires province. The Bahía Blanca estuary is located in the temperate climate zone, with well-defined seasons. This implies that the summers and winters are rigorous and the intermediate seasons more benign. Significant spatial variations in temperatures and precipitation characterize the region. They relate to exposure to dominant air flows, orientation of the coast and presence of ocean currents. In the study area, there is a continuous flow of different types of air masses, which cause a significant variability in weather conditions (Capelli de Steffens and Campo 1994).

The southwestern region of the Buenos Aires province is under the influence of semi-permanent anticyclone centers located in the Atlantic and Pacific Ocean (Figure 1a). These two major systems generate typical air masses and determine the dominant synoptic conditions of the area. The Subtropical Anticyclone center of the South Atlantic originates a warm and humid flow of air from the North that affects the Argentine coast in particular in the Buenos Aires province. This movement is responsible for most of the rainfall in the coastal plain which characterizes the Bahía Blanca estuary. The Northern flow returns to the ocean after downloading the rainfall in the Ventania hills and in the coastal plain south of Buenos Aires. That is why the winds that reach the estuary from the N and NW have continental characteristics. On the other hand, the South Pacific anticyclone originates air masses that enter the Patagonia region with a SW-NE direction. The winds from the SW sector are always dry. Cyclonic centers flow from the S and SW across the Andes and then through Patagonia towards the estuary. The convergence of all the air masses from the Atlantic and Pacific oceans drives the formation of fronts that often cause heavy rainfall. Another weather system that sometimes has influence on the region is a low pressure center located in the NW of Argentina. This low pressure center reinforces the Northern circulation, causing a significant instability in the study region.

The typical climatological features and weather systems of the area are described in this chapter. The Bahía Blanca meteorological data (38°44' S - 62°10' W) represent the weather conditions in the inner part of the estuary and the Coronel Rosales data the outer ones. Unfortunately, the time series are not of equal length. The Bahía Blanca data is a time series of almost 100 years, but the Coronel Rosales of only 5 years (2000-2004). Significant differences in the synoptic conditions between the internal and external sectors of the estuary were the reasons for the installation of a meteorological station in the Coronel Rosales harbor (Piccolo and Galíndez 1984).

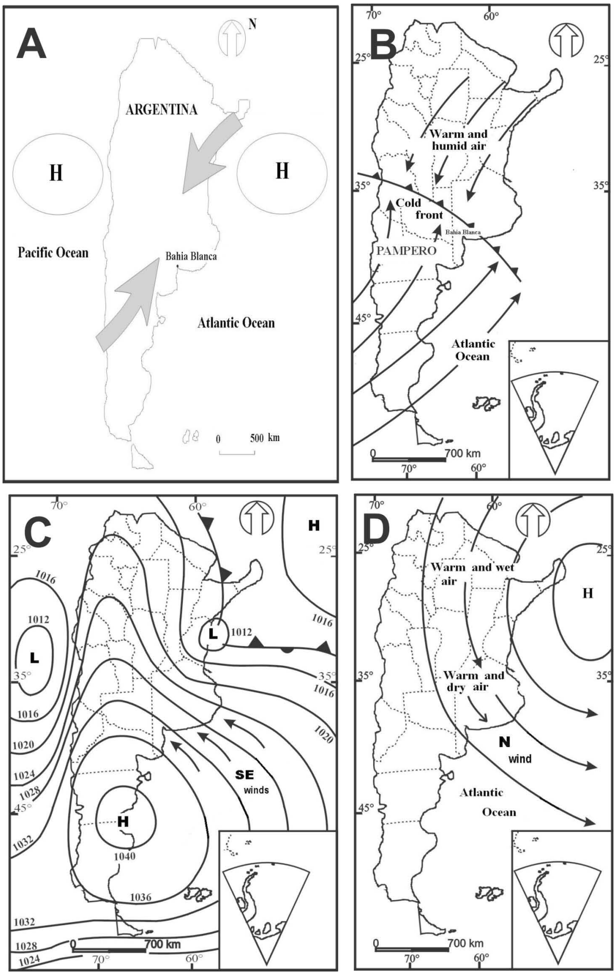

FIGURE 1: *a) Mean location of the ocean anticyclones that generate typical synoptic weather conditions in the Bahía Blanca estuary; b) Typical synoptic conditions during the Pampero; c) synoptic conditions for Southeast Winds and d) Typical weather conditions for North Winds (Modified from Capelli de Steffens and Campo 2004).*

2 TYPICAL SYNOPTIC WEATHER SYSTEMS

Weather conditions are crucial for the success of numerous activities (economic, strategic, research, entertainment, etc.) carried out from the harbors. The Bahía Blanca estuary is affected by various synoptic weather systems which depending on their origin present different characteristics. Three main weather systems are characteristic in the estuary: the Pampero (wind from the Pampas), Southeast winds (Sudestada) and the North Wind. These flows have different directions and a great influence on wave height and the amplitude of the tides in the estuary.

2.1 Pampero

It is a synoptic condition that develops in the course of approximately a week, strong winds blow from the South. It is a cold, fresh or temperate wind according to the season but always dry. Its speed exceeds 40 $\mathrm{km\,h^{-1}}$. In its initial phase precipitation frequently occurs and it is then called *Wet Pampero*, otherwise it is called *Dry Pampero*. In which case dust storms occur particularly in arid regions and it is characterized by the presence of fine dust, which significantly affects visibility and even the health of the population. The beginning of a Pampero coincides with the passage of a cold front.

The front is generated by two air masses. One is generated by the South Pacific anticyclone. The air mass crosses the Andes Mountains and downloads its moisture on their windward slopes. The other air mass is originated by the South Atlantic anticyclone that dominates the Argentine plains (Figure 1b). It is a warm and wet flow from the North. In a second phase, the cold front moves toward the NE. In its flow across the southern Buenos Aires province, the atmospheric pressure increases and the relative humidity decreases. In its final stage the sky is clear, calm, and the air temperature drops significantly.

2.2 Southeast Wind (Sudestada)

Continuous precipitation and wind from the SE and E characterize this weather condition that affects the coast of the province. Regular to strong winds from the Southeast sector blow with speeds over 35 $\mathrm{km\,h^{-1}}$. It is accompanied by persistent rains and relatively low temperatures. The air mass that generates this synoptic system is a detachment from the South Pacific anticyclone that travels across the Patagonian region, favored by a high pressure system especially during winter and spring. The trajectory of the anticyclone is W-E and when located in the Atlantic Ocean off the coast of Buenos Aires province it causes serious damage (Figure 1c). The strongest flows are found in the month of October with approximately one week of cold weather and rain. The devastating effect on inhabited coastal areas is great. Coastal inundations occur, boats and buildings located on the beach are usually damaged, etc.

2.3 The North Wind

The North wind prevails in the southern Buenos Aires province. It blows more frequently in spring and summer, generating high temperatures, changes in atmospheric pressure and dryness in the environment. It is usually accompanied by clouds of dust and smoke from the fires that frequently occur in the summer in the plain regions of the province. A direct consequence is the reduction in visibility due to particulate matter, which may affect ground and air transportation. The western edge of the South Atlantic anticyclone originates this airflow and discharges its humidity on the coast, center of the country and the pampas plain. Consequently, the South of the Buenos Aires province receives a mass of dry and warm air from the North that produces irritability in the population due to the excessive concentration of positive ions (Figure 1c).

3 MEAN METEOROLOGICAL FIELDS

Climatological Data from the Argentine National Weather Service (S.M.N. 1992) were analyzed to study the mean meteorological parameters. Rainfall shows a gradual eastward increase at the estuary latitude. From a total of 300 mm in Viedma - Patagones (40°50' S - 63° 00' W), it increases to 530 mm in Hilario Ascasubi (39°22' S - 62°39' W), 613 mm in Bahía Blanca and 841 mm in Tres Arroyos (38°33' S - 60°25' W). The distribution of rainfall reveals the influence of the Westerly winds regime that imposes characteristics of dryness on the entire Patagonian coast. Towards the Southwest an arid continental climate is typical. One example is presented in Table 1 with climatic information pertaining to El Rincón Lighthouse station located on the eastern tip of the Verde peninsula where the mean annual precipitation is 498 mm in a distinctly maritime area typical of the Bahía Blanca estuary (S.M.N. 1941-1990).

In the estuary, the maximum rainfall values come in the months of March (90.9 mm) and October (80.9 mm). The minimum rainfall occurs during the winter. As a rule, June is the driest month (16.5 mm) (Figure 2). This fact is reflected in the landscape and vegetable farming characteristic of the region. The humidity is high for the region due to the proximity of the estuary. The annual average atmospheric pressure is normal, 1013.1 hPa, although 1037 hPa maximum and 977.1 hPa minimum have been measured in July 2002 and in March 2001, respectively.

The thermal regime in the area shows a large difference between winter and summer, while spring and fall present great similarity in temperature. The mean annual temperature is 15.1 ℃ with thermal annual amplitudes evidencing the moderating effect exerted by the ocean in the coastal areas (Campo and Capelli de Steffens 2000). The beginning and end of the summers present significant rainfall and high temperatures. Winters are cold and frost occurs on average about 8 days a month in June and July, dropping to 7 in August. Snowfall and fog are exceptional in the region. On average, fog may occur about 16 days a year with the highest occurrence in winter.

The spatial and temporal variability in the precipitation is large. The precipitation is closely related to increasing temperature. The thermal amplitudes decrease toward the East showing a transition from continental to ocean climate. The presence of the estuary originates low temperatures in its vicinity. Simultaneously, the heat island phenomenon which characterizes Bahía Blanca city causes higher temperatures inside the city. As a consequence, there are significant differences in temperature between Bahía Blanca city and its periphery. The average annual temperature of Bahía Blanca is 2 ℃ higher than in Ing. White and 0.5 ℃ higher than that obtained in the adjacent rural environment (airport station). Broadly this can be attributed to the influence of man-made heat generated by the city of Bahía Blanca through its multiple activities (Capelli de Steffens and Piccolo 1987).

The last 100 year of temperature and precipitation analysis is presented in Table 2. Average annual temperatures decreased up to 1950. In the following two decades, a slight increase of 0.2 ℃ in the average temperature was found and in 1980 the temperatures increased another

0.2 °C. The seasonal distribution of temperature and precipitation show that summer and spring follow the trend of annual increases in temperature and decreases in precipitation. The rain in the summer during the 1981-90 decade only reached 70% of the total rainfall of the previous period, but in the last decade it reached the values of the 1971-80 one once more. The mean temperatures are uniform in general but in the last decade a significant increase in temperature and precipitation was recorded.

Several singularly extreme temperatures were recorded last century. On January 21, 1980, 43.8 °C was reached. This value was the absolute maximum temperature in the century, while the absolute minimum was recorded on July 4, 1988 at -11.8 °C during a so called "cold wave" which lasted for thirteen consecutive days.

Another important theme is the variability of the precipitation observed in different sectors of the estuary. The monthly rainfall varies substantially at only a few miles distance. As an example, Figure 3 shows the monthly rainfall for the year 2001 in three locations near the estuary: Ing White harbor, Puerto Rosales harbor and Malaver lagoon located approximately 8 km from Puerto Rosales harbour. In Malaver lagoon it rains more than in the other two stations and in Puerto Rosales more than in Ing. White. The analysis of the 2000-2004 period presented the same results (Piccolo and Diez 2004).

3.1 Winds

The winds are persistent all year round. Their annual mean velocity is 22.5 km h^{-1}. The number of days a year with strong winds is significant (196), always over 43 km h^{-1}. An analysis of the wind speed in different periods shows a great variation (Figure 4). The decade 1971-1980 presented the highest mean values all along. The minimum mean values registered in the period 1901-1950 can be explained by several reasons. One is the change of location of the weather station. It is now 83 m above mean sea level (MSL); it used to be 20 m. Obviously, this timeseries thus has a serious methodological flaw.

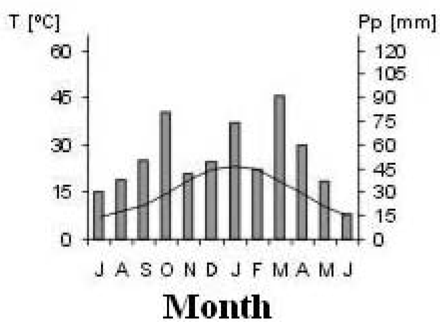

FIGURE 2: *Relationship between Precipitation (Pp) and temperature (T). (Period 1981-1990) (Modified from Capelli de Steffens and Campo 2004).*

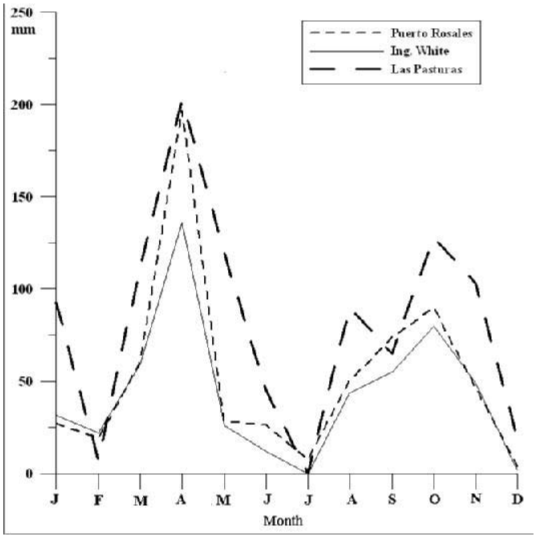

FIGURE 3: *Precipitation in three locations near the Bahía Blanca estuary during the year 2001.*

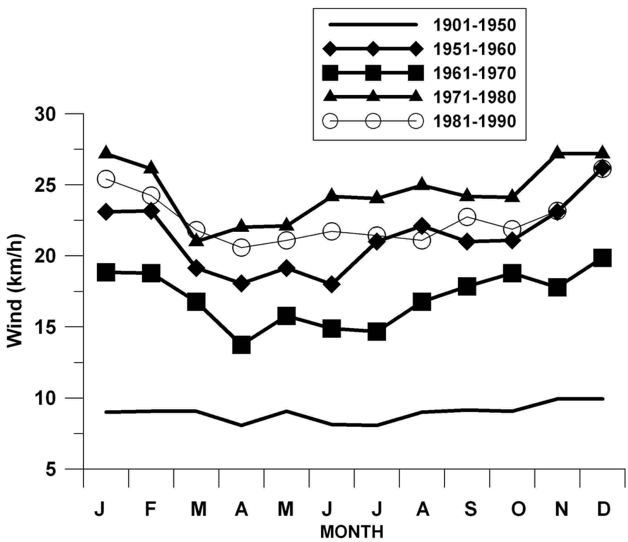

FIGURE 4: *Wind velocities in the Bahía Blanca estuary in different decades. (Modified from Capelli de Steffens and Campo 2004).*

Autumn is the season that presents the lowest wind velocities, whereas the end of the spring and the summer present the highest. This could partly explain the disparity in records such as the value of the beginning of the century with an annual average of 9 km h^{-1} which contrasts dramatically with subsequent registrations. Another reason is that the average velocity was taken from 50 years of records, therefore the result can be affected by smoothing.

Storms are also frequent in the region. The maximum record of the century was obtained on March 21, 1981. West and Northwest winds reached a maximum intensity of 160 km h^{-1}. The damage in the city was noticeable. Figure 5 shows the annual wind rose for Bahía Blanca for the period 1981-1990 where the predominance of the Western, North, and Northwest directions is emphasized. Calm days only reach 10% of total registrations. Along the year the prevailing wind is from the North during all seasons. In the summer winds from the East, Southeast and South are frequent and the number of calm days is low. In autumn the frequency of winds increases especially from the Northwest and West with the highest velocities in winter. In the spring winds diminish and the prevailing directions are from the West and Northwest.

On the other hand, the average wind speed for Puerto Rosales is 22.6 km h^{-1}. The highest gust observed was from the SSW with 132.5 km h^{-1} in the year 2002. Figure 6 shows the highest gusts recorded in the 1999 and 2002. They represent the years in which the minor and major gusts, respectively were observed. The more intense gusts correspond to winds from the SSW, SW, NNW and NW directions. The gusts in Puerto Rosales were greater than those in Ing. White for the 2000-2004 period. In both places the highest gusts were from the SW direction. Although the measurement period in the Puerto Rosales station is very short to establish general climatic characteristics of the area however, comparison with other stations located in the hinterland of the estuary, made it possible to determine that Puerto Rosales presents synoptic conditions which are much more exposed to the marine action and to the moderating effect of the sea (Piccolo 1985). The minimum Puerto Rosales temperatures were greater than those recorded in the hinterland of the estuary and the maximum temperatures were lower. However, the wind speed was higher than in the inner estuary region (Piccolo et al. 1989). There is similar variation in the other meteorological parameters.

4 CONCLUSION

A synthesis of the most important features of the Bahía Blanca estuary weather conditions is presented in this chapter. The estuary is located in a temperate climate. In the last 100 years mean temperatures oscillated around 15 °C. Annual precipitation varied between 540 mm and 712 mm. The spatial and seasonal variability of the precipitation is significant. Few studies were performed on the subject (Piccolo and Diez 2004). The winds are persistent all year round. Their annual mean velocity is 22,5 km h^{-1}. Strong winds, over 43 km h^{-1}, blow over the estuary 54% of the whole year. The estuary is affected by strong winds associated to distinct synoptic weather conditions. The characteristic wind from the South, from the pampas, is called Pampero. It is associated with a cold front which crosses the estuary. The pampero is usually a dry airflow. The Southeastern wind, denominated ”Sudestada” brings rain, fog and causes many problems to the population due to its bad weather conditions. Finally, the most frequent wind direction is from the North / Northwest.

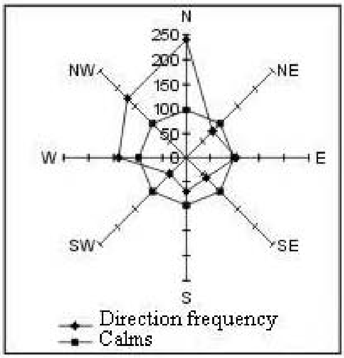

FIGURE 5: *Wind Rose at Bahía Blanca station (1981-1990). (Modified from Capelli de Steffens and Campo 2004).*

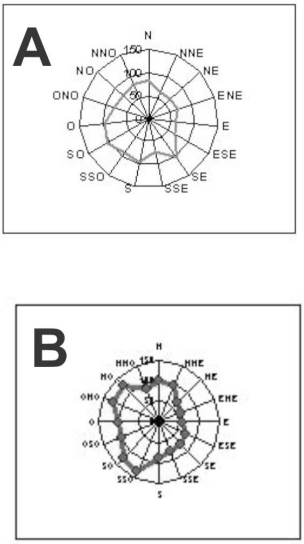

FIGURE 6: *Puerto Rosales gust wind rose (km h^{-1}) a) in 1999; b) 2002 (Piccolo and Diez 2004).*

TABLE 1: *Mean annual parameters in El Rincón Lighthouse station (Capelli de Steffens and Campo 2004).*

Parameter	**Annual Mean Values**
Atmospheric Pressure (hPa)	1013.3
Temperature (ºC)	14.4
Relative Humidity (%)	73
Mean Wind velocity (km/h)	15.8
Precipitation (mm)	498.0

TABLE 2: *Mean seasonal temperature and precipitation (National Meteorological Service Statistics).*

	Summer		Autumn		Winter		Spring		Anual	
Period	T (ºC)	Pp (mm)	T (ºC)	Pp (mm)	T (ºC)	Pp (mm)	T (ºC)	Pp (mm)	T (ºC)	Pp (mm)
1901- 50	22.8	156	15.3	165	8.5	61	15.2	158	15.5	540
1951- 60	21.8	157	14.5	177	7.9	86	14.3	138	14.7	558
1961- 70	21.7	185	14.9	164	8.4	84	14.6	141	14.9	604
1971- 80	21.9	240	14.7	205	8.1	78	14.7	189	14.9	712
1981- 90	22.7	168	14.6	188	8.1	86	14.9	174	15.1	614
1990- 00	22.3	237.6	16.6	173	9.2	115.4	15.4	158.8	15.7	684.9

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WATER CHEMISTRY AND NUTRIENTS OF THE BAHÍA BLANCA ESTUARY

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S.E. BOTTÉ, V. NEGRÍN, A. ARIAS, F. DELUCCHI AND R.O. ASTEASUAIN

1 INTRODUCTION

Different characteristics of the estuarine environments (i.e. geo-morphological, hydrographical, climatic or chemical ones) strongly condition the biodiversity within those systems (Harrison 2004). So, the range of values of the different parameters involved (temperature, salinity, pH, dissolved oxygen, inorganic nutrients, organic matter, etc) as well as their corresponding seasonality or anomalies would directly determine the scenario where the whole biological processes will occur also including populations movement along different spatial gradients due to physiological problems (Laprise and Dodson 1994). In addition, the distribution and availability of non-conservative constituents such as oxygen, nutrients or organic carbon determine the potential primary production of the system, and consequently the potential transfer of energy to higher trophic levels (Cantoni et al. 2003, Rydberg et al. 2006).

Bahía Blanca estuary has been the object of a large number of environmental studies over the last 30 years. From these studies, a large time-series database on physicochemical parameters has been assembled for the inner part of the estuary (Pto.Cuatreros and Pto.Ing.White; Figure 1). This database is a very useful tool to diagnose the environmental condition of the system considering that the inner part of the estuary includes not only the entry points of the main rivers from the region but also the largest human activities such as cities, industries and harbours, with their consequent impacts on the estuary. In addition, information has also been recorded through cruises along the Main Navigation Channel, as well as from many coastal studies on the tidal flats at Villa del Mar, Puerto Galván and Puerto Cuatreros (Fig. 1).

The main goal of the present chapter is to summarize the available information on the physicochemical condition of Bahía Blanca estuary, including the corresponding range of values of the considered parameters (temperature, salinity, turbidity, dissolved oxygen, inorganic nutrients (DIN, Dissolved Inorganic Nitrogen; DIP, Dissolved Inorganic Phosphorous and DSi, Dissolved Silicates), particulate organic matter and photosynthetic pigments (Chl-a and phaeopigments), and to identify seasonal variations, as well as the influence of external sources.

2 MAIN PHYSICAL-CHEMICAL CHARACTERISTICS WITHIN THE ESTUARY

2.1 Temperature

The analysis of the obtained data has shown a very stable behavior of this parameter along the Main Navigation Channel, from the head of the estuary and up to its mouth. In fact, non significant differences have been observed between the considered areas of the estuary (inner, middle and outer), which allows to sustain that both the ocean water temperature as well as the air one are the main responsible for the variations of this parameter within the estuary. Water temperature measured in the inner zone has demonstrated to be strongly regulated by the air temperature of the region, and their corresponding curves of measured values have shown quite similar trend in their distribution of values (Figure 2).

Extreme values recorded for temperature along the 1996-2006 period within the estuary varied between 5.1 °C and 26.4 °C, which have been measured on July'02 (winter) and January'04 (summer) respectively. The distribution of temperature values has followed a sinusoidal curve, which indicates the occurrence of a thermic cycle characteristic of the estuarine conditions (Newton and Mudge 2003, Harrison and Whitfield 2006).

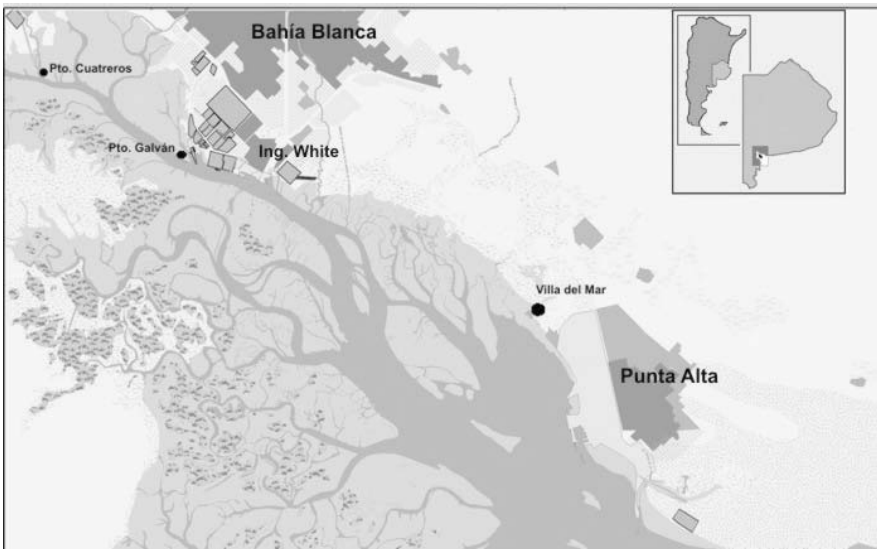

FIGURE 1: *Location of Bahía Blanca estuary and Villa del Mar, Puerto Galván and Puerto Cuatreros.*

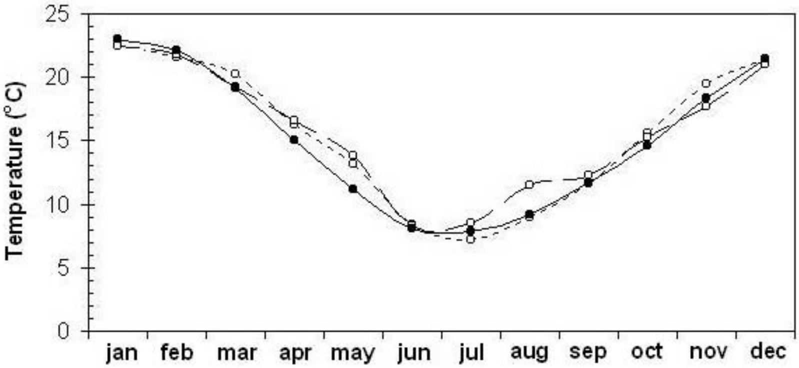

FIGURE 2: *Mean water temperature at Pto.Cuatreros (-○-) and Pto.Ing.White (- -○- -) (in 1974 - 2003) related to mean air temperature at Bahía Blanca (-●-) for the 1860-1990 period.*

2.2 Salinity

The distribution of salinity does not present a sharp gradient along the main channel of the estuary, as can be observed in many of these systems (Brockway et al. 2006, O'Callaghan et al. 2007). Even though a clear variation in salinity could be observed in the inner estuary, where a range from 17.9 psu to 41.3 psu has been recorded (Freije and Marcovecchio 2004). The very important fact that the estuary becomes ”hypersaline” in almost every summer transforms it in a negative one during that period, allowing an inwards flux. The observed distribution of salinity values fully agrees with previous reports by other authors on different estuarine systems (Ahel et al. 1996, Ysebaert et al. 2003).

2.3 pH

The distribution of pH values along the estuary has presented low differences, usually linked to seasonal changes and related to biological processes. So, the highest pH values have been recorded just after the occurrence of large phytoplankton blooms (winter and summer), reaching up to levels of $\sim$9 (Popovich et al. 2008). This seasonal distribution pattern as well as the importance of this parameter has been highlighted by several authors for different estuarine systems (Jin et al. 2006, Melville and Pulkownik 2006).

2.4 Turbidity

Turbidity in the inner area of the estuary seems to decrease seawards, considering that near the head it ranges between 50 and 300 NTU, while seawards it decreases to less than 200 NTU in the middle area of the estuary, and at the open ocean observed values close to the mouth of the estuary are lower than 30 NTU. This fact can be related with both the occurring of the main sediment sources in the inner area (streams, rivers, sewage outfalls, harbours) as well as the increasing depth from the head to the mouth of the estuary which generates lower sediment resuspension (Cuadrado et al. 1994, Perillo et al. 2005). The analysis of Pto,Cuatreros and Pto.Ing White time series has shown a slightly lower mean values at IW than at PC, and in both cases the maximum values have been recorded during stormy winters. This distribution trend agreed with the reports from other authors for different estuaries at other latitudes, not only obtained from field work (Irigoien et al. 1999) but also from remote sensing ones (Chen et al. 2007).

2.5 Dissolved oxygen

The distribution of dissolved oxygen within Bahía Blanca estuary has shown adequate values to support a significant biological production, with average levels close to 7 mg l^{-1}, and reaching up to approximately 13 mg l^{-1} during the highest productive periods (winter and late summer) (Popovich and Marcovecchio 2008). The highest concentrations of dissolved oxygen were always recorded in the inner area of the estuary; the spatial distribution trend of this parameter has shown a very stable level along the whole estuary (Popovich and Marcovecchio 2008). This observation has also agreed with the corresponding distribution of the oxygen saturation percentage that also reached the highest values in the inner region of up to 130% during phytoplankton blooms (Popovich and Marcovecchio 2008).

2.6 Dissolved Nutrients

Bahía Blanca estuary has been recognized as a nutrient-enriched environment, maintaining significant levels of these inorganic compounds during most of the year (Freije and Marcovecchio 2004). A typical spatial pattern for nutrients has to decrease from the inner zone of the estuary to the mouth (Figure 3). Thus, the mean levels of NO_2^- + NO_3^- , PO_4^{3-} and Silicates have varied from 7.76 $\pm$ 6.13 μM, 1.85 $\pm$ 1.07 μM and 80.22 $\pm$ 27.53 μM, respectively, at the inner area; and 1.36 $\pm$ 1.61 μM, 1.30 $\pm$ 0.32 μM and 20.22 $\pm$ 9.62 μM respectively, at the mouth (Popovich and Marcovecchio 2008). On the other hand, a very high stock of ammonium is usually available within the system, mainly also in the inner area, with mean values of 32.32 $\pm$ 25.78 μM, and reaching up to a peak of 102.8 μM registered in Jun'02 (Figure 4) (Freije and Marcovecchio 2004, Popovich et al. 2008). This is a very important point, considering that this species has never been completely depleted, and so represents a permanent potential stock of nitrogen for the estuary.

2.6.1 Dissolved Nutrients on the Villa del Mar tidal flat

Villa del Mar is a small resort town located on the middle Bahía Blanca estuary, $\sim$5 km from Punta Alta city ; it is a small dock where an intense sport fishery activity exist. This estuarine area shows predominant distribution of different fin-fish species, while the halophyte vegetation is dominated by *Spartina alterniflora* Loisel (Negrin et al. 2007).

Preliminary studies carried out in the tidal flat of Villa del Mar during 2006 and 2007, show that the concentration of DIN, DIP and DSi in surface estuarine water (SEW) (23.12 $\pm$ 10.46 μM, 2.91 $\pm$ 0.99 μM and 84.9 $\pm$ 7.3 μM, respectively) was lower than those determined in porewater of the usually flooded tidal flat (PW_I) (59.90 $\pm$ 49.22 μM, 16.96 $\pm$ 10.44 μM and 361.1 $\pm$ 66.1 μM, respectively) and in the occasionally flooded tidal flat (PW_{II}) (54.04 $\pm$ 8.97 μM, 20.40 $\pm$ 8.47 μM and 417.3 $\pm$ 37.20 μM, respectively) (Table 1). In these studies the occasionally flooded tidal flat was considered as the flat that only floods under certain climatic conditions, especially when strong winds came from the southern region. NH_4^+ was the dominant fraction of the DIN. In PW_I the concentration of NH_4^+ was significantly greater than in PW_{II}; meanwhile this one had the highest concentration of NO_3^-, especially during the springtime and the summer, which is being related to processes of nitrification.

The analysis of these results compared to the zones with vegetation showed a higher concentration of DIN (especially ammonium and nitrate) in flats without vegetation than in the vegetated ones. This fact has been related with the use by the vegetation (Negrín et al. 2008).

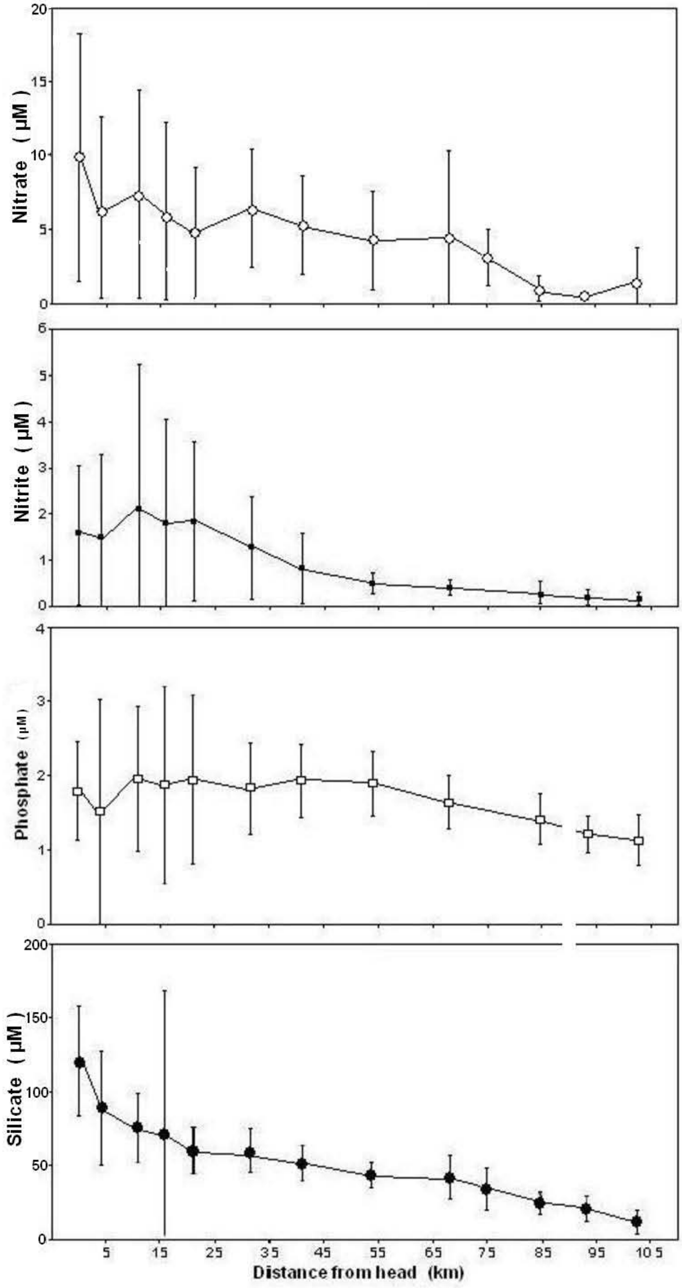

FIGURE 3: *Variation of nutrient concentrations (nitrate, nitrite, phosphate, silicate ; μM) along the Main Navigation Channel from Bahía Blanca estuary.*

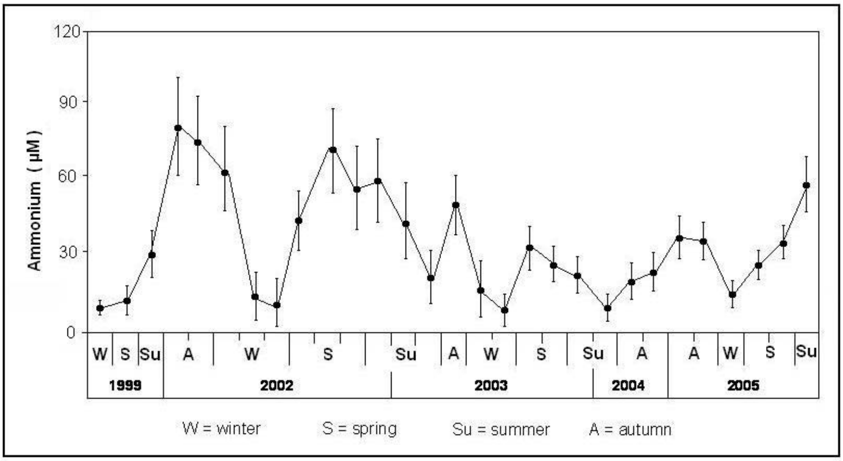

FIGURE 4: *Seasonal distribution of ammonium (mean value ± standard deviation ; µM) at the inner area of the estuary.*

2.6.2 Dissolved Nutrients in Puerto Galván tidal flat

Puerto Galván at the inner area of the estuary has a reduced surface due to the continuous increasing industries located within the area over the last decades. This site is surrounded by a large petrochemical nucleus, refineries and fertilizer plants, as well as deep water harbours. This means that periodical dredging, artisanal and commercial fisheries, and oil and cereal cargo vessel traffic usually affect this area, as well as the direct impact of both Bahía Blanca and Ing. White cities. The occurrence of both *Spartina alterniflora* and the benthic macroalgae *Enteromorpha* sp. characterize its higher intertidal area (Botté 2005).

TABLE 1: *Dissolved Inorganic Nitrogen, Dissolved Inorganic Phosphorus and Dissolved Silicates Concentrations (in µM) in Surface Estuarine Water (SEW), Porewater in usually flooded tidal flat (PW_I) and Porewater in occasionally flooded tidal flat (PW_{II}) in Villa del Mar.*

		winter	spring	summer	autumn
DIN	**SEW**	28.78	13.80	14.89	35.02
	PW_I	110.91	9.61	26.85	92.25
	PW_{II}		48.15	64.36	49.60
DIP	**SEW**	3.54	3.57	1.63	2.89
	PW_I	38.46	1.79	5.99	21.60
	PW_{II}	46.45	10.76	26.63	23.82
DSi	**SEW**	85.88	81.40	79.92	93.28
	PW_I	1025.84	437.20	328.71	317.47
	PW_{II}	470.59	459.15	404.63	388.03

According to the data compilation of dissolved inorganic nutrients in Surface estuarine water (SEW) and Pore water (PW) in the tidal flats of Puerto Galván from 2000 to 2002, obtained by Botté (2005), the concentrations of DIN, DIP and DSi were higher in PW than in SEW. In autumn 2001 (El Niño year), both in SEW and in PW were recorded the highest concentrations of DIN (150.57 $\pm$ 31.88 μM and 321.59 $\pm$ 29.80 μM, respectively) and DSi (113.1 $\pm$ 17.9 μM and 193.6 $\pm$ 54.5 μM, respectively); thus showing an important contribution from the terrestrial drainage due to the water-sediment interaction. Then, the concentration of DSi was stable around values of 90.5 $\pm$ 7.9 μM in SEW and 150.9 $\pm$ 13.7 μM in PW.

DIP did not show great variations in SEW (2.54 $\pm$ 1.18 μM), meanwhile in PW the highest concentrations were recorded in autumn (8.84 $\pm$ 2.80 μM) and winter (8.48 $\pm$ 3.20 μM). In winter the values of DIN were low (44.44 $\pm$ 0.24 μM in SEW) with minimum values in PW (74.20 $\pm$ 25.46 μM). In spring and summer the concentrations were 66.58 $\pm$ 8.82 μM and 44.91 $\pm$ 6.96 μM in SEW, and 133.90 $\pm$ 79.47 μM and 114.52 $\pm$ 50.03 μM.

Botté (2005) has made a comparison between concentrations of NH_4^+ in porewater of Puerto Cuatreros, Puerto Galván and Maldonado (a resort area placed between both harbours) where Puerto Galván highlighted with a total mean of $\sim$72 μM. In the above mentioned work it has been suggested that the tidal flat of Puerto Galván would act as a source of nutrients, without considering the important anthropogenic load that this place could be receivng.

2.6.3 Dissolved Nutrients in Puerto Cuatreros tidal flat

Puerto Cuatreros is a good representative example of the inner zone of the system, with scarce vegetation on the tidal flats, high nutrient concentrations, and high phytoplankton biomass (Gayoso 1998, Spetter 2006, Popovich et al. 2008). This area has an average depth of 7 m, with a vertically homogeneous and highly turbid water column (Píccolo and Perillo 1990). The Sauce Chico River, whose watershed comprises highly agriculture and cattle breeding lands, is the main freshwater source for the study area, with a mean annual runoff of 1.9 $m^3\ s^{-1}$, which can increase up to 10 to 106 $m^3\ s^{-1}$ with the autumn rainfall (Píccolo et al. 1990).

Puerto Cuatreros is the more evaluated site about nutrients dynamics in the inner zone of the Bahía Blanca estuary. Freije and Marcovecchio (2004) summarized the recorded information over more than 20 years of nutrient data compilation in surface estuarine water of this place. They concluded that the nutrient dynamics in Puerto Cuatreros has a typical behaviour in summer, when short duration pulses of growth of small phytoplankton species take place and they usually decrease the concentration of dissolved inorganic nutrients for some days; in autumn, when the maximum nutrient concentrations connected with the mode of rains of this place can be found; in the bloom period (a phase that recurs every year) when the concentration of nutrients abruptly falls, usually reaching its annual minima and therefore constituting a limitation to the bloom; and, at the end, a phase called ”recuperation” when the concentration of each nutrient increases and which occurs usually in springtime.

The distribution analysis of DIN, DIP and DSi in Puerto Cuatreros from 2001 to 2003 showed the previously depicted dynamics (Figure 5). However, Spetter (2006) observed during the years 2004 and 2005 some changes in the expected dynamics. The distribution of concentration of DIN in Puerto Cuatreros, in 2001 and 2002, presented a typical behaviour, with high values in autumn (92.04 $\pm$ 30.23 μM) and spring (57.91 $\pm$ 18.87 μM), coinciding with rainy periods (Spetter et al. 2008); and a large decrease in winter (28.72 μM, July 2001; 17.38 μM, July 2002), and summer (21.74 μM, Dec. 2001; 36.13 μM, Dec. 2002) (Figure 5). In contrast, lower concentrations than in previous years were observed from March 2003 to November 2005 (31.68 $\pm$ 13.43 μM) (Spetter et al. 2008).

The winter diatom bloom has been recognized as the most important event in the phytoplankton annual cycle in the Bahía Blanca estuary (Gayoso 1998, Popovich 2004). DIN -unlike P and Si- has been responsible for the highest primary production in this area (Popovich et al. 2008). Recent studies have demonstrated that the winter/early spring phytoplankton bloom (diatoms dominated by *Thalassiosira* spp. and *Chaetoceros* spp.) in the inner zone of the Bahía Blanca estuary starts in June consuming NO_3^- as first source of Nitrogen and depleting all dissolved inorganic nutrients until August (Popovich et al. 2008). The NH_4^+ seemed to be the main source of nitrogen (Spetter et al. 2008).

Small forms of phytoflagellates (10 - 20 μm) occurred year-round with maximal abundance in summer (Gayoso 1999); this suggests that they would be responsible for the strong DIN decrease (Spetter et al. 2008); however, in summer of years 2004 and 2005 other groups which could be responsible for the large decrease of nutrient concentrations were found (CA Popovich, personal communication). According to Perillo et al. (2004) the Sauce Chico River is the main freshwater input to the inner area of the Bahía Blanca estuary, in agreement with Lara and Pucci (1983) who have highlighted the influence of the above mentioned river in the studied area.

Seven years of historical data for DIN in Sauce Chico River give a mean concentration in the order of 60 μM. This high concentration of DIN compounds was related to the anthropogenic activities in the corresponding watershed, which cross trough an area intensively used for both agricultural and livestock farming activities. Spetter (2006), Spetter et al. (2008) have demonstrated that the Sauce Chico River is an important source of DIN in the inner zone of the estuary, especially during heavy rain periods.

Popovich et al. (2008) have considered Phosphorus as the main potential limiting nutrient for the winter diatom bloom such as it had been reported in many coastal and estuarine systems (Benitez Nelson 2000, Ehrenhauss 2004). Figure 5 shows that the distribution of DIP concentration from 2001 to 2005 followed the expected tendency, with high values in autumn (2.72 $\pm$ 0.54 μM) and spring (2.47 $\pm$ 1.49 μM), minima in winter (1.58 $\pm$ 0.72 μM) and values of 2.19 $\pm$ 0.78 μM in summer. It should be noted that the authors found, during this study, the minimum concentration of DIN and DIP (6.71 and 0.72 μM respectively) in January 2004, instead of in winter as it was previously depicted.

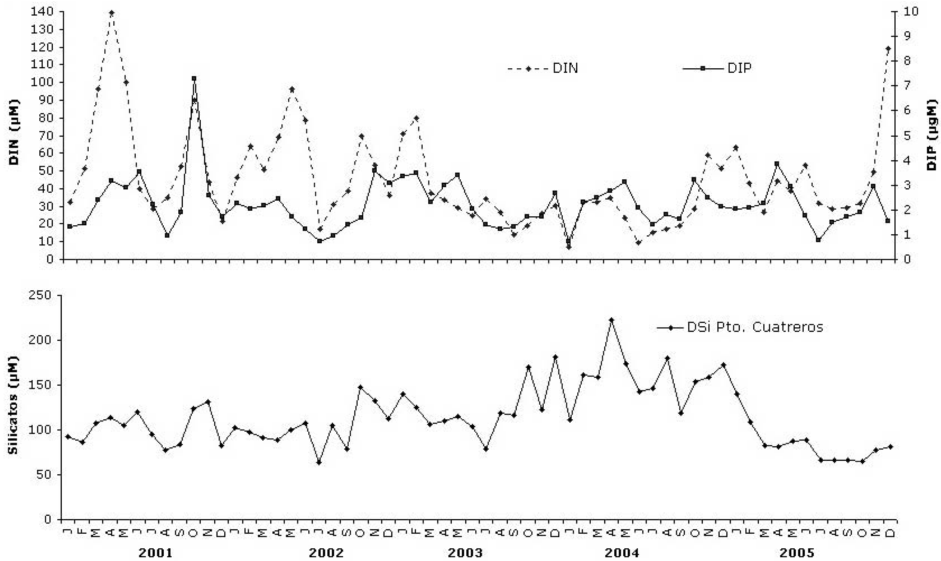

FIGURE 5: *Monthly distribution of DIN, DIP and DSi in surface estuarine water in Puerto Cuatreros. (Extracted from Spetter 2006, Spetter et al. 2008).*

The concentration of DSi showed minima in winter (104.1 $\pm$ 33.5 μM) and an increase in spring and summer (118.1 $\pm$ 33.0 μM) (Figure 5). In 2004, the highest concentrations of the whole studied period were observed (158.4 $\pm$ 28.9 μM) and they were related to the influence of the Sauce Chico River (956.0 $\pm$ 300.6 μM). During the year 2005 the concentrations of DSi were the lowest (84.4 $\pm$ 21.6 μM) of the whole analysed period. Those variations on the typical dynamics of the dissolved inorganic nutrients in surface estuarine water of Puerto Cuatreros are being connected with changes in the environmental factors such as lower precipitations and increase or decrease of temperature (Spetter et al. 2008). Studies carried out by Del Blanco (2007) show that the concentration of nutrients in the tidal flat of Puerto Cuatreros is not affected by the advance of tides.

During the years 2003 and 2004, the dynamics of dissolved inorganic nutrients in surface estuarine water (SEW) was analysed with respect to the observed dynamics in pore water (PW) (Figure 6) and the results showed that NO_2^- in SEW (1.30 $\pm$ 1.06 μM) was higher than in the PW (0.26 $\pm$ 0.19 μM); NO_3^- showed a similar trend (8.87 $\pm$ 6.46 μM and 0.99 $\pm$ 1.10 μM respectively). Unlike this, NH_4^+ presented an inverse trend, with higher levels in the PW (22.15 $\pm$ 13.50 μM) than in the SEW (14.37 $\pm$ 8.99 μM) (Spetter et al. 2007a). The nitrification, the biological process which transforms reduced forms of nitrogen to nitrate (Herbert 1999, Koops and Pommerening-Röser 2001), seemed to occur at the end of spring within sediments due to the presence of NO_3^- in PW. The ammonification, the release of NH_4^+ from the nitrogenous organic matter supplied to the sediments (Herbert 1999), would develop throughout the whole year, although nitrate reduction process would only be present in later summer (Spetter et al. 2008). Both processes take part in the stage of "recuperation" of nutrients described earlier.

In the case of DIP in surface estuarine water and porewater, it has not shown great differences (Figure 5 and 6) and its concentration in SEW was 1.99 $\pm$ 0.90 μM and in PW 1.57 $\pm$ 0.89 μM (Spetter 2006). With respect to the concentration of DSi, it was higher in porewater (207.9 $\pm$ 90.7 μM) than in SEW (144.9 $\pm$ 47.0 μM) (Figure 5 and 6). The comparison of the concentration of DSi in porewater extracted from the innermost fractions of sediment presented a marked tendency to increase with deeper sediments. It was suggested that this phenomenon is a consequence of the dissolution of biogenic silica in the sediments (Spetter 2006), according to Ehrenhauss et al. (2004).

2.7 Chlorophyll a

Simultaneously with the occurrence of these nutrient distribution processes within the estuary, the one for the corresponding photosynthetical pigments (i.e. chlorophyll a) have also been identified for this system (Figure 7). Thus, a clear decreasing trend of chlorophyll concentration from the inner area (mean value 10.77 $\pm$ 4.97 μg l^{-1} with values reaching up to 18 μg l^{-1} down to the outer one (mean value 3.19 $\pm$ 1.74 μg l^{-1}) has been observed (Figure 7). Nevertheless, it is important to highlight that the Chl-a levels within the estuary has never been null, which indicates that the system is a permanently productive estuary, and its lower values are similar to those usually recorded in coastal marine waters from the Argentine Sea (Marcovecchio 2000). In addition, during the last three decades Chl-a values of approximately 42 μg l^{-1} have been recorded in different years (Gayoso 1998, Popovich and Marcovecchio 2008). Simultaneously with this pigment distribution and concentrations, the values of net primary production (NPP) determined at the inner area of the estuary have reached up to $\sim$300 mg C m^{-3} h^{-1}, which could be mentioned as being among the highest records reviewed in the international literature (Marcovecchio and Freije 2004).

2.8 Particulate Organic Matter

The high primary production levels determine the occurrence of high concentrations of organic matter within the system (mean values $\sim$2000 mg C m^{-3} for both Pto.Cuatreros and Pto.Ing White), with the maximum levels coinciding with the peaks of chlorophyll a. The depletion of nitrate, nitrite and silicate together with the increase in dissolved oxygen levels, indicate that most of the determined POM originates from primary production. In addition, different organic matter sources occur within the system, including sewage outfalls, rivers and streams, etc., which could significantly modify the OM available stock for the system.

3 CONCLUDING COMMENTS

Bahía Blanca estuary is a very large system, whose functioning is clearly characterized by several processes that on the whole determine the extent of the biological production occurring there. The distribution of the structural parameters within the system is very stable, mainly in terms of temperature, pH and turbidity. Salinity presents a relative stability, though significant variations occur at the inner area, where it could alternatively be increased or decreased according to the season. The estuary is usually nutrient enriched, and compounds of nitrogen (especially ammonium) are always available, even though the concentration of oxidized nitrogen compounds (NO_2^- and NO_3^-) and phosphorus used to be fully depleted during the periodical phytoplankton blooms. In addition, high levels of silicate are usually available in the estuary, mainly in the inner area, which is roughly adequate to support the biological demand within the system.

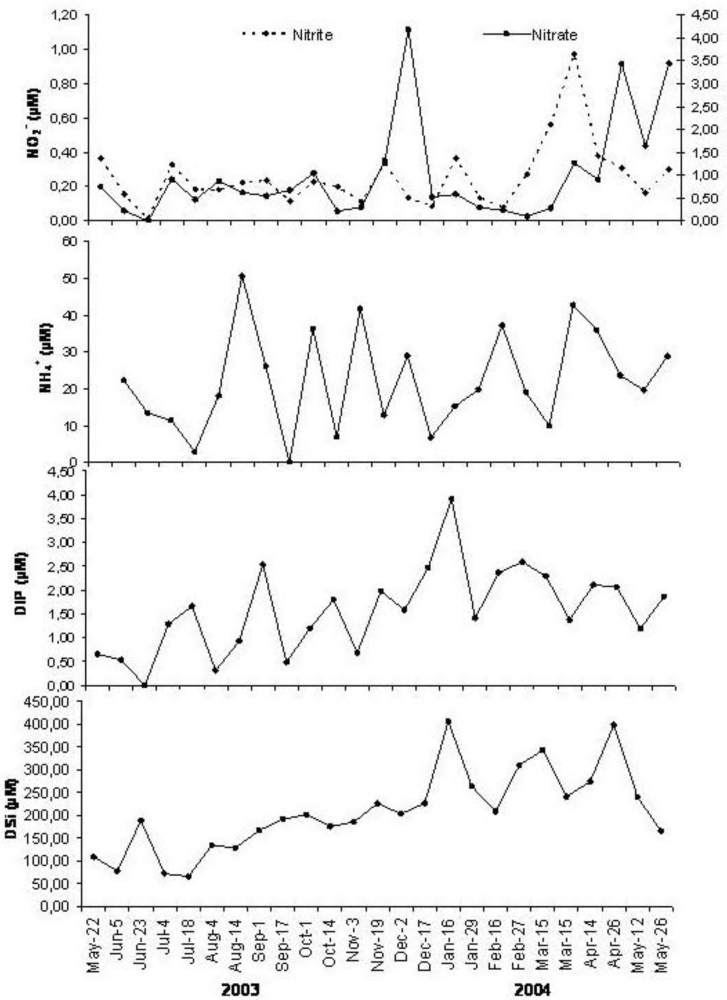

FIGURE 6: *Concentration of nitrite, nitrate, ammonium, DIP and DSi in porewater (PW) at Puerto Cuatreros during May 2003 - May 2004. All concentrations are in μM. (Extracted from Spetter 2006, Spetter et al. 2008).*

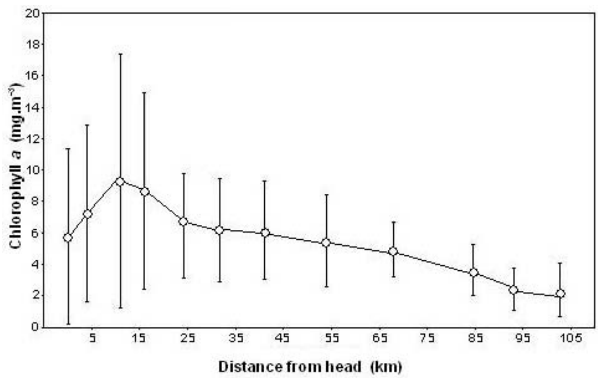

FIGURE 7: *Chl-a concentration along the Main Navigation Channel.*

Concerning primary production, the most important annual period is late winter - early spring, when the highest phytoplankton blooms have historically occurred. It is developed so, because during this time the nutrients (N, P and Si) are largely available, and both the temperature and light intensity are sufficiently low ($\sim$5 to 7 °C, and 400-700 μE m^{-2} s^{-1} respectively; after Popovich et al., this volume) as required by the dominant diatom species (*Thalassiosira curviseriata*) responsible for this bloom. Thus, very high levels of Chl-a were detected during this phenomenon (with values reaching up to 55 mg m^{-3}), representing densities of $\sim$13x 10^6 cells l^{-1} or net primary production of $\sim$300 mg C m^{-3} h^{-1}.

The very high amounts of organic matter generated by these biological processes ensure the regenerated nutrients production, through mineralization processes occurring within the estuary (Spetter 2006). The obtained results seemed to indicate that a predominant liberation of ammonium was observed from the estuarine sediments in the inner estuary, even significant amounts of oxidized nitrogen compounds (NO_2^- and NO_3^-) were eventually also produced (Spetter 2006). In addition, these are the first nitrogen nutrients to be consumed during the phytoplankton bloom, and just when both NO_2^- and NO_3^- were depleted the NH_4^+ started to be consumed (Popovich et al. 2008). This productive cycle, regulated through bio-geochemical joint processes has functioned well for a long time (at least during the past 30 years, when these studies first started). Consequently, this is a very good scenario to control the evolution and progress of the estuary chemical processes, as well as to monitor the potential occurrence of changes within the identified trends.

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COMPOSITION AND DYNAMICS OF PHYTOPLANKTON AND ALORICATE CILIATE COMMUNITIES IN THE BAHÍA BLANCA ESTUARY

C.A. POPOVICH, V.A. GUINDER AND R.E. PETTIGROSSO

1 PHYTOPLANKTON

1.1 Introduction

Estuaries are extremely heterogeneous systems where physical, chemical, geological and biological factors interact closely. These heterogeneous conditions cause seasonal and spatial variation in the estuarine environment and therefore on phytoplankton dynamics. Short-term variability (daily, monthly, seasonal) and long-term variability (annual, decadal) in phytoplankton community structure may be induced by tides, light availability, stratification, advection, precipitation, nutrient loading, zooplankton grazing as well as climatic changes, among others (e.g. Cloern 1996, Smayda 1998). In addition, the floristic composition of estuarine phytoplankton communities varies according to biogeographical aspects on both regional and local scales.

Documenting phytoplankton dynamics is an essential step in assessing the role of estuaries in biogeochemical cycles, as well as in determining the long-term responses of coastal ecosystems to anthropogenic stress. One of the most relevant features of long-time phytoplankton studies is that they allow the detection of potential changes in the successional annual pattern and interannual variations. Water assessments of the inner zone of the Bahía Blanca estuary are available for a 28 year period (1978-2007). This data set (probably the most extensive continuous record from South America) describes the variations and changes observed in phytoplankton biomass, abundance, composition, and environmental variables. Specifically, the phytoplankton abundance and composition were studied during the periods: 1978-1981, 1988-1994, 2002-2003 and 2006-2007. Besides this intensive long-term monitoring at the head of the estuary, other studies have been carried out along the longitudinal gradient of the Principal Channel (Popovich 1997, Guinder et al. 2007c, Popovich and Marcovecchio 2008), in the outer zone of the estuary (Sabatini 1982) and within the inner shelf offshore from the Bahía Blanca estuary (Gayoso et al. 1994).

This chapter summarizes all the available historical information about the phytoplankton community in the Bahía Blanca estuary, with emphasis on timing, magnitude, triggering and composition of the phytoplankton winter-early spring bloom, in order to evaluate their role as a key event in this environment. It is recognized that phytoplankton composition is a natural bioindicator because of its complex and rapid responses to fluctuations of environmental conditions. The long-term monitoring of dominant species and their relationships with environmental conditions in the Bahía Blanca estuary provides motivation to investigate many questions: Can changing bloom dynamics (frequency, duration, magnitude, species composition) represent a tool to evaluate an element of anthropogenic impact? Are the changes observed since 2000 in the phytoplankton community an effect of the increasing human activity in the Bahía Blanca estuary? A 28-yr period is a time scale over which we must separate anthropogenic influences from natural variability. Thus, in this context, this chapter provides an ecological baseline to evaluate changes in the water quality in the Bahía Blanca estuary.

1.2 Seasonal patterns of the most representative species

Phytoplankton seasonal patterns of temperate coastal environments are characterized by great variability, but they usually include major blooms during spring and summer and minor ones in early autumn (Smayda 1980, Hallegraeff and Jeffrey 1993). In the Bahía Blanca estuary, the occurrence of a winter-early spring diatom bloom is the most important event in the phytoplankton annual cycle (Figure 1). Similar behaviours have been observed in Narragensett Bay, USA (Pratt 1965), Peel-Harvey estuary, Australia (McComb et al. 1981) and Málaga Bay, Spain (Jiménez et al. 1987).

The genus *Thalassiosira* is the most conspicuous component of phytoplankton in the area. The species described are: *Thalassiosira curviseriata* Takano (Figure 2, a and b), *T. anguste-lineata* Fryxell et Hasle, *T. pacifica* Gran et Angst (Figure 2, e, f and g), *T. rotula* Meunier, *T. eccentrica* (Ehrenberg) Cleve (Figure 2, h), *T. hibernalis* Gayoso, *T. hendeyi* Hasle et Fryxell, and *T. minima* Gaarder (Figure 3, a and b) (Gayoso 1981a 1989, Popovich 1997). *T. curviseriata* is clearly the most abundant species in the phytoplankton annual cycle (Popovich and Gayoso 1999, Popovich et al. 2008); it is observed year-round with a very strong peak in winter. This species accounted for 60-90% of the total number of cells in the annual bloom (Figures 4 and 5). Its greatest densities vary from 2.8×10^6 cells l^{-1} to 12.7×10^6 cells l^{-1}. Other abundant species during the winter bloom are: *T. anguste-lineata, T. rotula, T. pacifica, T. eccentrica* and *T. hibernalis* (Figure 4). *Thalassiosira minima* is usually more abundant during spring and summer and *T. hendeyi* appears almost all year-round but does not bloom.

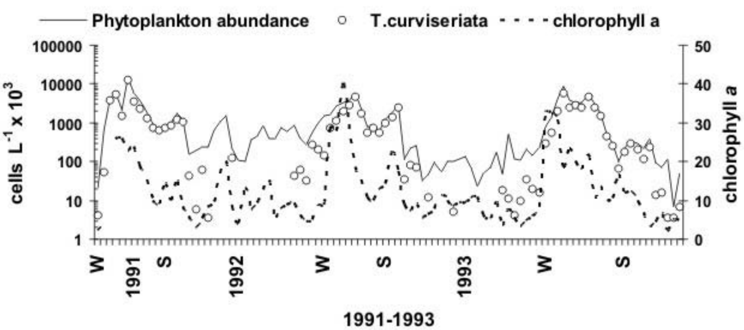

FIGURE 1: *Seasonal variation of the phytoplankton and* Thalassiosira curviseriata *abundance and Chl-a (μg l^{-1}) in the Bahía Blanca estuary during 1991-1993. W (winter) and S (spring). (Graph adapted from Popovich and Gayoso 1999)*

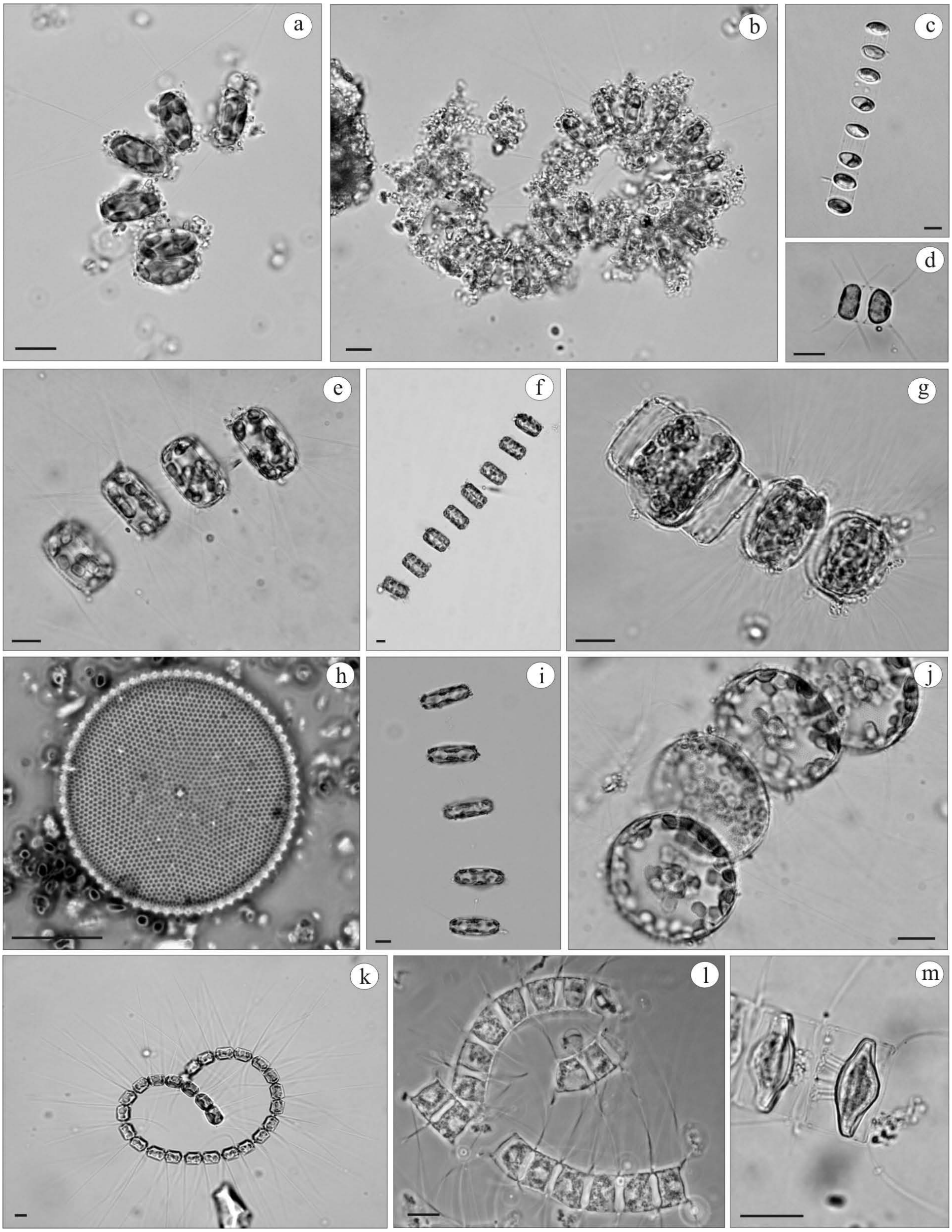

FIGURE 2: *Blooming species: Thalassiosira curviseriata (a and b), Skeletonema costatum (c), Chaetoceros ceratosporus (d), Thalassiosira pacifica (e, f and g), T. eccentrica (h) Thalassiosira sp. (i and j), Chaetoceros debilis (k and l) and C. diadema with resting spores (m). Scale bar = 10 μm.*

Chaetoceros Ehrenberg, is the second most important genus in the annual cycle in terms of cell density and number of species (Figure 5). The species described in the area are: *Chaetoceros ceratosporus* var. *brachysetus* Rines et Hargraves (Figure 2, d) , *C. debilis* Cleve (Figure 2, k and l), *C. diadema* (Ehrenberg) Gran (Figure 2, m), *C. similis* Cleve, *C. subtilis* Cleve and its variety *C. subtilis* var. *abnormis* Proschinka-Lavrenko (Gayoso 1988, 1999) and a small *Chaetoceros* sp. characterized by delicate setae. *Chaetoceros subtilis* tends to be present in summer and autumn whereas the other *Chaetoceros* spp. are frequent during the winter bloom.

Skeletonema costatum (Greville) Cleve (Figure 2, c) and *Asterionellopsis glacialis* (Castracane) Round are of secondary importance during the winter bloom, however their occurrence shows conspicuous interannual variations. *Ditylum brightwelli* (West) Grunow (Figure 3, d) is very variable in abundance throughout the year whereas *Leptocylindrus minimus* Gran (Figure 2, e), *Guinardia delicatula* (Cleve) Hasle (Figure 3, f) and *Cerataulina pelagica* (Cleve) Hendey tend to become dominant in spring and summer. T*halassiosira hendeyi* and *Paralia sulcata* (Ehrenberg) Cleve (Figure 3, g and h) and the tychopelagic species *Cylindrotheca closterium* (Ehrenberg) Reimann and Lewin (Figure 3, j) are present most of the year. The tychopelagic species *Gyrosigma attenuatum* (Kütz.) Rab. is present in the water column associated with turbulence processes.

Small phytoflagellates (10-20 μm) are present throughout the year with maximum abundance during summer. The dinoflagellates *Scrippsiella trochoidea* (Stein) Loeblich III (Figure 3, l), *Prorocentrum* sp. and *Protoperidinium punctulatum* (Pauls) Balech, are important in late-spring and early summer. The abundance of *S. trochoidea* ranges between 80x 10^3 cells l^{-1} (Popovich et al. 2008) and 2.7x 10^6 cells l^{-1} (Gayoso 1999). During the summer unidentified species of gymnodinians and Cryptophyceae are dominant with a maximum peak up to 1.5x 10^6 cells l^{-1} (Popovich 1997, Gayoso 1999). Conspicuous interannual variations in the abundance of phytoflagellates were found in the estuary. Some aspects of these groups, such as biomass and zooplankton relationship, have been considered by Sabatini (1987). The Xantophyceae *Ophiocytium* sp. is an important phytoplankter in the estuary during spring when it reaches densities up to 10^5 cells l^{-1}.

1.3 Winter-spring diatom bloom: timing, magnitude, trigger and succession

Weekly sampling at the fixed station over a long time period has shown that the typical winter bloom in the Bahía Blanca estuary usually begins in July and finishes in early September. Maximum abundance has ranged between 3x 10^6 cells l^{-1}1 and 12.74x 10^6 cells l^{-1} with maximal Chl-a values ranging up to 54 μg l^{-1} and primary production determinations reaching up to 300 mg C m^{-3} d^{-1} (Freije and Gayoso 1988). Factors regulating bloom inception and decline show seasonal and interannual variations. In the Bahía Blanca estuary a decrease of temperature, high nutrient concentrations in autumn (Popovich 1997, Popovich et al. 2008) and a relaxation in zooplankton grazing pressure may indeed trigger the inception of the bloom.

On the other hand, the collapse of the winter-spring bloom in August or September appears to result from a combination of intensive copepod grazing and nutrient limitation (Popovich et al. 2008).

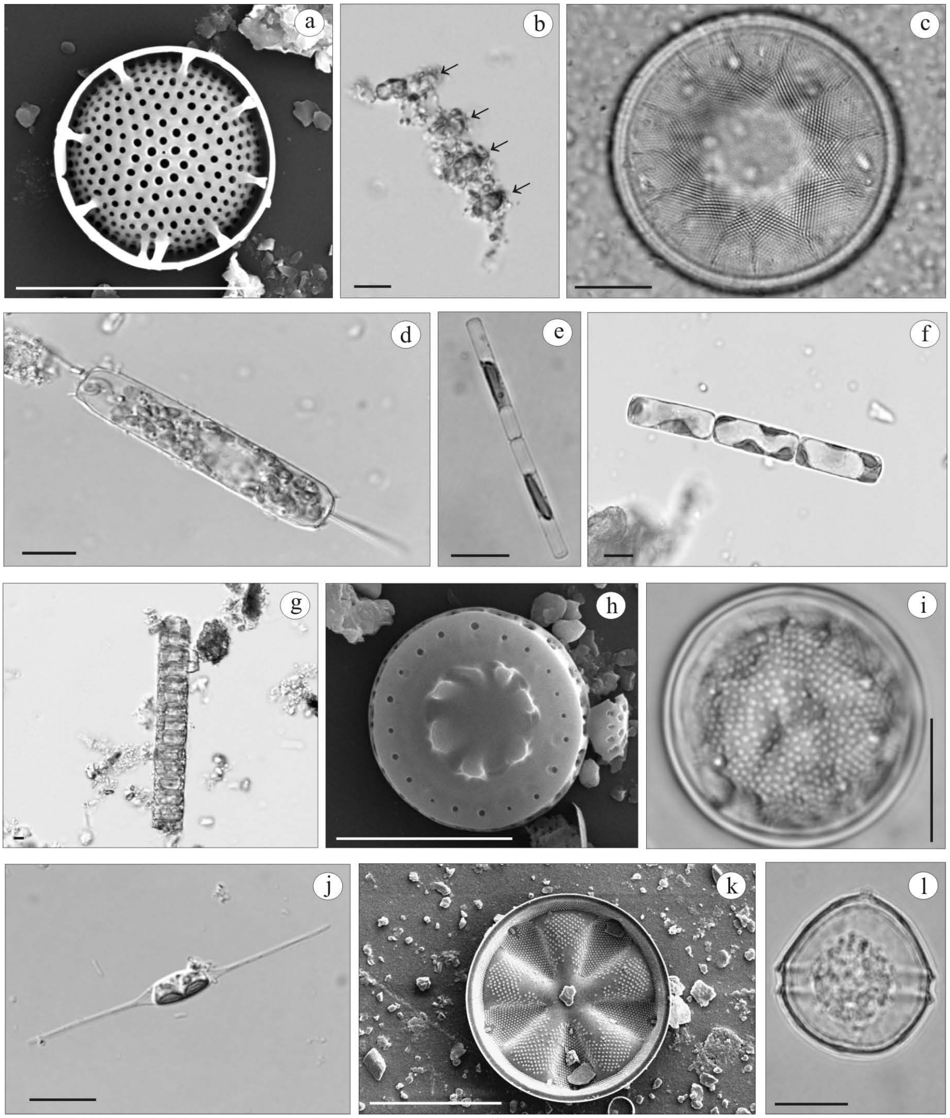

FIGURE 3: *Non blooming species. Thalassiosira minima (a and b), Podosira stelliger (c), Dytilum brightwellii (d), Leptocylindrus minimus (e), Guinardia delicatula (f), Paralia sulcata (g and h), Actinocyclus sp. (i), Cylindrotheca closterium (j), Actynoptychus adriaticus (k), Scrippsiella trochoidea (j). Scale bar = 10 μm.*

At the species level, the phytoplankton community is dominated by an assemblage of blooming species (*Thalassiosira* spp. and *Chaetoceros* spp.) in winter, which show a recurrent annual pattern (Figure 4). Some non-blooming species (other diatoms and dinoflagellates) also appear during the blooming season but they are less recurrent and show large interannual variations. According to Popovich et al. (2008), the early phase of the winter bloom is characterized by a marked dominance of *T. curviseriata* (i.e relative dominance $>$60%), whereas the later phase is characterized by the coexistence (i.e. single-species relative dominance $<$35%) of different blooming species.

Biogeographical considerations of the most important species have been discussed by Popovich and Gayoso (1999) and Gayoso (1999) who state that the assembly of Thalassiosira species characteristic of the winter-early spring bloom in the Bahía Blanca estuary differs from those found in other temperate estuaries. In Narragansett Bay, the winter-spring diatom bloom was dominated by *Thalassiosira nordenskioeldii* and *Detonula confervaceae* (Smayda 1980); in a Scottish sea-loch, the spring bloom was caused by *Thalassiosira angulata*, *T. anguste-lineata, T. gravida, T. nordenskioeldii* and *T. rotula* (Harris et al. 1995). *T. curviseriata*, the dominant species in the Bahía Blanca estuary, has a distribution restricted to eutrophic coastal waters. It was found in the Bay of Tokyo in Japan, associated with diatom blooms and water discoloration process (Takano 1981); and in coastal waters of Australia (Hallegraeff 1984) and Skagerrak, North Atlantic (Lange et al. 1992).

1.4 Relationships between winter diatom bloom and environmental factors

Spatial and temporal studies along the longitudinal gradient of the Principal Channel of the Bahía Blanca estuary (80 km) showed that Chl-a concentrations, phytoplankton abundance and nutrient levels tend to decrease from the head to the mouth of the estuary (Popovich 1997, Guinder et al. 2007c, Popovich and Marcovecchio 2008). Indirect evidence (e.g. higher levels of biomass and favourable growth conditions, such as elevated euphotic to mixing depth ratios and nutrient levels) supports the idea that the most important primary production in the estuary occurs in the inner zone during the winter diatom bloom. Thus, knowledge of the regulatory mechanisms of the winter bloom is a necessary step to predict possible changes in the water quality in the inner zone.

The seasonal pattern and interannual variations of phytoplankton are currently interpreted as being the result of changing environmental conditions, considered as selective factors on different species assemblages. In nutrient-rich and highly turbid systems, such as the Bahía Blanca estuary, the phytoplankton growth rate is usually limited by light. Interestingly, the start of the bloom, usually during June-July, coincides with the minimum annual temperatures (around 5 °C), shortest daylengths (9h light, 15 h dark) and lowest incident irradiances (400-700 μE m^{-2} s^{-1}). As previously mentioned, *T. curviseriata* is a key species in the winter-early spring diatom bloom (Popovich and Gayoso 1999). Therefore, its behaviour can be used, for instance, to analyze bloom dynamics and environmental factor relationships.

The increase in abundance of this species in the inner zone of the estuary was negatively correlated with temperature (Popovich 1997). Autoecological studies of *T. curviseriata* showed that this species grows well at relatively low temperatures (up to 1 doubling per day at 5-10 °C) and relatively low light intensity (maximum rate of growth between 70 and 80 *mu*E m^{-2} s^{-1}), its optimal range of growth being between 5 to 20 °C and 25 to 40 salinity values (Popovich and Gayoso 1999). These characteristics may explain the ability of this species to flourish during winter when light and temperature conditions are apparently limiting, and its presence almost all year round under variable temperature and salinity conditions. The species response to temperature could be a reference for detecting the effects of long-term climatological changes on phytoplankton dynamics.

Phytoplankton blooms in estuaries have been recorded as being associated with increasing nutrient loads. The inner zone of the Bahía Blanca estuary is considered to be highly eutrophic (Marcovecchio and Freije 2004). The Sauce Chico River, at the head of the Bahía Blanca estuary, appears to be an important potential source of N for new production (*sensu*, Dugdale and Goering 1967). In addition, nutrient regeneration processes resulting from water-sediment interactions, increase of zooplankton activity during the post-bloom period (Hoffmeyer 1994, Pettigrosso et al. 1997) and presumably other factors, such as adjacent salt marshes, may contribute to the nutrient increase that occurs during spring and summer.

The rates and pathways through which blooms act as agents of geochemical change are highly dependent on the species abundance and composition of the bloom communities (Cloern 1996). In the Bahía Blanca estuary during the winter bloom, high levels of phytoplankton biomass were associated with depletion of nitrate, as well as minimum values of phosphate and silicate. The increase of biomass (Chl-a up to 44.3 μg l^{-1}), which characterized the beginning of the bloom in 2002, was almost entirely due to *T. curviseriata* (64.5-81.8%). Its growth, mainly concurrent with dissolved inorganic nitrogen (DIN) decrease, was related to new production in this area. According to the observed trends of the DIN sources, preferential utilization of the nitrogen pool components might occur gradually, shifting the utilization of nitrate to ammonium. This unusual situation has been seen in many studies on nitrate uptake by phytoplankton at high ammonium levels, especially from coastal and estuarine environments (Comin and Valiela 1993, Kudela and Dugdale 2000).

Although this mechanism has not been fully understood, both cultivated and natural phytoplankton communities have shown an optimum nitrate uptake under low light conditions (Leonardos and Geider 2004). On the other hand, the observations in the Bahía Blanca estuary suggest that low phosphate concentrations contributed to the diatom bloom collapse more than DIN and silicate. Coincidently, a field-research from 1991 to 1993 reported a significant negative coefficient of a multiple regression analysis for phosphate concentrations and *T. curviseriata* (Popovich 1997). Phosphate limitation has already been reported in many coastal and estuarine systems (Benitez-Nelson 2000) and its largest effect was linked to the limitation of winter and spring production (Conley 2000 and references therein). The evidence from these studies strongly suggests that *T. curviseriata* plays an important role in the flux of dissolved inorganic nutrients in the inner zone of the estuary.

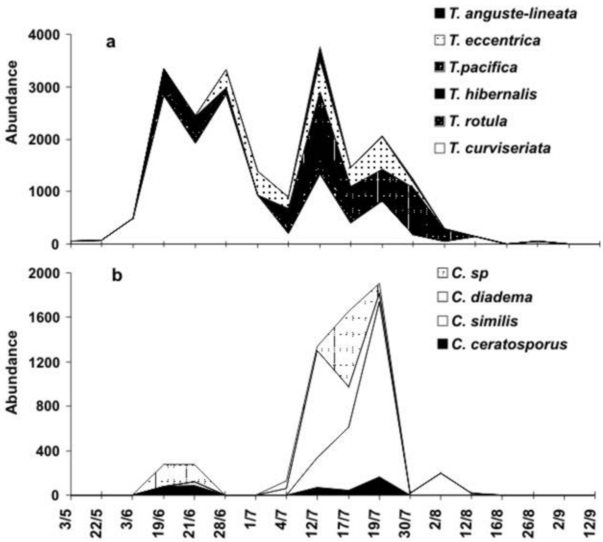

FIGURE 4: *Winter bloom succession during 2002. Abundance (cells x* 10^3 l^{-1}*) of Thalassiosira spp. (a) and Chaetoceros spp (b). (Graph adapted from Popovich et al. 2008).*

1.5 Changes in phytoplankton community structure and seasonality

The winter-early spring diatom bloom in the Bahía Blanca estuary has recurred throughout the 1978-2002 period and in 2007 (Gayoso, 1999, Popovich et al. 2007). Changes in the seasonal succession pattern of phytoplankton have been observed from 2003 until the present (Popovich et al. 2007, Guinder et al. 2007b). The main discrepancies in the seasonal pattern that make the period 2003-2007 conspicuously abnormal are: 1) the diminution or even absence of the winter-early spring diatom bloom, 2) a clear trend of decreasing abundance or even the disappearance of some of the typical blooming species, 3) the occurrence of short biomass peaks throughout the year and 4) the presence of a few uncommon diatom species in the system that become most abundant in summer and autumn (Guinder et al. 2006, 2007b).

In 2006 the absence of the winter-spring bloom (Guinder et al. 2006, 2007a, b) was responsible for a low annual mean Chl-a concentration (5.8 μg l^{-1}) compared to previous years which were characterized by a uni-modal phytoplankton pattern. During the past five years *T. curviseriata* was no longer the dominant species in winter as it had been for more than two decades. Moreover, it used to be a main part of the planktonic assemblage year-round, but since 2003 it has only appeared in short discrete periods and in low abundance (72.7x 10^4 cells l^{-1} in June 2006). The abundance of other diatoms, considered as blooming species in the estuary, was negligible or even absent, e.g. *Thalassiosira hibernalis, T. rotula, T. anguste-lineata* and *Chaetoceros similis*. Also, some non blooming species that used to be present in moderate abundance during the winter season (e.g. *A. glacialis*) have almost disappeared.

Conspicuous shifts in the seasonal pattern of phytoplankton were recorded in 2006, since biomass peaks occurred in mid summer, late autumn-early winter and late winter. These three abundance peaks were of similar magnitude and duration (Figure 5), but they were characterized by different species composition. The episodes of chlorophyll increase were also episodes of increasing phytoplankton cell number and were restricted to the upper part of the estuary (Guinder et al. 2007b, 2007c). The increasing frequency of phytoplankton peaks has been associated with nutrient enrichment in shallow estuaries and coastal waters (Cloern 2001). In recent years, the summer peak has been composed of diatom species that seem to gradually reach greater magnitudes year after year (Popovich et al. 2007, Guinder et al. 2007b). Moreover, in 2005 the annual maximum chlorophyll (20.94 μg l^{-1}) was in summer rather than in winter (12.21 μg l^{-1}) (Popovich et al. 2007).

For the past five years some novel species have become frequent in the estuary all year-round and have been responsible for the maximum biomass in summer and autumn. These diatoms are *Cyclotella* spp. (6-15 μm), *Thalassiosira* sp. (9-36 μm) (Figure 2, i and j), *T. minima* (Figure 3, a and b) and *Minidiscus chilensis*. Similar novel appearances followed by persistence, perennial predominance, or even establishment as keystone species, have been increasingly documented in other environments (Smayda 1998). The intensive proliferation and elevated biomass reached by these species in 2006 could be due to the absence or low densities of the typically dominant ones (e.g. *T. curviseriata, T. rotula, T. anguste-lineata, T. hibernalis*), which were probably competitively superior. In the Bahía Blanca estuary, interannual variability has been associated with changes in the relative abundances of some species, such as in 1981 and 1990 (Gayoso 1998, 1999), but the typical blooming species were also present in those years. For instance, the seasonal pattern of phytoplankton that has been observed since 2003 is totally abnormal compared with historical data.

For the past five years the seasonal behaviour of dissolved nutrients also has deviated from the characteristic pattern (Popovich et al. 2007). On the other hand, during the past three decades a gradual increase in surface water temperature has been perceived (Marcovecchio and Freije 2004). As previously explained, the start of the winter bloom takes place when the water temperature is markedly diminished and remains around 5 ℃ for a relatively long period. The minimum temperature recorded in winter 2006 was 7.6 ℃, while it was over 8℃ during the rest of the year (Guinder et al. 2007b). The cooler water seems to not only trigger the bloom inception and its permanence for more than two months but also promotes the growth of the typical diatom blooming species. Considering this, there is certainly strong evidence to suggest that large structural changes in phytoplankton seasonality could be explained by the mild winters and the consequently higher water temperatures.

Another factor that could be regulating the annual phytoplankton dynamics in the estuary is grazing by the mesozooplankton, which is usually lower in winter (Hoffmeyer 2004, Barría de Cao et al. 2005). The possible effect of grazing as a changing mechanism is uncertain, but it might be of some importance, taking into account the level of secondary production in the system (Pettigrosso et al. 1997). Quantitative studies of the phytoplankton community response to these variables are needed to interpret the observed community changes.

As well as internal forces of biological, chemical and physical variability, there might be external events (e.g. meteorological, climatological) that drive changes in the phytoplankton community structure in the Bahía Blanca estuary. Further studies should be carried out in order to ascertain which are the principal factors regulating the phytoplankton seasonality. The last five years might just have been an atypical period for the Bahía Blanca estuary or they could be indicating the beginning of gradual and directional changes that are occurring in the system. This temperate estuary is the only one in South America that has multidecadal data, so as well as being locally relevant, it also provides an invaluable database for relating conspicuous ecosystem shifts with trends in atmospheric oscillations or anthropogenic perturbations.

2 PLANKTONIC CILIATES

2.1 Introduction

Planktonic ciliates constitute a diverse group from a trophic and taxonomic point of view. They are often dominant components of the microzooplankton community in the 20-200 μm size range. They are key grazers within the ”microbial loop” which is responsible for the rapid remineralization of organic matter in the water column (Finlay 2001) and they play a key role in transferring energy from primary production to higher trophic levels. Some planktonic ciliates, such as tintinnids, have a lorica constructed by themselves, but they are only a small percentage of the entire ciliate biota. The most common ciliates are ”aloricate” forms, without a lorica. Although aloricate planktonic ciliates are usually found in high numbers in coastal waters (Pierce and Turner 1992), their distribution, taxonomic composition and ecological characteristics in estuaries in the South Atlantic Ocean, and Argentina in particular, are poorly known (Pettigrosso 2001, 2003, Pettigrosso et al. 1997, Pettigrosso and Barría de Cao 2004). The study of species composition and seasonal variation of the aloricate planktonic ciliates from the Bahía Blanca estuary was carried out in Puerto Cuatreros which was considered representative of the inner zone of the estuary and was characterised by variable temperature and salinity. The objective of this section is to examine information concerning the taxonomy and ecology of the aloricate planktonic ciliates, with emphasis on the phytoplankton bloom, in order to evaluate their role as a link between phytoplankton and higher trophic levels.

2.2 Species composition and geographical distribution

The Protargol staining technique, following Tuffau (1967), Lee et al. (1985) and Foissner (1991) was used to determine the taxonomic composition. Most of the ciliates recorded in Puerto Cuatreros belonged to the orders: *Choreotrichida* Small & Lynn and *Strombidiida* Petz & Foissner. These groups are commonly found in other coastal areas of the world with a similar range of temperature and salinity (Pettigrosso 2003) (Table 1). However, the same species, e.g. *Strombidium emergens*, have also been recorded at geographically different sites and quite different water temperatures (Weddell Sea; Gulf of Maine, USA and Puerto Cuatreros) (Pettigrosso 2003). Altogether, seven species of aloricate planktonic ciliates were recorded: *Strombidinopsis elongata* Song & Bradbury, *S. epacrum* Lynn & Montagnes, *S. capitatum* (Leegaard) Kahl, *S. acutum* Leegaard, *S. emergens* (Leegaard) Kahl, *S. dalum* Lynn, Montagnes & Small and *Cyrtostrombidium longisomum* Lynn & Gilron (Table 1) and they are all new records for the coastal waters of Argentina (Pettigrosso 2003).

2.3 Aloricate planktonic ciliates during the phytoplankton bloom

The studies carried out on aloricate planktonic ciliates during the phytoplankton bloom from 10 June to 22 September 1994, showed the presence of two peaks of total ciliate biomass and two peaks of total Chl-a (Figure 6). More than 60% of the first peak of ciliate biomass was formed by individuals of 103-104 μm^3 in size and was observed three weeks after the Chl-a peak corresponding to the <25 μm phytoplankton fraction. The second peak of ciliate biomass was produced at a low abundance of the largest species. It was constituted principally of ciliates 105 and >105 μm^3 in size and occurred two weeks after the second peak of Chl-a, which was formed by the >25 μm phytoplankton fraction (Pettigrosso et al. 1997).

The correlation test performed between ciliates of different (volume) size classes and the two fractions of Chl-a revealed that the <25 μ mchlorophyll a fraction showed a significant positive correlation with all size classes of the aloricate ciliates, except the smallest ones. This suggests that —although a correlation analysis cannot prove a cause-effect relationship —the smallest fraction of Chl-a was the main food source of the ciliates during the bloom period; however, their grazing impact on the phytoplankton was not relevant. The peaks of the ciliate biomass were mostly due to a reduction of the grazing pressure on the largest ciliates (top-down control) at high phytoplankton abundance, as occurs during the bloom period (Pettigrosso et al. 1997).

2.4 Abundance and species composition of aloricate planktonic ciliates from the wastewater discharge zone in the Bahía Blanca estuary

One of the main sources of organic pollution of coastal waters in the world is domestic sewage. The subject of planktonic ciliates in relation to organic pollution has not been studied in Argentine coasts, with the exception of the Río de La Plata coast (Rives 1997). In this context the variation in composition and abundance of the aloricate planktonic ciliate community in the wastewater discharge zone of the Bahía Blanca estuary was carried out in relation to physicochemical and biochemical parameters from 16 June 1995 to 20 May 1996. In order to compare results, sampling was carried out at two fixed stations, one in the wastewater discharge zone (WWDS) and the other (Boya 31) in the Principal Channel (Barria de Cao et al. 2003).

The aloricate ciliates recorded at the two stations were represented by the genera: *Strombidium, Strombidinopsis, Cyrtostrombidium, Strobilidium, Lohmmanniella* Leegaard 1915 and *Tontonia* Fauré-Fremiet. In general, the total abundance of aloricate ciliates was higher in Boya 31 than in WWDS and showed significant positive correlation with salinity and negative correlation with POM and phosphate. *Strombidinopsis* and *Strombidium* were the most conspicuous genera at both stations indicating a certain tolerance to high levels of organic matter and salinity variations. Meanwhile *Strobilidium, Lohmmanniella*, and *Tontonia* also appear to be tolerant even though they were represented by a smaller abundance. The exception was *Cyrtostrombidium* genus which was 50% lower in number at WWDS, demonstrating a certain degree of sensitivity to high organic matter levels and low salinity values.

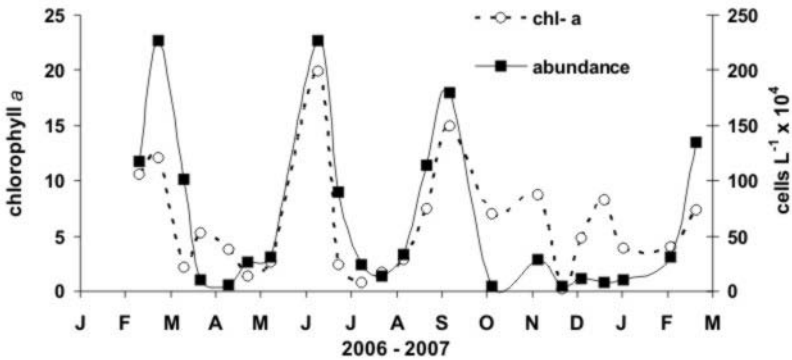

FIGURE 5: *Seasonal variation of phytoplankton abundance and Chl-a (μg l^{-1}) in the Bahía Blanca estuary during 2006-2007 (Graph adapted from Guinder et al. 2006).*

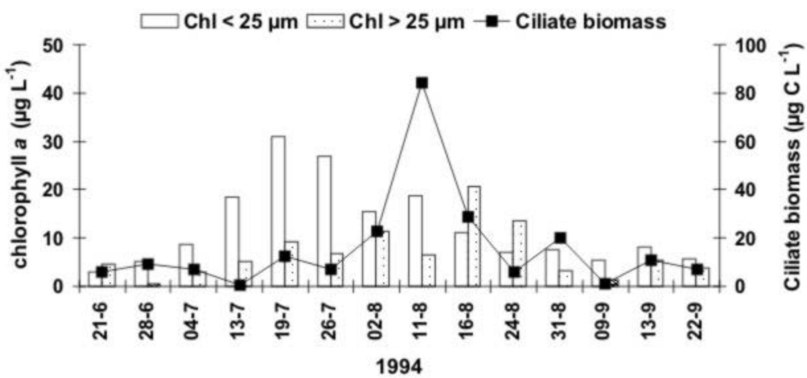

FIGURE 6: *Biomass of phytoplankton (μg Chlal^{-1}) and aloricate ciliates (μg C l^{-1}) during the 1994 phytoplankton bloom.*

3 CONCLUSIONS AND FUTURE WORK

Phytoplankton is an excellent indicator of ecological change. The structure of the phytoplankton community and related environmental factors has been observed in the Bahía Blanca estuary since 1978. The long-term patterns of the annual (winter-early spring) diatom bloom behaviour, the key species and their environmental tolerance ranges have been identified. A modest increasing trend has been observed in phytoplankton abundance and composition since 2000. Secondary diatom blooms during summer-autumn have been more frequent. In addition, both a lower magnitude and duration of the winter diatom bloom has been recorded.

The specific causes of these changes in the bloom dynamics have not been established; however it is possible to hypothesize that changes in nutrient dynamics and environmental conditions, particularly temperature, may play major roles in the observed changes in the phytoplankton. The dynamic nature of the phytoplankton coupled with multiple scenarios (temperature increase, nutrient input, anthropogenic influences, light availability, level of suspended sediment) and the resultant impacts on biogeochemical and habitat conditions, trophic state and food web structure should be studied in a prognostic model in order to evaluate and help manage ecological change in this coastal ecosystem.

TABLE 1: *Species composition, occurrence and geographical distribution of aloricate planktonic ciliates from the Bahía Blanca estuary, Argentina. (1) (Song and Bradbury 1998), (2) (Pettigrosso, 2003), (3) (Lynn and Montagnes, 1988), (4) (Montagnes et al. 1988), (5) (Lynn et al. 1988), (6) (Lynn and Gilron 1993), (7) (Leakey et al. 1994).*

Species	Occurrence	Salinity	TºC	Geographic distribution
Strombidinopsis elongate	Winter	29.6-35.7	8.8 -11	(1) Yellow Sea, China (2) Puerto Cuatreros, Bahía Blanca estuary, Argentina.
Strobilidium epacrum	Spring	30-38	12 -18	(3) Isles of Shoals, Gulf of Maine. (2) Puerto Cuatreros, Bahía Blanca estuary, Argentina.
Strombidium capitatum	Spring-Summer	29-30	12-21	(4) Great Harbor, Woods Hole, Massachusetts, USA. (2) Puerto Cuatreros, Bahía Blanca estuary, Argentina.
Strombidium acutum	Spring-Summer	30-35	12-19	(3) Isles of Shoals, Gulf of Maine, USA. (5) Perch Pond, Falmouth, Massachusetts, USA (2) Puerto Cuatreros, Bahía Blanca estuary, Argentina.
Strombidium emergens	Winter	30-33	9-12	(3) Isles of Shoals, Gulf of Maine, USA. (5) Weddell Sea (69º46'-70º21'S, 08º53'-11º00'W). (2) Puerto Cuatreros, Bahía Blanca estuary, Argentina.
Strombidium dalum	Spring	30-31	9-13	(3)Isles of Shoals, Gulf of Maine, USA. (6) Coastal waters near Kingston Harbour, Jamaica; (2) Puerto Cuatreros, Bahía Blanca estuary, Argentina.
Cyrtostrombidium longisomum	Spring	29-35.5	11-22	(6) Coastal waters near Kingston Harbour, Jamaica. (7) Plymouth Sound, Southampton waters. (2) Puerto Cuatreros, Bahía Blanca estuary, Argentina.

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COMPOSITION AND DYNAMICS OF MESOZOOPLANKTON ASSEMBLAGES IN THE BAHÍA BLANCA ESTUARY

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1 INTRODUCTION

In addition to its role in the cycling of nutrients and pollutants in estuaries and other aquatic ecosystems, zooplankton produces a large fraction of the total secondary production supporting the higher trophic levels. This community thus constitutes the structural and functional link between primary producers and the larger benthic and pelagic organisms (Cushing 1975). In estuaries, these heterotrophic organisms are successfully adapted to highly variable conditions and tolerate a wide range of environmental stressors. Estuarine zooplankton is also much more tolerant to higher perturbation levels (pollution by toxic chemicals, dredging, eutrophication) than that from continental shelf waters (Uye et al. 1992, Esteves et al. 1997).

Research on the biology, ecology and biodiversity of mesozooplankton assemblages in the Bahía Blanca estuary dates back to approximately 30 years ago, providing a large database of accumulated information (Guerrero et al. 1976, Hoffmeyer 1983, 1986, 1990, 1994, Mianzan and Sabatini 1985, Sabatini 1988, 1989, 1990, Hoffmeyer and Prado Figueroa 1997, Hoffmeyer and Torres 2001, Piccolo and Hoffmeyer 2004). More studies have been carried out on zooplankton of the Main Channel and the inner zone of the Bahía Blanca estuary than of the outer zone. The inner zone is the area that has suffered most modification as a consequence of a combination of anthropogenic pressures including those of an industrial nature (oil refineries, petrochemical industries) and those related to domestic sewage, dredging and shipping. This chapter provides information on the state of the mesozooplankton assemblages in the Bahía Blanca estuary and the changes that have occurred over the last decades.

2 SPATIAL AND TEMPORAL DYNAMICS

The mesozooplankton of the inner zone of the Bahía Blanca estuary was first studied during 1974-1976 by Guerrero et al. (1976). The holoplankton fraction was made up of Calanoidea, Harpacticoidea, Cyclopoidea and Monstrilloidea, all copepods, and the meroplankton of decapod larvae from the Grapsidae family, polychaetes, gastropods, bivalves and the cirripeds *Balanus amphitrite* and *B. venustus niveus* (Valentinuzzi de Santos 1971). Zooplankton abundance increased during November 1974, reaching its maximum value in the warm season, with approximately 90,000 ind m^{-3}. Calanoid copepods appeared throughout the entire study zone, whereas harpacticoids were more abundant in the innermost stations. Brachyuran zoeae were the most abundant larvae in all the sampling stations, with a major dominance in spring and summer, whereas barnacle larvae dominated only in the innermost zone.

Hoffmeyer (1983) performed a characterization of mesozooplankton from the mouth to the head of the estuary during 1979 - 1981. The holoplankton was made up mainly of copepod species such as *Acartia tonsa, Paracalanus parvus, Labidocera fluviatilis, Calanoides carinatus, Oithona nana, Monstrilla sp., Monstrilla helgolandica, Tisbe varians, Harpacticus chelifer, Nannopus palustris, Robertsonia propinqua, Dactylopodia tisboides, Heterolaophonte sp. (aff pauciseta)* and *Euterpina acutifrons*.

P. parvus and the large calanoid copepods *C. carinatus* and *L. fluviatilis* were only present in the outer zone of the estuary. In contrast, *A. tonsa* occurred in the whole study area throughout the year (Hoffmeyer 2004a, Hoffmeyer unpub.). Its abundance showed a bimodal curve with maxima in early autumn and spring and a minimum in early winter. This species was considered a key taxon whose numerical significance impacted heavily on total mesozooplankton abundance (Sabatini 1989).

In terms of meroplankton, Hoffmeyer (1983) reported larvae of *Tubularia crocea, Ciona intestinalis, Conopeum sp, Spionidae sp, Balanus sp, Mollusca* and Grapsidae, the main taxon during 1979-1981 being Grapsidae larvae, which appeared in spring and increased in abundance in summer in response to increasing temperatures (Hoffmeyer 2004a). Subsequent studies conducted by Cervellini (1986, 1988) in the same study area from 1983 to 1985 revealed high larvae densities of two species of Grapsidae: *Chasmagnathus granulatus* (84% of total abundance) and *Cyrtograpsus altimanus* (60% of total abundance) in the inner zone of the estuary.

3 CHANGES IN MESOZOOPLANKTON STRUCTURE AND DYNAMICS

Seasonal zooplankton succession in the inner zone of the estuary during 1990-1991 was compared to that from 1979-1980. The results showed the presence of *Eurytemora americana*, a new copepod species, and a noticeable increase in *B. glandula* larvae (Hoffmeyer 2004a). According to Hoffmeyer (1994), *E. americana* was accidentally introduced via ballast water from ships. Since its arrival, this species has increased in density over the years, co-existing with *A. tonsa* during June-July to October and producing a slight exclusion of this copepod. *E. americana* remains in the plankton for around 4 months and then disappears, a phenomenon related to the strategy of diapause egg production (Hoffmeyer et al. in press).

Nauplius and cypris larvae of *Balanus glandula* were first recorded during the 1990-1991 period by Wagner et al. (1993). This new immigrant was first registered for Mar del Plata coasts (Spivak and L´Hoste 1976) and its distribution subsequently expanded southward along the coasts of Buenos Aires province and Patagonia (Rico et al. 2001, Orensanz et al. 2002, Hoffmeyer 2004a, 2004b). Since its arrival in the Bahía Blanca estuary, the population has grown rapidly, competing successfully for space with *B. amphitrite*, the other cirriped species. During the annual cycle 1996-1997, the plankton showed a similar seasonal succession compared to historical data (IADO, 1997), though a lower specific richness and abundance was detected.

Recent studies performed during 2006-2007 showed the presence of various developmental stages of *L. fluviatilis* and *C. carinatus* in the inner zone of the estuary (Hoffmeyer, 2008). An increase in P. parvus abundance was also recorded in the same zone. The presence of these species here, as opposed to the shelf waters adjacent to the estuary where it is usually found, can be explained by a zooplankton translocation caused by the exchange of ballast water between the outer and the inner zone of the estuary. Comparison of these results with those of 1979-1981 indicates an enhancement of the distributional range of *L. fluviatilis* and *C. carinatus* and an increase of *P. parvus* in the inner zone of the estuary. In the same study a higher abundance peak of the invader *E. americana* and a lower abundance peak of *A. tonsa* were observed than previously reported. Obviously, these peaks affected the total mesozooplankton abundance (Figure 1).

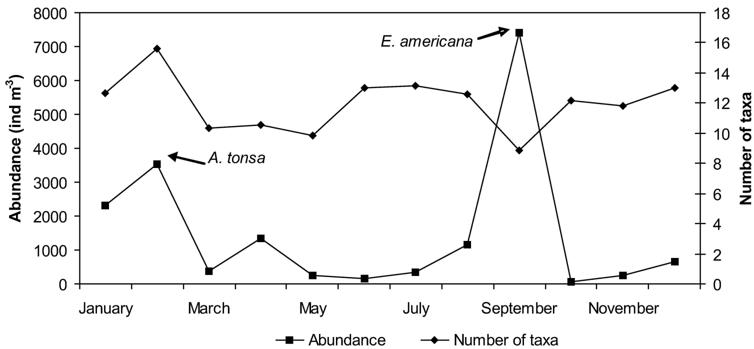

FIGURE 1: *Mesozooplankton abundance (ind m^{-3}) and number of taxa in the inner zone of the Bahía Blanca estuary, during 2007 annual cycle (extracted from Hoffmeyer 2008*

4 LOCAL IMPACTS ON THE MESOZOOPLANKTON ASSEMBLAGES

4.1 Effect of wastewater discharge on mesozooplankton diversity

Wastewater from the city of Bahía Blanca is discharged into the estuary after only primary treatment. To analyze the effect of this effluent on mesozooplankton diversity, a sampling programme was carried out from June to November 1995. Four stations were sampled, one in the wastewater discharge zone and the others in the Main Channel (Biancalana 2003). The polluted area exhibited high values of particulate organic matter and nutrients, particularly phosphates. In general, diversity (Shannon-Wiener index) decreased towards the wastewater discharge site, revealing a decrease in mesozooplankton total abundance and a low number of taxa in the discharge zone. These taxa are probably more resistant to the particular features of this site. Two zones were differentiated by cluster analysis, one at the point of discharge of the effluent and the other, far from this site. The results indicated changes in the mesozooplankton assemblages due to the effluent effect.

4.2 Effect of a power plant cooling system on mesozooplankton survival

The impact of the power plant cooling system (Comandante Luis Piedrabuena) in the Bahía Blanca estuary on the survival of mesozooplanktonic organisms was evaluated during the 1990-1992 period (Hoffmeyer et al. 2005). Mortality rates were calculated for juveniles and adults of *A. tonsa* and *E. americana*, and for larvae of the crab *Chasmagnathus granulata* and the cirriped *Balanus glandula*. Mean total mortality values were up to four times higher at the water discharge site than at intake, though for all species, significant differences were only registered in post-capture mortality. No evidence of greater larval sensitivity was found. The sharpest decrease in overall mesozooplankton abundance was found in areas close to heated water discharge.

5 CONCLUSIONS

The present work shows the changes in the structure of mesozooplanktonic assemblages in the Bahía Blanca estuary over the last 30 years. Such changes include the invasion of new species, the spatial and temporal variation of abundance patterns, a decrease in biodiversity and the extension of the distributional range of allochtonous species. The anthropogenic pressures caused by industrial and harbor development could partly explain the variation in the mesozooplankton assemblages. The continuous monitoring of this ecosystem will provide valuable information to document temporal variability and understand how anthropogenic factors interact with natural conditions. This ecological information will help to explain the cause-effect relationships between environmental changes and the biotic communities of the Bahía Blanca estuary.

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SALT-MARSHES: ROLE WITHIN THE BAHÍA BLANCA ESTUARY

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1 INTRODUCTION

Two important plant communities exist within estuarine environments: salt-marshes in temperate and warm temperate latitudes, and mangrove swamps on tropical and sub-tropical coastlines (Dawes 1991, Cagnoni 1999). These wetlands develop in the intertidal zone, a geomorphic environment that permits both the deposition of fine sediments and the establishment of vegetation (Luternauer et al. 1995, Davidson-Arnott et al. 2002), making them of particular interest from a biogeochemical point of view.

Salt-marshes are the main sites for the dynamic exchange of water, sediment and organic matter between uplands and coastal waters (Gordon et al. 1985). They can either act as a source or a sink of nutrients and contaminants (Weis and Weis 2004, Botté 2005, Hempel et al., submitted) depending on the current biogeochemical characteristics. Macrophytes have also been shown to play important roles in marsh biogeochemistry through their active and passive circulation of elements (Weis and Weis 2004). The production of organic matter in these environments is also of great interest because of the formation of detritus by halophytic plants, which may be an important component in the food chain in coastal waters due to high primary production and frequent tidal connection (Chalmers et al. 1985).

The Bahía Blanca estuary in Argentina, on the northern limits of Patagonia (situated between 38° and 39° S), is a good case study for evaluating the transfer of nutrients, metals and organic matter between the coastal zone and the open sea, because stands of halophytic plants are present, as described in others coastal salt-marshes (Luternauer et al. 1995, Mitsch and Gosselink 2000), which are normally dense and varied. A study of the biogeochemical balances of nutrients, metals and other contaminants which can be modified by an increase or decrease in load would provide an important data base that could be included in dynamic models such as MOHID. Moreover, ecological studies of the role of salt-marshes would provide information that would assist decision makers in land use planning within these systems, when considering future development (e.g., industrial locations, port facilities, fishery fleet, oil transport, processing and storage, etc).

Research on the production of *Spartina alterniflora* and *Sarcocornia perennis* is being carried out, as salt marshes are considered to be one of the more productive systems in the world (Gosselink 1980, Dawes 1991, Brewer 1994) and they cover large areas in this estuary. Studies of the function of macrophytes in nutrient and metal cycles are relatively recent. So, this chapter offers a brief overview of the biological and chemical characteristics of the salt marshes in the Bahía Blanca estuary in order to assess current knowledge.

2 STRUCTURE AND FUNCTIONS OF SALT-MARSHES

Salt-marshes are found along all the margins of the Bahía Blanca estuary except on the northern margin where various human activities have been developed (e.g. cities, ports and industries; Figure 1), as well as on all the tidal flats in this system. Salt marshes comprise a variety of plants: rushes, sedges and grasses, and they can be divided into two zones. The upper or high marsh is flooded irregularly and its upper limit is dependent on the extent of flooding caused by extremely high tides. The low marsh is flooded twice a day and its extent is delimited by the high and low water contour lines. The dominant salt marsh species in the Bahía Blanca estuary (Figure 2) include: *Spartina alterniflora* (Smooth cord grass) in the lowest areas that are most frequently flooded, *Spartina densiflora* (Dense-flowered cord grass) occurring along the higher marsh areas that are occasionally flooded, and *Sarcocornia perennis* ("jume") which grows under water-saturated soil conditions with high salinity. Beyond the reach of the ordinary tides, some shrubs grow e.g. *Atriplex undulata, Cyclolepis genistoides, Lycium chilense, Heterostachys ritteriana, Allenrolfea patagonica* and *Geoffroea decorticans*, and more or less extensive natural grassland develops where *Distichlis spicata* and *Distichlis scoparia* are dominant (Verettoni 1961, Lamberto et al. 1997, Nebbia and Zalba 2003, Nebbia 2005).

The distribution of species tends to be regulated by competition at the higher elevations, whereas at the lower sites it is governed by physiological tolerance to salt content. Only a few isolated shrub species are observed on the shoreline of the inner Bahía Blanca estuary where the soils contain a large amount of salt, but, as the land rises and the salinity diminishes, the percentage of bare ground decreases and less tolerant species are found (Verettoni 1961, Nebbia 2005). On the outer area of the estuary, *Spartina alterniflora* marshes develop as discontinuous patches near to the limits of normal high tides (Figure 3).

A comparison of the data of the last census (2002-2003) with that of 1961 shows an important similarity in both the dominant species and their abundance, although several species show significant increases in the area covered (e.g. *S. densiflora* and *A. undulate*) (Nebbia and Zalba 2003). The surface coverage in the Bahía Blanca salt-marshes (based on satellite image analysis between 2002-2004) by the three dominant species is: 9,193 ha of *Spartina alterniflora*, 65 ha of *Spartina densiflora* and 20,376 ha of *Sarcocornia perennis* (Isacch et al. 2006).

As a result of human activities in the Bahía Blanca estuary in the last 25 years, several salt-marshes have been drained or filled with silt, sand or municipal waste, so they can no longer survive and finally disappear. The percentage of estuarine habitat which has been lost due to dredging or land fill activities has not been quantified. However, the Argentine Institute of Oceanography has been studying changes in the coastal habitats of the Bahía Blanca estuary for many years and scientists have evaluated changes in certain areas, such as the Puerto Rosales area, by comparing aerial photographs obtained between 1936 and 2000, as well as using LANDSAT images of the coast from 1998, 2000 and 2003. The changes observed show an expansion of *Spartina alterniflora* habitats, especially in the last 10 years (Federici et al. 2003).

3 CHARACTERIZATION OF DIFFERENT SCENARIOS

3.1 Middle area of the estuary

The salt marshes at Villa del Mar, a small village near to Punta Alta, approximately in the middle of the Bahía Blanca estuary, have particular characteristics. Differences have been found in plant height and in the patterns of standing and exported biomass in the *S. alterniflora* marsh, which can be attributed to topographical factors associated with differences in the exposure to tidal water. The water would be acting as a stress and forcing factor that is also a vehicle for the exportation of dry material (Gonzalez Trilla et al. 2007).

Net Primary Production (NPP) represents the real production rate of new biomass (Begon et al. 2006). NPP is a key function of ecosystems and plays a main role in the carbon cycle and energy balances. Primary production in the *S. alterniflora* marsh at Villa del Mar was measured using clipped-plot techniques which gave values of 969 $g\,m^{-2}\,yr^{-1}$ in the low marsh and 745 $g\,m^{-2}\,yr^{-1}$ in the high marsh (González Trilla et al. 2007), showing the importance of exposure to tidal water in this system. On the other hand, the net production, measured by harvesting methods, was greater in the high marsh than in the low marsh, which was observed in every season, as well as for the whole year. The accumulated average (above-ground tissues only) in one year were 1,496.1 $\pm$ 51.8 $g\,m^{-2}$ and 351 $\pm$ 15.4 $g\,m^{-2}$, respectively (Negrin et al. 2007, Figure 4). This might indicate that this species prefers unsaturated soils, even though it is adapted to flooding, which could be due to the reduced rate of nutrient absorption in saturated conditions (Cagnoni 1999).

Another important characteristic of the marshes is the detritus that is generated from the decomposition of dead plant tissue (Tenore 1977, Currin et al. 1995, Weis et al. 2002). Nutrients, organic matter and metals are removed from the sediment through the detritus and put into the surface substrate and then released into the sea water and so it plays an important role in the cycling of elements (Hopkinson and Schubauer 1984, Quan et al. 2007). The input of nutrients to the system by decomposition, which is being evaluated in this area using the litter- bag technique (Bocock and Gilbert 1957, Odum and de la Cruz 1967), depends on the elemental composition of the tissues and the biomass production. The biomass of *S. alterniflora* at the Villa del Mar site is mainly found in underground tissues and they seem to show a higher phosphorus concentration than the aerial parts, so the input of this element by this species could be of great importance (Negrin et al. 2006). On the other hand, as mentioned in the previous paragraph, the above-ground biomass of the high marsh is abundant, so the input of nitrogen and phosphorus from this source should be also considered. The amount of element input can be estimated by multiplying the biomass with the corresponding fractional concentration of each element, as was done for the *S. alterniflora* marsh in Villa del Mar (Table 1, from Negrin et al. 2007).

FIGURE 1: *Different human activities in the middle part of the estuary (Puerto Galván and Cargill, respectively); each one showing relict patches of* S. alterniflora*.*

FIGURE 2: *Examples of salt-marsh plants in the Bahía Blanca estuary. a)* Spartina alterniflora*, b)* Spartina densiflora*, c)* Sarcocornia perennis*.*

TABLE 1: *An evaluation of the seasonality of nitrogen and phosphate inputs from above-ground tissues of* S. alterniflora*.*

	Low marsh		High marsh	
	N	P	N	P
Winter	1.129	0.084	368.417	30.364
Spring	0.824	0.104	268.632	28.642
Summer	0.892	0.071	412.672	31.273
Autumn	0.593	0.057	162.363	19.334
Total	**3.438**	**0.316**	**1212.084**	**109.613**

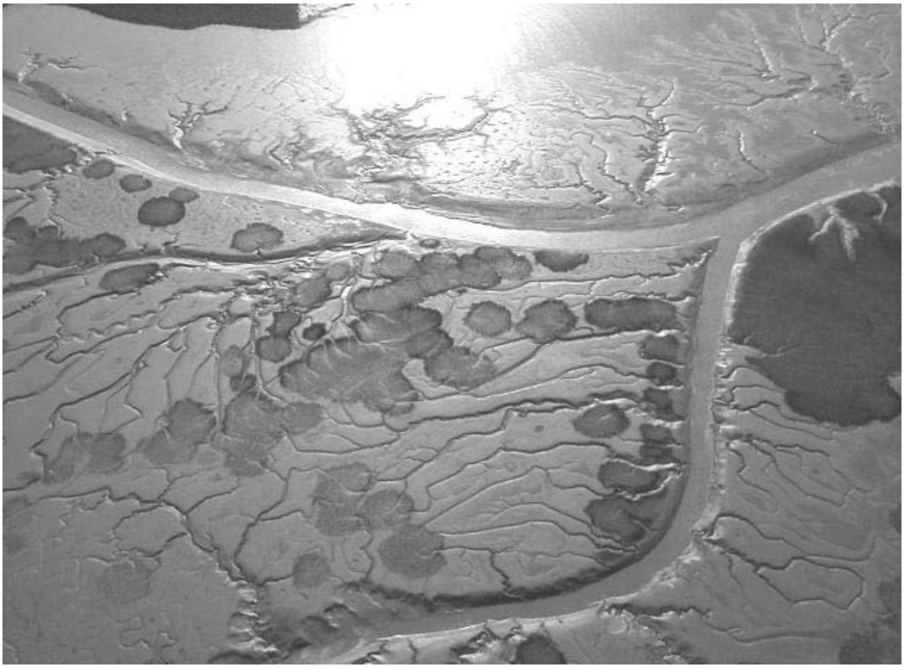

FIGURE 3: *Patches of* Spartina alterniflora *in the outer areas in Bahía Blanca estuary.*

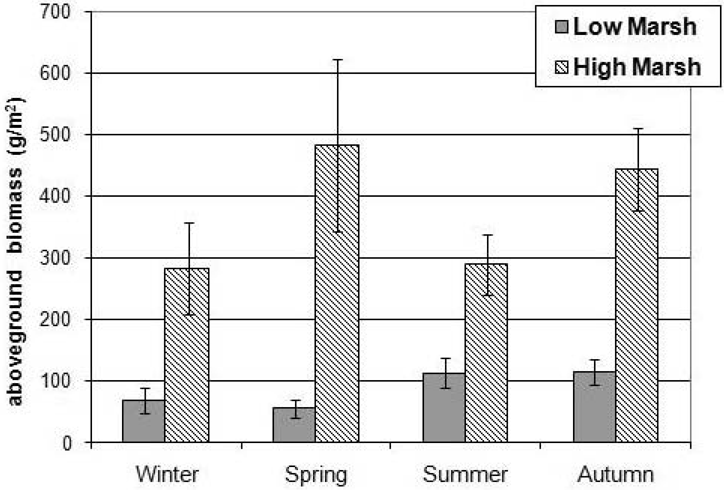

FIGURE 4: *Seasonal changes in above-ground biomass in zones with different degrees of exposure to tidal water. Differences in all seasons between both sites were significant but seasonal changes within each zone were not significant (n=4 for winter and spring, and n=8 for summer and autumn; values are means with one SE bar) (from Negrin et al. 2007).*

Furthermore, the amount of detritus generated by *S. alterniflora* in respect to the green tissues is relevant information. Recent studies have shown that live/dead ratios were greater in the low marsh than in the high marsh throughout the whole year, indicating a greater loss of dead tissues due to greater influence of tidal water and, as a consequence, a lesser importance of the detritus as a source of food in this zone (Negrin et al. 2007, Figure 5). In addition, other potentially harmful elements, such as heavy metals, can also be passed into the environment through detritus and finally to consumers (Weis and Weis 2002, Botté 2005, Hempel et al., submitted).

Marsh macrophytes are also very important in biogeochemical cycles when the concentration of nutrients and organic matter in areas with vegetation and bare ground are compared. In the *Spartina* marshes at Villa del Mar, the unvegetated areas show a greater concentration of nitrogen nutrients than the vegetated ones, with an annual mean of 41.20 vs. 2.28 μM, and 54.29 vs. 27.33 μM for nitrate and ammonium, respectively. These are the more abundant nitrogen species and therefore the main source of this element to the plants (Negrin et al., in press). In addition, high concentrations of particulate organic matter (POM) occur within this estuary (Freije et al., this volume), with even higher levels in the porewater from vegetated areas, with annual mean values at the Villa del Mar site of up to 20 mg C l^{-1} and without any significant differences with respect to the bare tidal flats (Negrin et al., in press),

3.1.1 Phytoremediation

Salt-marshes are often considered strategic for the retention of contaminants. Several authors have reported that *Spartina alterniflora* can accumulate metals from sediments via the roots and translocate some portion to the above-ground tissues (Weis and Weis 2002, Botté 2005). So, *S. alterniflora*, and also other typical macrophytes such as *Sarcocornia perennis*, could be used to immobilize metals and store them in roots under the ground and/or in the soil ("phytostabilization"), preventing the movement of pollutants from sediments into the food webs of marshes and waters near to the shore (Weis and Weis 2004, Botté 2005). The discovery of metal hyper-accumulating properties in certain plants has suggested that it may be feasible to use these plants for environmental restoration ("phytoremediation"), especially those species that have a high biomass production, such as *S. alterniflora* (Ensley 2000). These plants can also export different elements to the system, regulating their concentration and transference between matrixes.

The *S. alterniflora* near to Villa del Mar does not show any clear differences in the distribution of metals between the medium and high salt-marshes, but there are some differences in the concentrations within tissues. Greater values have been shown in underground tissues in the medium as well as in the high salt-marshes (Hempel et al., submitted). Moreover, differences were found in the distribution of metals in the sediments: they are more concentrated in tidal flats (and also nitrogen nutrients show a similar distribution, Negrin et al., in press) than in vegetated areas, in sites where human activity is greater, whereas in areas with less impact the sediments linked to underground parts of *S. alterniflora* have a higher metal content (Hempel et al., submitted).

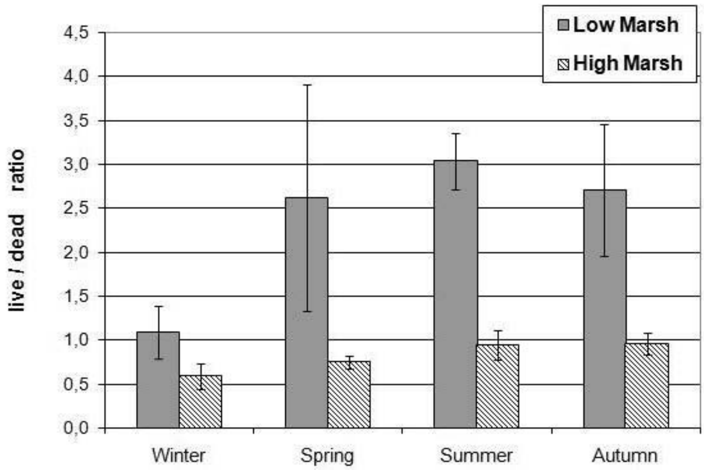

FIGURE 5: *Seasonal changes in live/dead ratios in zones with different degrees of exposure to tidal water. Only differences between both sites in summer and autumn were significant. (n=4 for winter and spring, and n=8 for summer and autumn; values are means with one S.E bar) (from Negrin et al. 2007).*

3.2 Inner area of the estuary

In the inner part of the estuary, between Puerto Galván and Puerto Cuatreros, the organic matter concentrations in the porewater show the same tendency. The levels are much higher in the interstitial water from tidal flats (mean values up to 34 mg C l^{-1} for Puerto Cuatreros, Maldonado and Puerto Galván) than in the subsurface sea water (Botté 2005).

3.2.1 Phytoremediation

Sarcocornia perennis has been found together with or replacing *Spartina alterniflora*, especially in the inner marshes of the Bahía Blanca estuary. Its expansion can alter edaphic conditions because *S. perennis* is physiologically and morphologically different from *Spartina* spp. (Lamberto et al 1997) as well as metal and nutrient cycling processes (Windham et al. 2003, Botté 2005). Comparative studies of the metal dynamics between *S. perennis* and *S. alterniflora* species have been carried out in the inner zone of the estuary. The *Spartina* marshes showed higher values of essential metals whereas *Sarcocornia* marshes mainly accumulate toxic elements e.g. cadmium or lead. Moreover, *S. perennis*, which is the main species in the Puerto Cuatreros area, sequesters more metals in above-ground tissues than *S. alterniflora*, whereas the latter species has shown that it retains more metals in underground structures (Botté 2005).

The sediments associated with the underground parts of the plants have shown higher concentrations than those from the bare flats for most of the studied metals (Botté 2005). This suggests an important role of salt-marshes, because elements such as metals would be less available in the environment as a whole if they are stored in plant tissues.

4 CONCLUSIONS

The results obtained so far show that salt-marsh macrophytes function as regulators in shallow and highly variable ecosystems, such as the Bahía Blanca estuary. Salt-marshes are of particular interest because of their vulnerability to different human activities and a better understanding is needed of the way they respond to environmental changes. The densely vegetated salt-marshes in the Bahía Blanca estuary may influence the nutrient, organic matter and metal fluxes between the tidal flats and the water body. The way in which the metals and nutrients are distributed within plants and the associated sediment and the extent of uptake can have important effects on their residence time in plants and in wetlands, and their potential transfer to the sea.

Salt-marshes are capable of exporting materials that are stored in the underground tissues of macrophytes to the surrounding ecosystem via detritus, and they act as good regulators in the biogeochemical balance of nutrients, organic matter and metals. All this information will have implications for the Coastal Management of the Bahía Blanca estuary, providing an appropriate scenario for use of models, which might help to give an integrated image of this environment.

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SOCIO-ECONOMIC ISSUES IN THE BAHÍA BLANCA ESTUARY

N. PIZARRO AND M.C. PICCOLO

1 INTRODUCTION

The Bahía Blanca estuary is situated in the south-west of the Buenos Aires province surrounded by the Coronel de Marina Leonardo Rosales (Coronel Rosales), Bahía Blanca and Villarino districts (Figure 1). Each of them present special demographic features, among which Bahía Blanca stands out because of it demographic influence and economic and educational significance. Many anthropogenic activities are carried out on the coastline. Figure 2 shows the main industrial sites located in the coastal area. The most important one is a Petrochemical Complex located at Ing. White in the Bahía Blanca district (Figure 1). The harbour area is included in the regulation policy of the Environmental Law N°12350. The aim of this law is to protect the environment and its main objective is to create a Special Programme for Preservation and Optimization of the Environment Quality by monitoring and controlling the gaseous emissions and the liquid effluents originated at the industries. However there are sites of the estuary that are not protected by this law. Examples are the presence of a refinery, an effluent treatment plant, etc. very close to the city.

The main socio-economic problems identified at the Bahía Blanca estuary, after numerous meetings and surveys with stakeholders related to the estuary activities, are: (1) the unequal growth of the population at the three districts which surrounds de estuary; (2) the continuous loss of the fishing resources which create serious problems for the local fishing community; and, (3) the quality of the Bahía Blanca estuary waters for recreational purposes. Due to the different themes, in this chapter these subjects are separately developed.

2 DEMOGRAPHIC DYNAMICS OF THE DISTRICTS OF THE BAHÍA BLANCA ESTUARY

The evolution of the population of the three districts that surround the Bahía Blanca estuary (Bahía Blanca, Coronel Rosales and Villarino) was analyzed with the data from the National Population and Housing Census of the National Institute of Statistic and Census (INDEC) over the last four decades (1960-1990). The result of the population data (age, density and activities) was related to the evolution of the industry in the area and the activities of the three major ports located on its coasts. They are Cuatreros, Ingeniero White and Puerto Belgrano harbours (Figure 1).

The population density of the districts did not show significant variations. According to the last population census Bahía Blanca presented a density of 118.3 inhab km^{-2} in 1991 and 123.8 inhab km^{-2} in 2001. At the Coronel Rosales district the density varied from 45.4 inhab km^{-2} in 1991 to 46.4 inhab km^{-2} in 2001 and in Villarino, the district with a bigger area but a lesser population, showed 2.1 inhab km^{-2} in 1991 and 2.3 inhab km^{-2} on the last census.

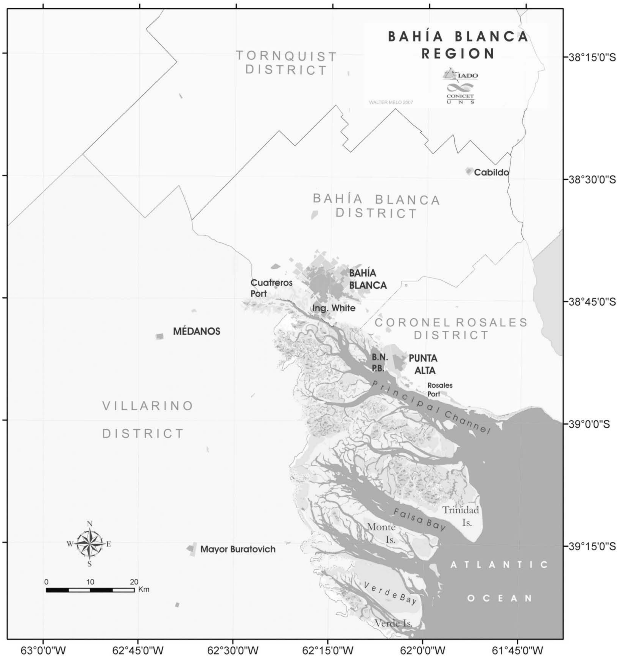

FIGURE 1: *Surrounding districts location of the Bahía Blanca estuary.*

An analysis of the total population data of Bahía Blanca for the years 1960-1970-1980-1991 and 2001, compared to the data of the Buenos Aires province and the whole country, shows that Bahía Blanca has a moderate intercensus growth with a marked decrease up to 1991. The Villarino district differs substantially from the others, presenting a low growth during the first periods and a small recovery during the last ones (Figure 3).

The Bahía Blanca district showed a gradual demographic growth during the periods 1970-1980 and 1991-2001, however the growth is small compared with the whole Buenos Aires province in the other intercensus periods values were similar. The situation is different when the growth is considered at national level because the population growth of the district is faster during the periods 1960-1970 and 1970-1980, is the same in 1980-1991 and decreases during the last period. The Coronel Rosales district shows a greater population growth in the period 1960-1970, but noticeable decreases in the two following intercensus periods and slightly increases in the last one. This situation may be influenced by the elimination of compulsory military service and a severe restructuration of the Navy, which had a strong impact because of the range of influence of the Puerto Belgrano Navy Base. Villarino, mainly a rural district, presents a very low growth up to 1980, a situation that changes during the 1980-1991 period, and coincides with the national and province growth during the last period, which indicates a stable growth. The decrease observed may well be caused by the agricultural crisis and the increase by the boom of horticulture, mainly onion crop.

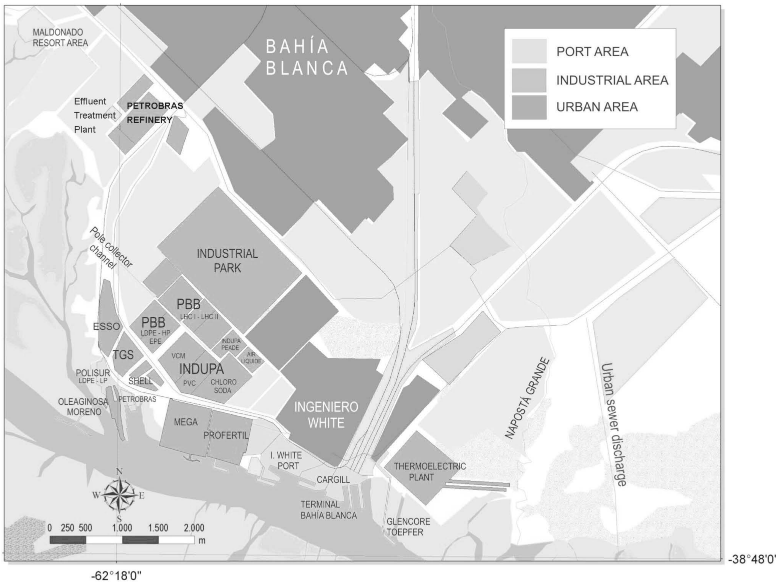

FIGURE 2: *Localization of the main industrial sites at the petrochemical complex in the Bahía Blanca district.*

2.1 Sex ratio

This parameter is defined as the proportion of males to females in a given population, usually expressed as the number of males per 100 females. In the Bahía Blanca district a small value is found due to the higher life expectancy of women compared with that of men (Figure 4). This is particularly true for people over 65 (Figure 4d). In the Coronel Rosales district, because of the presence of the Navy Base, the population is characterized by more males than females, situation that is similar in the Villarino district, where men are needed to perform typical rural activities

2.2 Age population analysis

The three districts surrounding the estuary present different age ratios (calculated as the number of persons 65 years old or over per hundred persons under age 15). Bahía Blanca stands out with the biggest index, which is increasing. In fact, the ratio changes from 33.8 people in 1980, to 52.3 in 2001. The index shows the increase of the aged population. The Coronel Rosales district also shows an important ageing index (from 31 in 1980 to 39.8 in 2001), while Villarino is the district that has the lowest, based on a bigger birth rate.

Bahía Blanca is the district with the most demographic weight. According to the 2001 census it had 284.776 inhabitants, with an increase of 4.62%. In former census the growth was bigger. From 1970 to 1980 it rose 18.65%; between 1980 and 1991 it rose 16.30%. As stated by the latest census (Figure 5a), a remarkable decrease of the birth rate is observed, especially during 2001 (-10.46%) A slight increase of the active population (mainly in groups from 15 to 19 and from 20 to 24) is noticeable. The main reason is the number of students that come from the nearby cities to study at the National University (Universidad National del Sur) located in Bahía Blanca city. The figures show an important increase of the passive population (over 65 years old, 21.44%).

The Coronel Rosales district has 60.892 inhabitants according to the 2001 census (Figure 5b). A slight increase of 2.27% with regard to the former census (1991-2001) was found. In the period 1980-1991 there is a decrease of 0.53 in the 15 to 19 year olds because of the elimination of the compulsory military service on the Puerto Belgrano Navy Base. As in the rest of the country, in this district the birth rate shows an evident decrease (-6.78%) and the over 65 year old group presents the biggest increase (20.91%). The Villarino district had, according to the 2001 census, 26.517 inhabitants (Figure 5c). This district presents the biggest population increase, reaching 6.56% between 1991 and 2001. The over 65 year old group increases the most (23.76%). A slight decrease in birth rate was found, an increase in males from 5 to 29 year olds and in females, from 5 year olds to the top of the pyramid.

2.3 Urban and rural population

The urbanization and the rural area of the three districts are very different. As mentioned before, Bahía Blanca has a mainly urban population (98.7%), Coronel Rosales shows a slightly smaller proportion (94.1%) and in Villarino only 61.5% of the population is urban (Figure 6). According to the data of the latest census the scarce rural populations of the Bahía Blanca and Coronel Rosales districts have slightly decreased. Villarino shows more significant changes in 1980 the rural population was up to 50%, in 1990 48%, and according to data of the 2001 census, this figure decreases to 38% (Figure 6). Summarizing, the population of the districts that limit the Bahía Blanca estuary is decreasing significantly. Nevertheless, over the past 20 years the number of industries that have settled in the Petrochemical Complex and ports has grown rapidly. Therefore, in the coming years, the authorities should take steps to generate a successful development of the region.

3 FISHERIES

The coastal regions are transition and interaction spaces between land and marine environments. Fisheries which develop in these areas are characterised by an environmental complexity (different oceanographic conditions, contamination problems because of the spilling of waste and contaminants from urban and industrial areas); biological complexity (great variety of species, breeding areas, reproductive areas, shelter areas for young stages); and finally social complexity (a lot of manpower employment, allows the survival of important coastal communities). The study of these regions must be focuses from a multidisciplinary and integral point of view, to make decisions for a sustainable development (Perrotta et al. 2007).

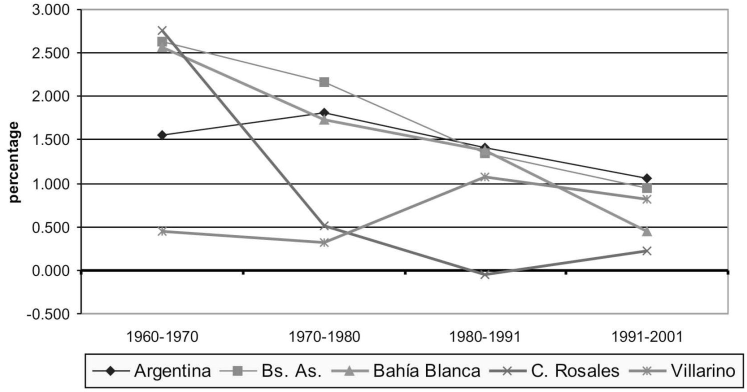

FIGURE 3: *Population growth by decades (Source: INDEC National Census of Population and Housing 1960, 1970, 1980 and 1991, National Census of Population, Home and Housing 2001).*

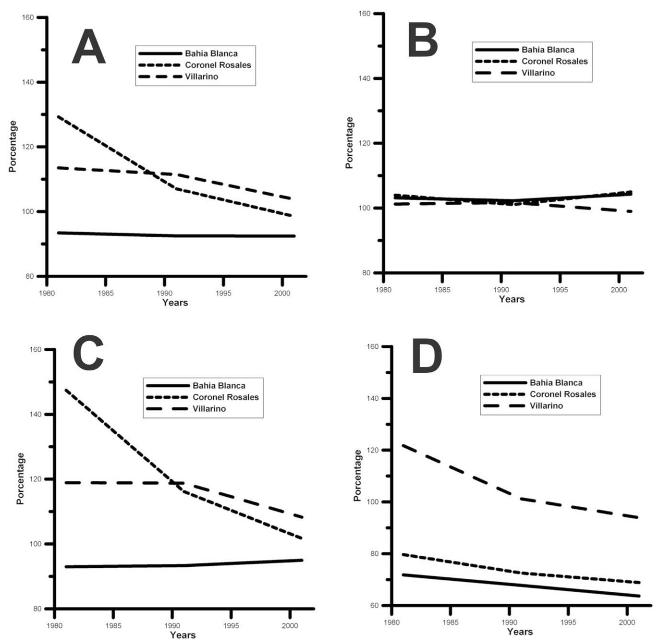

FIGURE 4: *a) Sex ratio, c) Sex ratio, under 15 years old, c) Sex ratio, 15 to 64 years old and d) Sex ratio over 65 years old.*

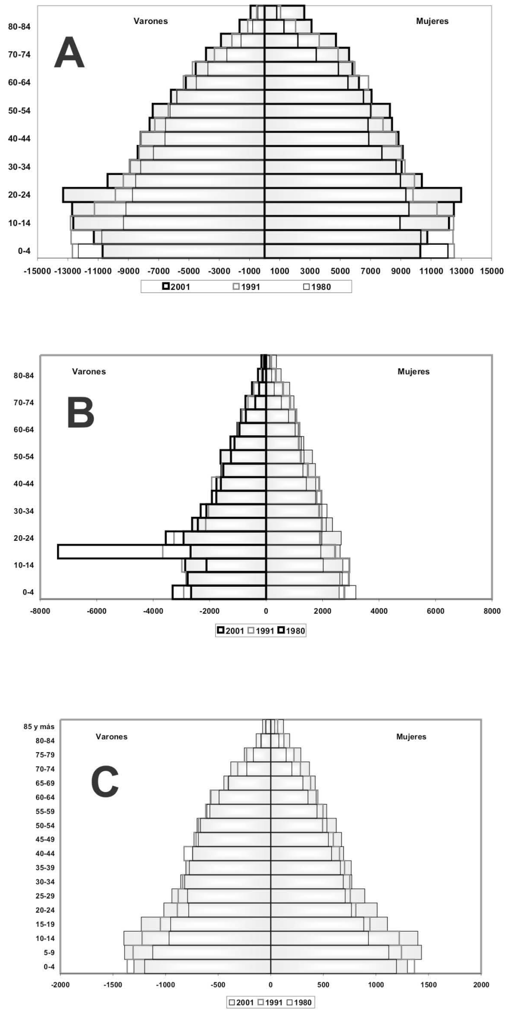

FIGURE 5: *a) Bahía Blanca; b) Coronel Rosales and c) Villarino population pyramids.*

As a fishery, the Bahía Blanca harbour has little national relevance. The harbour at the city of Mar del Plata monopolises this activity in the Buenos Aires province, receiving about 95% of the fish captures, including species from the Bahía Blanca estuary. To the south of our country, the main harbours of Chubut and Santa Cruz provinces receive quantities similar to those Mar del Plata does. The unloading at Bahía Blanca is about 40.000 tons, whereas Mar del Plata unloads 450.000 tons, Chubut 400.000 tons and Santa Cruz 300.000 tons. On the other hand, the unloading at the local harbour consists of species from deep waters, at open sea, mainly from the south. Among these species are Patagonian tooth fish, Patagonian cod, longtail hake, southern blue whiting and grenadier.

Artisanal fishing is important for the Ing White community located at the Bahía Blanca district because many families depend on this activity for their livelihood. According to data from the Commercial Exploitation Department of the Subsecretariat for Fishing Activities of the Buenos Aires Province there are 114 fishing ships operating in Ingeniero White and Coronel Rosales harbours (Figure 7). Only 10 of them are more than 13 meters long (GBA 1996a, b). The main species captured at the estuary are striped weakfish (15% of the annual catch), Patagonian smooth hound (15% of the annual catch), white croaker (10% of the annual catch), silver side (3% of the annual catch) and flounder (2% of the annual catch). The latter, although it has a low percentage compared to the other species, has a very high commercial value and makes a good profit (GBA 1998a,b; Suquele y Colautti 2004). The fishing period at the estuary extends from September to May, depending on the species (Pizarro et al. 2007). On the other hand, crustaceans like Argentine stiletto shrimp and Argentine red shrimp, are very important for the fishermen in Bahía Blanca, although most of the catches are made in Riacho Azul, a natural reserve of the San Blas and Anegada bays, located at the south of the estuary (Figure 1).

Sport fishing is another important activity in the estuary. During the first week of May the Shark Week takes place. It is an eagerly awaited competition in which many fishermen compete for the best shark capture. Another fishing event occurs every year during six four-hour days of fishing at the Punta Ancla beach on the northern coast of the estuary. This annual competition is for bed silver side fish.

3.1 Historic evolution of the fishing activities at the Bahía Blanca estuary

The Bahía Blanca estuary is an area for fish feeding and breeding. The same as other protected areas like lagoons and coastal zones it fulfills an important role for marine species conservation. This estuary is one of the most important protected areas of the Buenos Aires province coastline. Previous studies of the region concern taxonomy (Berg 1895, Ringuelet and Aramburu 1960, Menni et al. 1984) of a very small number of species. López Cazorla (1987, 2004) made a detail study of the fish fauna of the estuary in order to determine its spatial and temporary distribution and the level of utilization of the environment at the different stages of each species (young and adult).

The seasonal fluctuations of the fish populations in this type of environment are due to numerous reasons, temperature being the main factor. The increase of the number of species and individuals is directly connected with the increase of the water temperature; a high number of species is recorded in summer and a lower number in winter. Although most of the species are suitable for human consumption, only five present a significant economical interest. These are: striped weakfish (*Cynoscion guatucupa*), white croaker (*Micropogonias furnieri*), Patagonian smoothhound (*Mustelus schmitti*), flounder (*Paralichthys orbignyanus*) and silver side (*Odonthestes argentinensis*) (López Cazorla 2004).

Artisanal fishing is the activity carried out by one or more fishermen, using small ships, on areas near the seaside (Fernández 1999). According to the Federal Fishing Council of the

Buenos Aires province, artisanal fishing is defined as an activity involving catching and extracting sea products like fish, crustaceans, molluscs and seaweeds; done in a direct, personal and regular way by fishermen at 12 miles from seashore, by means of simple extraction techniques with direct involvement of fishermen. Beach nets, gillnets, fish traps and different types of hooks can be employed from the seaside, with or without the assistance of small boats. Artisanal fishing can be performed in geographic areas that differ in the characteristics of water and environment. Artisanal fishing must be differentiated from the industrial, since artisanal fishing is an activity that constitutes the main support of many families, which is why there is a great concern (La Nueva Provincia 2004).

During the last 20 years artisanal fishermen have been complaining about the continuous loss of the fishing resources at the estuary. Important governmental institutions reported the extermination of many marine species. During the year 2002 the fishermen pointed out that white croaker had disappeared and that other species of the estuary, such as striped weakfish and Patagonian smoothhound were facing extinction (La Nueva Provincia 2002). Nevertheless, studies on the evolution of this resource at the estuary are scarce. In 2001 a law for fishing emergency was passed. Therefore, the main objective of this item is to discuss the evolution of the artisanal fishing in the Bahía Blanca estuary related to changes in the fisheries legislation, fishing equipment and the commercialization of the product in the different estuary harbours located at the Bahía Blanca estuary.

The decline of the fishing industry since 1980 has caused a change in the fishery legislation. To preserve the resource, the fishing equipment used by fishermen was modified (Pizarro et al. 2007). A cooperative institution used to concentrate all the fishing information and the number of captures from the estuary. Because of the fishing activity crisis, this cooperative was closed and replaced by private companies. Later, the legislation and the ways of reporting the fishing capture change. Nowadays the Subsecretariat for Fishing Activities of the Agricultural Affairs Ministry of the Buenos Aires Province is in charge of gathering the information.

The artisanal fishing volume and its corresponding income have been decreasing for the last 25 years. According to artisanal fishermen, the period 1978-1981 was when the activity reached its maximum captures (CedePesca 2004). During those times the total amount of extractions was only an extremely low portion of the resource. The quantity and capacity of the boats set the limit of the captures. There were more than fifty 12 meters long boats. Depending on the season they fished flounder, white croaker, striped weakfish, Patagonian smoothhound and silver side, among other species. The analysis of the capture data provided by fishermen for the period under study confirms the decrease of the resource at the Bahía Blanca estuary. The decrease of the captures is significant and gradual year by year for all species.

The most important resource in the Bahía Blanca area was the striped weakfish. This species historically represents between 45 and 70% of the total captures. During the year the adult stage presents two peaks of abundance: the first in April - May and the second, and more important in August - September. The adult stage of the fish enters the region to feed and recover after reproduction (López Cazorla 2004). Figure 8a shows the reported captures on chosen years. The decrease of the capture is noticeable. Similar results can be observed for all the species, with significant annual variations, as is common for this type of resource.

Another example is the white croaker (Figure 8b). At the seashore of the Buenos Aires province, the white croaker is fished during the whole year, but not at the Bahía Blanca estuary, where the young stages appear in autumn whereas the adult stages are fished in spring and summer. The latter are more abundant during the period between November and March (López Cazorla 2004). Annual variations are very significant during the period under study, but the decrease of the resource is an undeniable fact. Other species show similar fluctuations.

3.2 Future fishery resource

To study the commercial fishing activity, capture, production and commercialization processes must be considered. All over the world these activities are in charge of different economic partners, but in Argentina different economic groups have developed them. The analysis of the resource sustainability is essential, since it is one of the greatest problems that arise from the sector's crisis. This is a new issue in the country, because the national exploitation was always under de maximum sustainable capture level for each species.

All partners involved in artisanal fishing agree about two important issues. The first one is that the activity is undergoing a profound crisis, mainly due to a lack of an integrated coastal management. The second is that there is consensus to define the main problem, i.e. the scarcity of the resource inside de estuary. The decrease of the resource has many causes. One is the rise of the harbour activity at the Bahía Blanca estuary. The continuous dredging and the water contamination due to waste spilling have harmed the food for fish, and as a consequence the quantity of fish coming into the estuary is smaller since the food is not enough (CedePesca 2004).

Other opinions indicate that the main reason for the scarcity is bad management of the resource out of the estuary, which causes more captures, and as a consequence fewer fish coming into the estuary. In the mid-eighties the artisanal Mar del Plata fleet was restructured into nearby and faraway fleets and started fishing further south. It was then that the decrease of fish captures started. The new fishing area where the Mar del Plata fleet was fishing (Claromecó, Monte Hermoso) is where many species lay eggs. Their fishing impacts on the resource evolution (CedePesca 2004). According to López Cazorla (2004) this is the case with striped weakfish. The origin of the changes observed in the region may be related to the extensive fishing of this species, both in the Uruguay-Argentine common fishing area and in the Rincón area, which is located to the south of the Buenos Aires province, close to the mouth of the estuary.

This last factor is closed related to the modifications of the fishing legislation in force in the Buenos Aires province. These laws allow artisanal fishing from Mar del Plata to move to the south. Unfortunately modifications of the law were late and affect breeding areas of these marine species. Faced with different sector pressure because of the almost exhaustion of certain species and social unrest, the government passed a new law. This law resizes the sector, considering the decrease of captures, establishes a different way of distribution of the fishing rights through a share system, indicates a precise control mechanism on the exploitation of the resource and greater scientific accuracy in decision making regarding the ecosystem (Rodríguez et al. 1999).

4 MALDONADO RESORT AREA: UTILIZATION BY THE LOCAL COMMUNITY

Among the tourist attractions of Bahía Blanca, the Maldonado resort area, located at the estuary, is the most important recreational area of the city. By the end of XIX century there was a small harbour favoured by fishermen because of its proximity to the urban area. It was named ”Puerto Maldonado” and had a small inn and accommodation services. Around the year 1885 the residents of the city started going there during summer. By 1898 tents had been installed, there was a small restaurant and people could rent bathing suits. During the summer there was also a transportation service eight times a day.

The first important modification of the environment took place in 1918, when Mr Pedro Somaruga, the new owner, inaugurated the ”Bahía Blanca Great Resort”, although in the course of time, the place returned to its original name. At that time the resort had a 500 m diameter swimming pool with access ramps and stairs on either side, 140 single booths and 40 bigger booths for families” ”The water in the swimming pool changed at every tide, and according to customs of that time, the pool was divided into two separate sectors: one for women and the other for men” (Rimondi 1977). The features of the place remained almost unchanged until recent times. Due to the growing popularity of the resort, the Bahía Blanca Municipality issued a permit to Pedro Somaruga to build and exploit an economical tram for the next 10 years. The tram was narrow type rail (600 mm), powered by gasoline, and had open coaches for 28 seated passengers and a maximum of 6 coaches.

In December 1923 the tram service started with a total travelling distance of 4122 meters. The tram runs during summer, at 30 minutes intervals. It was very popular among the citizens due to its low cost, but because the starting point was in the outskirts (Malvinas street), for most people it was time consuming and costly to reach” (Rimondi 1977). This was why people started using other means of transportation, such as buses ”As early as 1925 the tram stopped its service” On February 1942 a strong storm completely destroyed the resort devastating all the booths. One year later a new owner rebuilt it, including a lodging house and a restaurant. On October 12, 1949 the government of the province expropriates it (until that time it had always been privately). Due to the progressive damage, the Municipality started taking charge of maintenance and finally in October 1964 the resort was placed under Municipal control.

At present, it has a salt water swimming pool for adults and one for children. Water comes from the Bahía Blanca estuary through specially built lock gates. They are opened once a day to change the water of the pool. There is a leisure activity sector with grills, tables and benches,

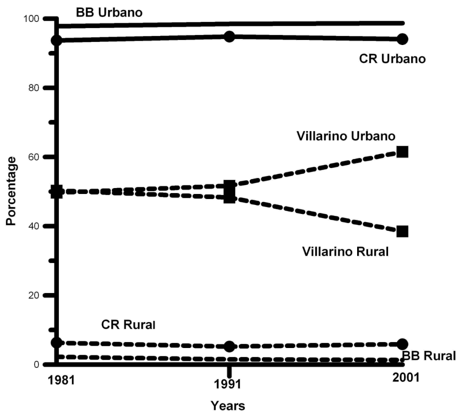

FIGURE 6: *Percentage of total urban and rural population of Bahía Blanca (BB), Coronel Rosales (CR) and Villarino. Period 1981-2001.*

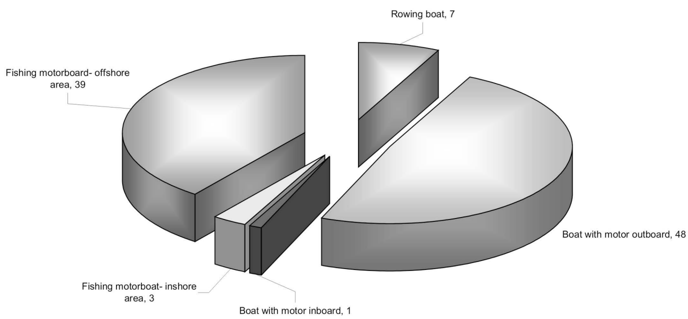

FIGURE 7: *Fishing fleet at the estuary (after GBA 1998a)*

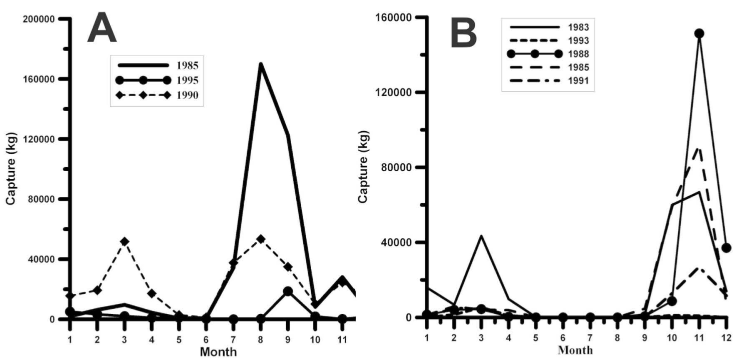

FIGURE 8: *Reported captures on chosen years of a) striped weakfish and b) white croaker (Pizarro et al. 2007)*

umbrellas, tents, mini-market and a sanitary building with bathrooms, showers, lockers and external showers, among other facilities. A mobile first aid facility, in charge of Red Cross personnel is available during the whole season, from December to mid of March (Piccolo et al. 2006). During 2004 some changes were made: 120 tents and several umbrellas were added; repair of the infirmary sector adding new medical services; renovation of the sanitary sector; construction of a cement pavement on part of the swimming pool bottom; repair of the stairs and lock gates; construction of concrete paths inside the resort, renovation of the restaurant and repair of the grills.

Because of the importance to the population of the Maldonado pool and considering the pollution problems of the Bahía Blanca estuary it was decided to study the quality of its water. Several meetings were carried out with the authorities of the Social and Human Development Secretary of the Municipal Government. Their main concern was to have pollution- free waters and their archives have given us useful information.

4.1 Visitors

Data provided by the Bahía Blanca Municipality show an important number of persons attending the resort during all the seasons particularly the 2002-2003 one with 67380 visitors. The month with the most visitors is always January, with values oscillating between 25344 (2001-2002 season) and 34667 (2002-2003 season, Table 1). The day with the most visitors of each season is Christmas day (December 25). Season 2004-2005 was an exception because of the unfavourable weather conditions, with low temperatures, 19 mm of rain and winds over 100 $\mathrm{km\ h^{-1}}$ (with gusts reaching 180 $\mathrm{km\ h^{-1}}$) (Piccolo et al. 2007).

In January, Sunday is the day with the biggest crowd except those days with bad weather conditions. January 1st is one of the days with the largest number of visitors. Season 2004-2005 is the one that shows the biggest crowd on Sundays, with a total of 11610 persons and a record of 3895 attendances on Sunday 16. January 2003 has the biggest attendance (34667 persons). During the 2000-2001 season, characterized by high temperatures the quantity of people attending the resort was not massive. For example, on January 15 (Monday), with more than 37 °C only 951 persons attended. On the previous day, when the temperature reached only 28 °C, the visitors were 2195. These figures show that the resort is mainly used on Sundays (Figure 20). These characteristics are the same for all seasons, with a gradual decrease of visitors starting in February and more noticeable in March, when most of the population ends their holidays and the school year starts.

4.2 Swimming pool waters quality

Maldonado resort is one of the few places where the inhabitants can be near the sea with their families. It shows an important number of visitors during summer and it is also used as a summer camp for children. Therefore, the resort infrastructure and the quality of the water must be well preserved. Adequate maintenance is necessary. Water quality controls are sporadic, and some samples have had values unfit for human use. When this happens, the floodgates are opened to change the water in the pool. The Bromatology Department of the Bahía Blanca Municipality carries out microbiologic controls on the waters of the swimming pool. The data supplied by the Department shows that for the 2000-2001 season, one analysis was made before the bbeginning of the activities (December 14). Its value was 40 CFU/100 ml of *Enterococci* (whereas the maximum limit is 35 CFU/100 ml). Later 13 controls were carried out showing values under the mentioned limit (Piccolo et al. 2007). On December 17 and 29 (2003) values were under the limit. However, 21 days later (January 19) a new control showed 99 CFU/100 ml and another one on January 26 indicated 100 CFU/100 ml. As these values are not suitable for human use, a frequent replacement of the pool water is recommended.

TABLE 1: *Visitors to the Maldonado resort by season and month (Source: Data provided by the Bahía Blanca Municipality)*

	2000-01	2001-02	2002-03	2003-04	2004-05
December	9055	10455	10760	3372	2615
January	28272	25344	34667	27028	34278
February	19508	18589	17367	9831	17133
March	5490	1723	4586	1017	2981
Total	**62325**	**56111**	**67380**	**41248**	**57007**

5 CONCLUSION

Different socio-economic aspects of the districts surrounding the Bahía Blanca estuary were analyzed. Three main themes arise: 1) the unequal population growth of the three districts around the estuary; 2) the continuous loss of the fishing resource, which generates serious problems for the local fishing community; and 3) the quality of the waters of the estuary for recreational activities. In regards to population, the Bahía Blanca district presents a moderate intercensus growth but with a marked decrease, following the national and provincial trends. On the other hand, the Coronel Rosales district shows a notorious decrease until 1991; and Villarino, unlike the others, presents a scarce growth during the first periods and a slight recovery in the last ones.

The continuous decrease of fish captures arise as one of the main problems for the coastal community. After an analysis of the evolution of the fishing laws of the Buenos Aires province and the decrease in the fishing of the main species at the Bahía Blanca estuary, we may infer that the main cause for the scarcity of the resource was bad management. Outside the estuary an indiscriminate capture of all species was found. This is the main reason for the marked fishery decrease inside the estuary in the last two decades. The Maldonado resort is unique in the Bahía Blanca city. Its pool es filled with waters from the Bahía Blanca estuary. In general, the bacteriological data indicated that the water is apt for public use, except on some days when the attendance was high. Therefore, a more frequent change of the pool water is recommended.

In summary, many socio economic issues were analyzed in the estuary area. Only the most significant socio-economic problems of the communities surrounding the Bahía Blanca estuary were described in this chapter. There still are many issues to be studied to help those communities. The area is developing very fast, for this reason research should continue. One of problems to analize is what will happen to the Maldonado resort when the new discharge of liquid sewage from a neighborhood under construction near the area will start to work.

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POLLUTION PROCESSES IN BAHÍA BLANCA ESTUARINE ENVIRONMENT

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1 INTRODUCTION

Natural systems dynamics, relationships and equilibrium can be significantly affected due to human interventions which alter (either eventually or structurally) their activities imbalance. Each environment has its own characteristics which determine the response against these changes. Main human processes generating the strongest effects on coastal marine systems are: non adequate use of soils (including unplanned urbanization), untreated domestic or industrial sewage discharge, harbour activities, and incorrect solid waste disposal. Simultaneous occurrence of these processes can generate different kind of impacts on natural systems, and their magnitudes will depend not only on phenomena intensity but also response ability of the considered environment.

Bahía Blanca estuary, at the southern Buenos Aires Province Atlantic littoral, is an excellent study case, because it is a large transitional environment with a great Human activity within its inner area, including most of the above mentioned processes. This environment has been particularly studied since the 1970s, including its water physico-chemical parameters, associated biological processes and pollutants occurrence. These studies allow to characterize the potential effects on the system, as well as to recognize its response ability. The present chapter includes a brief overview on the occurrence, levels and distribution trends of the main pollutants determined within Bahía Blanca estuary (i.e. heavy metals, hydrocarbons) in both the abiotic and the biological compartments. The identified trends are analyzed within an historical viewpoint, which allows pointing out evolutive processes on the estuarine environmental quality. This information is largely useful to make concrete monitoring and management programs within the estuary.

2 HEAVY METALS

The studies on heavy metals at Bahía Blanca estuary started on the early 80's, including the corresponding data on sediments, suspended particulate matter (SPM), estuarine water and biota. All these data have been obtained through the application of internationally standardized methodologies (wet acid mineralization processes followed by atomic absorption spectroscopy -AAS-) opportunely described and compiled by Botté (2005), Marcovecchio and Ferrer (2005), and De Marco et al. (2006). Most of these results were obtained within the inner area of the estuary, even additional information exist on other regions along the system (Figure 1). In all cases, the analytical quality (AQ) of the developed analysis were checked against internationally certified reference materials, provided by the National Institute for Environmental Sciences (NIES) from Tsukuba (Japan) (mussel and pepperbush tissues, marine and estuarine sediments), as well as by the National Institute of Standards & Technology (NIST) from Boulder, Colorado (USA) (mussel tissue, marine sediments). Statistical comparisons were developed using analysis of variance (ANOVA), mean values assessment (Tukey's test), correlation analysis and single linear regression analysis (Sokal and Rohlf 1979).

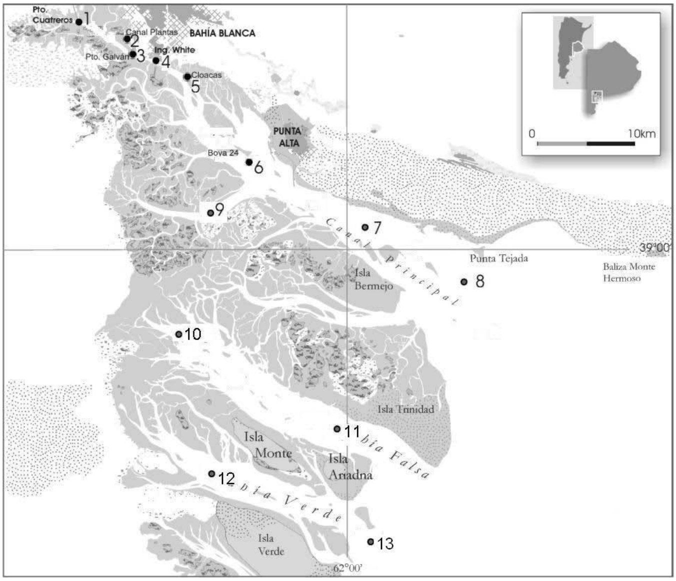

FIGURE 1: *Location of sampling stations within Bahía Blanca estuary.*

2.1 Total content of heavy metals in sediments

Mean values of the studied metals (Cd, Cr, Cu, Fe, Pb, Zn) in surface sediments from the inner area of the estuary are presented in Table 1. It is possible to observe that -even with quite different magnitudes- most of metals concentrations show decreasing trends from the head of the estuary (station #1) down to the outer area (station #8) (Figure 2). In the studied grid it was determined the highest concentrations of most metals in the inner area, between station #1 and #5, corresponding to the inner area where drain a channel receiving the industrial effluents, two harbors (Pto. Galvan, Pto. Ing.White), and the Bahía Blanca city sewage outfall discharge receiving channel, respectively. These facts are reflecting the anthropogenic impact at the inner system. Interestingly, both discharges (the industrial and the sewer ones) presented similar contents of heavy metals, and consequently were pointed out as the most important sources of these pollutants in the system. Furthermore, the differences recorded between metals concentrations on inner sediments and those from the outer zones were highly significant ($p < 0.01$), indicated that a strong dilution effect was occurring within the system and pointed out that the tidal energy influence along time (twice a day) clearly redistribute the introduced metals, modifying the background level and also homogenizing the estuary sediment concentrations.

TABLE 1: *Heavy metal concentrations ($\mu g\ g^{-1}$, d.w.) in surface sediments within Bahía Blanca estuary. (concentrations in $\mu g\ g^{-1}$, d.w., except Fe in $mg\ g^{-1}$, d.w.) (mean value $\pm$ standard deviation).*

Sampling station	Total heavy metals concentrations					
	Cd	Cr	Cu	Fe	Pb	Zn
Station #1	0.62 ± 0.15	7.86 ± 1.85	16.01 ± 1.85	18.96 ± 2.31	15.72 ± 2.86	52.96 ± 5.67
Station #2	0.77 ± 0.14	8.44 ± 1.51	14.85 ± 1.88	18.89 ± 2.32	17.76 ± 2.98	52.34 ± 5.82
Station #3	2.23 ± 0.31	8.73 ± 1.53	18.10 ± 1.88	20.68 ± 2.98	19.82 ± 2.23	60.21 ± 4.86
Station #4	2.06 ± 0.30	9.21 ± 1.64	16.46 ± 2.08	19.51 ± 2.91	18.45 ± 2.67	59.02 ± 5.43
Station #5	0.96 ± 0.24	6.45 ± 1.53	13.83 ± 2.65	16.19 ± 2.87	14.42 ± 2.30	48.06 ± 5.27
Station #6	0.15 ± 0.08	3.56 ± 1.45	5.88 ± 1.73	12.34 ± 2.38	8.53 ± 2.12	25.73 ± 4.92
Station #7	0.12 ± 0.03	3.22 ± 1.07	4.89 ± 1.51	10.12 ± 1.79	6.17 ± 1.63	23.24 ± 5.06
Station #8	0.09 ± 0.03	2.96 ± 0.78	4.95 ± 2.05	10.16 ± 1.59	7.11 ± 2.03	21.66 ± 6.14
Station #9	0.17 ± 0.04	3.38 ± 1.17	5.41 ± 1.39	13.04 ± 2.53	8.93 ± 2.06	24.46 ± 5.06
Station #10	0.15 ± 0.04	3.35 ± 1.37	5.56 ± 1.76	11.78 ± 1.38	7.16 ± 1.83	23.63 ± 4.68
Station #11	0.09 ± 0.01	2.51 ± 0.53	5.01 ± 1.93	10.04 ± 1.44	5.04 ± 1.63	19.93 ± 4.76
Station #12	0.07 ± 0.02	2.15 ± 0.44	4.61 ± 1.32	11.31 ± 1.89	5.39 ± 1.93	16.36 ± 4.38
Station #13	0.07 ± 0.03	2.27 ± 0.47	3.97 ± 0.96	10.15 ± 1.21	4.89 ± 1.32	15.66 ± 2.86

Sampling stations located on the southern coast of the estuary (station #9 to #13; Figure 2) presented, in every cases, extremely low concentrations of the studied metals (Table 1). This occurred as expected, considering their distance from the recognized anthropogenic sources of metals. In this sense, it can be pointed out that station #9 behavior appeared to be influenced by the closer stations at the Main Navigation Channel, while the other ones represented the background levels of the studied metals at the estuary (Ferrer et al. 1996, Andrade et al. 1996).

In addition, metals geochemical fractioning within Bahía Blanca finest sediments ($<$63 μm fraction) were studied at the inner area of the estuary, applying a sequential extraction method described by Lacerda et al.(1988) and Maddock and Lópes (1988). This methodology has provided information about the metal content of five geochemical fractions: (i) F_1, exchangeable adsorbed metals; (ii) F_2, oxidisable metal complexes; (iii) F_3, metals in carbonates; (iv) F_4, metals in reducible compounds; and, (v) F_5, residual metals. The obtained information is quite trascendent in terms of environmental management, considering it allows to forecast potential effects of metals by quantification of the potentially bioavailable fraction (PBF = $F_1 + F_2$) (Marcovecchio et al. 1998), which indicates the percentage of metal which could be assimilated by the organisms within this environment. Furthermore, clearly identify the percentage of non-mobile (F_5) or more mobile (F_3 or F_4) metals fractions. These results are quite interesting for the Environmental plannification, considering they provide a good approach of the real circulation of metals within the systems.

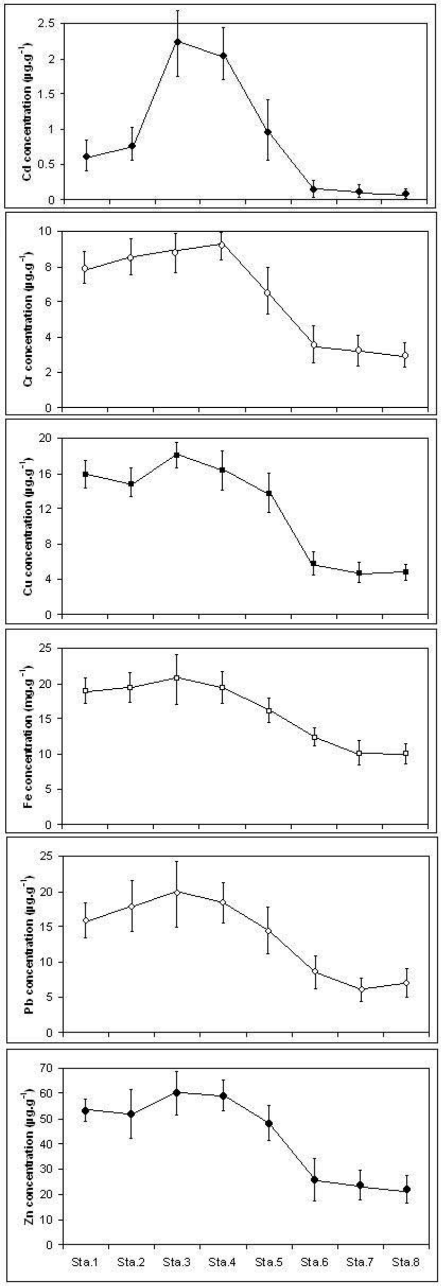

FIGURE 2: *Heavy metal concentrations ($\mu g\ g^{-1}$, d.w.) in surface sediments from the Main Navigation Channel within Bahía Blanca estuary (mean value $\pm$ standard deviation).*

In the case of Bahía Blanca estuary PBF $\sim$13% - 40% for Cd, $\sim$3% - 25% for Zn, $\sim$19% - 59% for Cu, $<$ 3% for Cr, 4.5% - 15% for Pb, and percentages close to zero for Fe were opportunely determined (Marcovecchio and Ferrer, 2005). According to Salomons and Förstner (1984) or Baisch and Wasserman (1998) the assessment of geochemical partitioning is an excellent tool to identify the influence of human activities on trace metal distribution within natural systems. Increased values of Me- F_1 and Me- F_2, as recorded in the present study, are environmentally significant because they comprise the potentially bioavailable fraction (PBF), which indicates metals that can be assimilated by organisms (Marcovecchio et al. 1998). In the same way, increasing values of Me- F_3 (in carbonates) and Me- F_4 (reducible compounds) indicate greater mobility and reactivity within the system (Baisch and Wasserman 1998). In the present case of study the results showed that most of the considered metals are in bioavailable or mobile forms, an only a limited fraction is non-mobile ones. This point is of great interest in terms of MOHID application to study de distribution of these pollutants within the estuary.

2.2 Metal distribution within estuarine zooplankton

Zooplankton plays an important role within estuarine food webs as well as in the corresponding trace metals biogeochemical cycles, transferring them up to higher trophic levels, functioning as a link from primary producers to higher order consumers. Due to this feature and their large biomass, worldwide distribution, rapid turnover, potential role as indicators of the minimal lethal concentration of metals, and ability to accumulate heavy metals, zooplankton organisms are increasingly used as biomonitors in marine systems. The first studies on heavy metals concentration in the zooplankton from Bahía Blanca estuary started in 2004, and its aim is to detect areas affected by heavy metals through analyses of metals in zooplankton. So, macrozooplanktonic ($>$ 500 μm) and mesozooplanktonic ($<$ 200 μm) species were collected for metals determinations and quantitative / qualitative analysis at the inner area of the estuary (from V.Viejo to Ing.White). Besides, water samples were taken for metals determinations in suspended particulate matter (SPM) and physico-chemical parameters were considered to detect possible effects on metal contents.

The results on metal concentrations in zooplankton are shown in Figure 3, where important seasonal variations as well as differences among the sampling stations can be observed. On the other hand, spatial and temporal heterogeneity could be related either to changing in metal bioavailability and/or to the species involved with differences in diets and metal accumulation strategies. The results of zooplankton abundance and composition were in agreement with historical data (Hoffmeyer 2004). The highest abundances of mesozooplankton and macrozooplancton were recorded in spring and summer. Within the first fraction, *Eurytemora americana* and *Acartia tonsa* were the dominant copepods respectively. In the macrozooplancton the main taxa were Decapoda larvae (summer) and the mysids *Arthromysis magellanica* and *Neomysis americana* (spring). Presently, toxicity bioassays with Cd and Pb (dissolved and particulate) are being carried out in copepods. These studies comprise lethal and sublethal effects in adults of *A. tonsa* and *E. americana* through 96 hs semi-static acute assays. Preliminary results show that Cd is more toxic than Pb at the assay conditions.

3 MERCURY

According to Marcovecchio et al. (1986), Marcovecchio and Ferrer (1999) and Ferrer et al. (2000) the surface sediments of Bahía Blanca estuary have presented mercury levels reaching up to 1700 ng g^{-1} (d.w.), and a clear distribution trend could be demonstrated: sediments of the outer area of the estuary have shown significantly lower mercury contents than those from the inner one (Table 2). Moreover it is important to emphasise that a highly significant decrease of Hg concentration within the studied estuarine sediments has occurred during the study period (1982-2005) (De Marco et al. 2006). Also the levels in suspended particulate matter (SPM) reached up to 9958 ng g^{-1} (d.w.), according to reports from the area, with the same previously described decreasing trend over time (Table 2).

Finally, when mercury was analyzed in typical organisms of the estuary (i.e., fish species) residues of this metal were recorded in most of the analyzed specimens, though at very low concentrations. This fact has been observed during the last 20 years, even though mercury values within the analyzed fish tissues have significantly decreased ($p < 0.01$) over this same period (Figure 4) (De Marco et al. 2006).

Estuarine sediments and SPM mercury values were shown to be highest during the 1980s, and these levels were related to the use of this area for the dumping of industrial effluents as well as to the proximity of the former landfill zone of Bahía Blanca city domestic solid waste at that time (Marcovecchio et al. 1986). Hg concentrations in organisms from Bahía Blanca estuary (including edible species) were also higher than those recorded in the same species from the marine coastal zone (De Marco et al. 2006). This coherence among mercury contents in abiotic and biological compartments during this period suggest that physico-chemical conditions of the system (i.e. pH, redox potential -pE-, salinity, dissolved oxygen concentration, etc) were adequate to assure the transference and circulation of this metal along its biogeochemical cycle.

The change in mercury levels within Bahía Blanca estuarine system cannot be connected with simultaneous changes in its hydrographical and chemical conditions, keeping in mind that physico-chemical properties of the estuary have remained stable within their historical range of variation (Freije and Marcovecchio, 2004). Consequently, the occurrence of other kind of scenarios must be considered, and in this way different hypotheses could be taken into account (i.e., decrease of inputs, dredging and removal of sediments, etc.) and could be tested through MOHID application to check potential areas of accumulation of mercury within the estuary.

4 ORGANOTIN COMPOUNDS

Bio-fouling is the undesirable growth of biological organisms which may cause several problems to the shipping industry. In order to prevent this phenomenon, antifouling measures have been used since the Middle Ages. In the 1970s, the chemical industry developed a highly efficient antifouling coating based on organotin compounds, especially Tributyltin Oxide (OMI 1999). A few years later researchers found that TBT was highly toxic not only for the fouling organisms but also for others, such as gastropods and oysters.

The occurrence of organotin pollution in sediments from Bahía Blanca estuary has been studied applying a reference method published by UNEP (1994) and modified by Delucchi et al. (2007). This study concerns the inner area of the estuary, mainly close to the harbours (two large commercial ports and the largest harbour of the Argentinean Army, including dry dock facilities). The results of DBT and TBT concentration (expressed as $\mathrm{ng\ Sn\ g^{-1}}$ on dry weight) obtained in the sediment samples are presented in Table 3. On the other hand, the occurrence of DBT and TBT compounds on suspended particulate matter (SPM) has also been determined, keeping in mind this compartment is one of the major sinks for pollutants in the marine environment. In this way the contaminants can be transported and distributed all over the estuary adsorbed to the SPM until environmental conditions become favorable for sedimentation as part of the fine fraction in bottom sediment.

The results show that both DBT and TBT were adsorbed to SPM all over the evaluated sites (Figure 5). The highest levels of TBT were recorded at Puerto Ingeniero White and near the Navy Base Puerto Belgrano, with values ranging between 165.8 and 389.7 $\mathrm{ng\ Sn\ g^{-1}}$. The first results of studies on the occurrence of DBT/TBT within biota from Bahía Blanca estuary were obtained in specimens of the fine snail *Zidona dufresnei* and sole fish *Paralichthys* sp., collected on the inner area of the Main Navigation Channel (Table 4). In those specimens where TBT was detected, concentrations were higher than in sediments of the same area, which led to hypothesize the occurrence of a bioaccumulation process of these compounds.

The high adsorption and octanol:water coefficients of TBT produce its association to SPM and sediments mostly. This process, which was verified in the Bahía Blanca estuary, is reversible and these matrices could act as a source of dissolved TBT. The spatial distribution of TBT demonstrates that its concentration decreases rapidly as we move further from the main source, in this case the dry docks and main ports. The broad distribution of DBT may be due to an important degradation process and its redistribution adsorbed to sediments along the Bahía Blanca estuary.

5 HYDROCARBONS

5.1 Total Petroleum Hydrocarbons (TPHs)

Total Hydrocarbons are a mix of alkanes, alkenes and alkyne-based compounds and aromatic hydrocarbons or arenes, composed entirely of carbon or hydrogen. The majority of hydrocarbon sources are crude oil, natural gas and coal (fossil fuels). Several contaminants input pathways are well known at the Bahía Blanca estuarine system: air transportation and subsequent atmospheric (dry/wet) depositions, punctual discharges to the unified collector channel of the Petrochemical pole, diffuse spills to Saladillo de García and Maldonado river, direct releases to the Main Channel and indirect discharges to the wastewater network. At present, there is a lack of historical records on this kind of pollutants. Available information includes continuous studies on Total Petroleum Hydrocarbons (TPHs) for the last ten years (Lara et al. 1995, Marcovecchio et al. 1997, 2000, 2004, 2006) on subsurface water (dissolved/dispersed) and sediments. A few records exist from the beginning of the 1980's (Sericano and Pucci 1984) on organochlorine hydrocarbon levels.

An overview of the average levels of dissolved/dispersed TPHs in seawater for the period 1996-2000 shows low values, including several records below the analytic detection limit indicating, on average, low levels of hydrocarbon pollution (Marcovecchio et al. 1997, 2000). The distribution was homogeneous, with maximum values at the outer zone of the estuary (0.20-0.30 mg l^{-1}). Subsequent studies (Marcovecchio et al. 2004, 2006) reported one magnitude order higher values for particular stations, indicating the presence of acute pollution events added to the chronic background of hydrocarbons previously registered. The registered TPHs values in sediments were low -on average- (with many values below the detection limit) (Marcovecchio et al. 1997, 2000). Subsequent values (Marcovecchio et al. 2004, 2006) recorded punctual up raises of two magnitude order, following the seawater trend level for that period.

5.2 Polycyclic Aromatic Hydrocarbons (PAHs)

Polycyclic aromatic hydrocarbons (PAHs) are ubiquitous contaminants in the marine and coastal environments. Chemically, the PAHs group belongs to the aromatic or arene hydrocarbon family. Anthropogenic activities are generally recognised to be the most important source of PAHs release into the environment. At present, there are no previously published data on hydrocarbon contents such as PAHs on estuarine waters and sediments in Bahía Blanca.

Since April 2004, a wide range of sediment PAHs concentrations was measured for PAHs (Arias et al., 2008, submitted) ranging from less than 20 ng g^{-1} to more than 10000 ng g^{-1} (dry weight) -$\sum$PAHs which includes 17 PAHs ranging from two to six condensed rings- (Figure 6). The highest levels for PAHs were observed within the sediments collected near industrial waste discharges and harbours, ranging from 1200 ng g^{-1} to more than 10200 ng g^{-1}, with an average concentration of 3571 ± 987 ng g^{-1}; for the off-shore sampling PAHs levels ranged from 15 ng g^{-1} to 1550 ng g^{-1}, indicating Bahía Blanca estuary area to be an industrialized coastal one with chronic pollution (Baumard et al., 1998, Ponce-Velez et al., 2006).

Mussels PAHs content were assessed in the zone as part of the International Mussel Watch Project (Farrington and Tripp, 1995), the levels found in that study for the outer zone of the estuary were moderate to high, corresponding to sites located near Navy/commercial ports and/or large urban centers. Recent studies (Arias, personal communication) support previously published data recording moderate PAHs levels for local mussels from the inner industrialized zone of the estuary.

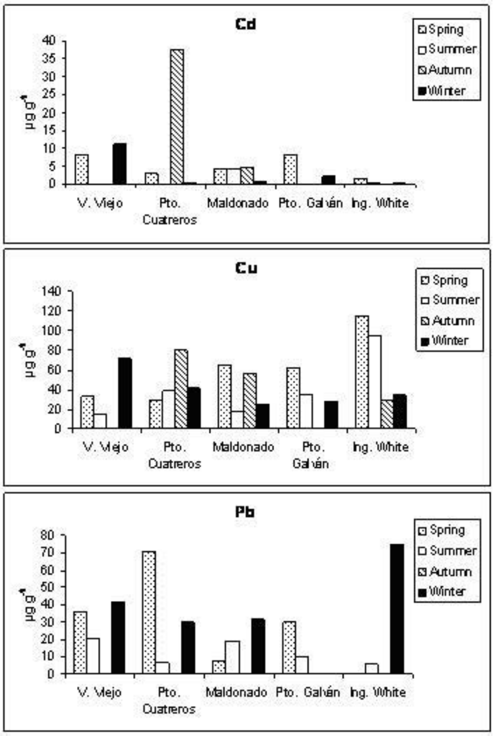

FIGURE 3: *Seasonal metal concentrations ($\mu g\ g^{-1}$, d.w.) in mesozooplankton along the sampling stations.*

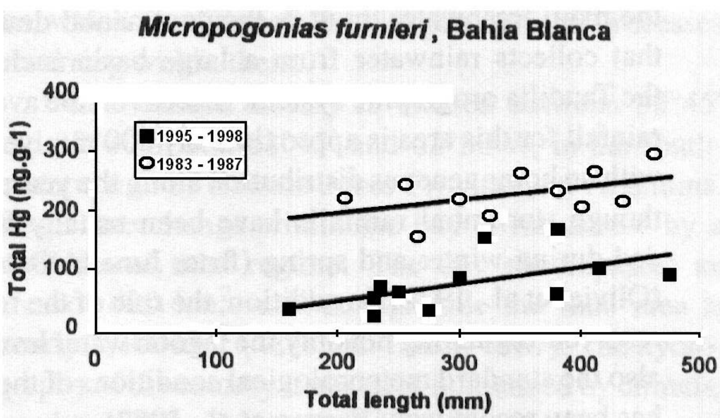

FIGURE 4: *Total mercury concentrations ($ng\ g^{-1}$, wet weight) in muscle of the Whitemouth Croaker (*Micropogonias furnieri*) from Bahía Blanca estuary (after De Marco et al. 2006).*

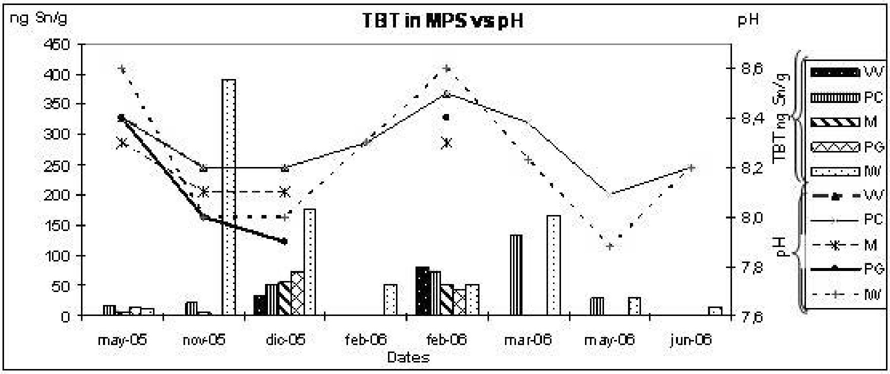

FIGURE 5: *TBT levels (ng Sn g^{-1}, dw) in suspended particulate matter and estuarine water pH from the study area (after Delucchi et al. 2007).*

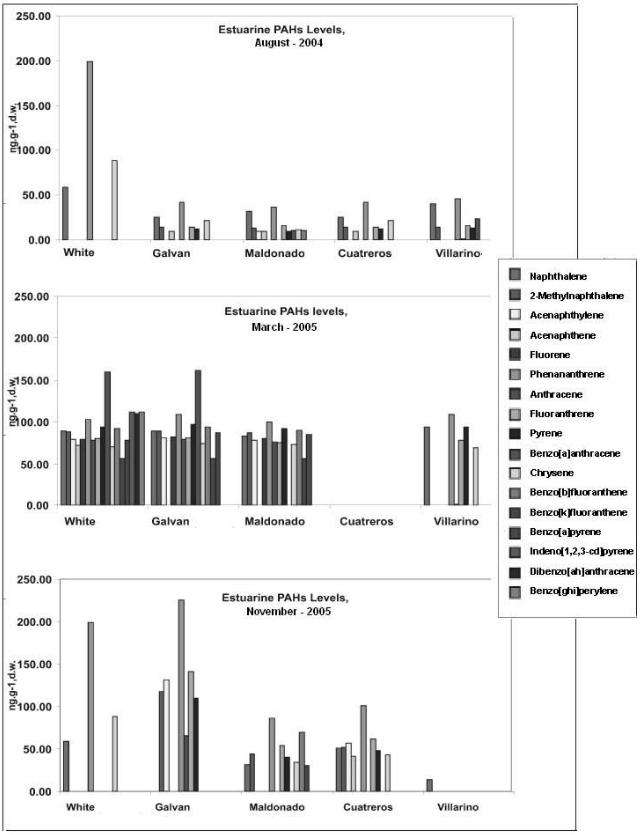

FIGURE 6: *PAH levels in Bahía Blanca estuary surface sediments over a two-years period (after Arias et al. 2008, submitted).*

TABLE 2: *Total mercury concentrations (ng g^{-1}, d.w.) in surface sediments from Bahía Blanca estuary.*

	TOTAL MERCURY (ng g^{-1}, d.w.)				
Period of study	**Surface Sediments**		**Period of study**	**Suspended Particulate Matter**	
	Outer Area	Inner Area		Outer Area	Inner Area
1982 – 1988	236 ± 84	1040 ± 388	**1982 – 1990**	3350 ± 1050	8320 ± 3620
1989 - 2000	174 ± 90	557 ± 182			
2001 - 2005	39 ± 20	98 ± 33	**1995 - 2002**	1090 ± 330	4210 ± 2520

TABLE 3: *DBT and TBT concentrations (ng Sn g^{-1}, d.w.) at the sites along the Main Navigation Channel in Bahía Blanca estuary (after Delucchi et al. 2007).*

	ORGANOTIN CONCENTRATION (ng Sn g^{-1})			
	DBT		TBT	
Sampling Station	Median	Range	Median	Range
Villarino Viejo	9.3	0 – 75.2	0	0 – 1.7
Puerto Cuatreros	17.0	0 – 74.6	0	0 – 2.0
Maldonado	14.2	0 – 67.1	0.6	0 – 35
Puerto Galván	13.8	2.1 – 67.1	1.9	0 – 170
Puerto Ing. White	13.6	3.2 - 33	1.8	0 – 11.6
Puerto Belgrano				
Site 1	63.8		21.9	
Site 2	1645.0		3288	3056 - 3227

TABLE 4: *DBT and TBT concentration (ng Sn g^{-1}, d.w.) in lyophilized tissue of estuarine organisms from Bahía Blanca.*

	ORGANOTIN CONCENTRATION (ng Sn g^{-1})	
Studied species	DBT	TBT
Zidonia dufresnei (fine snail)	0.3 – 78.0	0 – 55.6
Paralichthys sp. (flat fish)	0.2 – 5.3	0 – 25.7

6 CONCLUDING COMMENTS

1. Heavy metals were found in all the analyzed compartments, including both abiotic and biotic ones.
2. Bahía Blanca estuarine sediments have low to medium heavy metal concentrations.
3. Suspended particulate matter is the main heavy metals carrier in this estuary.
4. Organisms from the estuary (i.e. zooplankton) accumulate heavy metals, indicating that a transfer process of these elements from abiotic to biotic compartments exists.
5. Mercury has also been detected in all compartments in the estuary, but a highly significant decrease of its concentration has been recorded during the last decades.
6. Organotin compounds were also found in both abiotic and biotic compartments of the estuary, with harbour areas showing the highest concentrations.
7. Both TPHs and PAHs have been found in sediments from the estuary, at medium to high levels (this last case at point focus sites).
8. The pollution studies support the idea that the Bahía Blanca estuary is a system impacted by anthropogenic activities, and the institution of a permanent monitoring program in order to assess pollution problems is strongly recommended.
9. The use of numerical models (such as MOHID) would be a nice tool to forecast the potentiality of the identified processes within new scenarios, which could be generated from modified base conditions of the estuary.

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CHARACTERIZATION OF BAHÍA BLANCA MAIN EXISTING PRESSURES AND THEIR EFFECTS ON THE STATE INDICATORS FOR SURFACE AND GROUNDWATER QUALITY

F. LIMBOZZI AND T.E. LEITÃO

1 INTRODUCTION

Coastal zone water quality depends directly on existing local pressures but also on pressures originated in the drainage watershed basins, which effects are transported to the estuary mostly by rivers and by groundwater. This chapter characterizes the pressures originated from the watershed upgradient to the coastal area of Bahía Blanca estuary, in terms of quantity and quality. The state indicators characterization allowed a diagnosis of the reference situation in what concerns the surface water and groundwater quality that can potentially contribute to the estuary (Leitão et al. 2007). For this aim, the DPSIR (Driver-Pressure-State-Impact-Responses) methodology is applied. The state indicators used reflect the present situation of the inland waters due to the existing pressures and the actual conditions of the systems. Quantity indicators assess the changes in water amount due to changes in the pressures. For groundwater, the quantity 'state' indicator proposed is the piezometric level, while for surface water the chosen indicator is the flowrate discharge of rivers into the estuary. The changes in land use for agriculture and for livestock, as well as the placement of city sewage treatment plant are the selected scenarios for the case of Bahía Blanca estuary, being the quality indicators chosen in order to reflect the effects of these pressures.

2 STUDY AREA CHARACTERIZATION

The area considered for the watersheds characterization encompasses the northern boundary of the Bahía Blanca estuary, from the Sauce Chico river until the city of Punta Alta. This area lies in the pampean plain and includes the districts of Tornquist, Bahía Blanca and Coronel Rosales. It is characterized by two distinct drainage sectors: one with perennial and intermittent watercourses and the other only with temporary streams. The first occupies the totality of Bahía Blanca district and the western portion of Tornquist, while the latter corresponds to the Coronel Rosales district.

2.1 Climate

The climate of Bahía Blanca region is temperate, with annual mean temperatures between 14 °Cand 20 °Cand well differentiated seasons. In spite of the proximity to the Atlantic Ocean, the continental influence is evidenced by clear annual temperature oscillations. The precipitation regime confers a sub-humid or transition climatic character. The predominant winds come from northwestern quadrant, being the annual average wind speed 24 km h^{-1} for the period of 1971-1990. The annual average temperature for the same period was of 15 °C, being the average temperature in the warmest month (January) 23.2 °C, and 7.5 °Cin the coldest month (July). The minimum absolute temperature was of -11.8 °Cand the maximum absolute was 43.8 °C. According to the Thornthwaite climate classification, the weather in Bahía Blanca lies in the Dry-Subhumid group, with null or low water excess. In accordance with the Köpen climate classification, weather fits in the BS climatic type, semi-arid. The average annual precipitation for the period 1896-2000 obtained for Bahía Blanca is 584.6 mm with a predominance of rains in autumn.

2.2 Geology, soils, geomorphology and tectonics

2.2.1 Geology and soils

The study area of Bahía Blanca is included in the Colorado basin which is, in a very simplified way, characterized by faulted bedrock partially covered by a Paleozoic substratum. Since the Tertiary, no significant faults occurred and the overlying sediments were smoothly deposited towards the centre of the basin in a synclinal-like structure. Besides the quartzite outcrops from Paleozoic, the sediments from the upper tertiary and quaternary ages are the ones outcropping mostly in the study area (Bonorino 1988, Albouy 1994, Carrica 1998). From the base to the top, the stratigraphy of the region is composed by: an igneous-metamorphic complex of Precambrian-Paleozoic inferior age with depths between 1,430 and 1,920 m; quartzite rocks from Paleozoic age that form the mountain nucleus of the region; a complex set of layers of sedimentary origin, from Ceno-Mesozoic age, with different lithology and thickness, and other quaternary formations of small geographic expression and low hydrogeologic interest (Fidalgo et al. 1975, Bonorino 1988, De Francesco 1992). Soil characterization was carried out by Peña Zubiate and Maldonado Pinedo (1980), based on the soil taxonomy developed by the United States Department of Agriculture, who have found Mollisols, Entisols and Aridisols. Furthermore, the Soil Map of the Buenos Aires Province (INTA-CIRN 1989) identifies twelve soil types in the study area with Haplustolls, Argiustolls and Argiudolls being the main groups in the region.

2.2.2 Geomorphology and tectonics

In the study area, there are two main geomorphologic units: the mountains to the NE and the plain in the medium and low watershed areas. The structure of the sedimentary basin started to be shaped in Paleozoic age with the quartzite formations that, in the Permic Superior, were folded forming the present Sierras Australes de la provincia de Buenos Aires. Only in the hilly area, the Paleozoic formations outcrop, forming a positive core that is the main source of continental sediments of the region. Later, during Upper Jurassic and Medium Cretaceous times, this hydrogeologic basement was fractured and fragmented in blocs. The process was probably connected with the Atlantic Ocean opening and it resulted in an extensive fault system N-S and E-W that gave rise to gradients of 100-300 m between blocks that can even reach 1,000 m, accordingly to seismic information (Bonorino 1988).

The thick sedimentary cover started during Mesozoic times. By then, the basin was subsiding with no special orogenic period, but having some bedrock faults responsible for the slope of the sediments deposited. The main subsiding period happened during Miocene period when a sea transgression occurred. The basin filling occurred in the upper Miocene age, represented by the continental deposits, and in the Pleistocene there were a set of sea transgressions and regressions sediments that covered the wide valleys. During the Holocene there was a lift in the continental area due to a marine regression that allowed the outcrop of the Pleisto-Holocene marine sediments and the deepening of the rivers due to the decrease of its base level. From a geomorphologic point of view, the coastal zone is a platform of marine abrasion built over the plio-pleistocenic sediments carried in the last transgression in the Holocene period (Sala et al. 1985). The upper tertiary and quaternary formations form the phreatic aquifer of post-Pliocene age studied and presented in this chapter.

2.3 Hydrology

The study area is characterized by a hydrologic year that starts in the end of summer, in February, when the maximum precipitation occurs. Before this period the water reserves have reached a minimum and the drainage network in the upper watershed is partially inactive with the evapotranspiration processes being responsible for important losses of water. The small fraction of water not subject to evapotranspiration constitutes the soil water with diminutive drainage. Winter is the season with less rainfall. Nevertheless, the humidity in the watershed is high due to the low evapotranspiration rate and to the effect of aquifer recharge that also feeds the rivers, producing a rather high flowrate. Spring is the second rainiest season. The watershed has good water storage allowing a rapid surface runoff. The end of spring is marked by rare precipitation events that are summed up to a high evapotranspiration rate which are responsible for the decrease in water reserves, clearly seen in the river flowrate. This phenomenon has its clearest expression in summer.

The hydrodynamic conceptual model of the regional phreatic aquifer was defined by Bonorino (1988), Albouy (1994), Carrica (1998) and Bonorino et al. (1996), and synthesized in Bonorino et al. (2001). The recharge is done with the precipitation surplus in the entire watershed, being the most important sector the hill area and decreasing towards the coastal area. It is likely that some lateral and vertical circulation to the deeper aquifers also occurs. Therefore, the regional circulation scheme considers one preferential area of recharge close to the hills (between 7 and 8.5% of the rain, according to Carrica and Lexow 2004), one circulation area in the plains and a discharge area in the coastal area.

2.4 Hydrogeology

The main hydrogeological formation is ”La Norma”, which belongs to the Miocene superior-Pliocene (De Francesco 1992). This is the most interesting system in terms of continuity in water transmission, constituting the studied phreatic aquifer of this region (Carrica et al.

2003). It forms the lower limit of the system in the study area and outcrops in several parts of the basin. Its lithology is mainly fine sand to clay with a calcium carbonated cement as well as calcareous levels, known as ”Sedimentos Pampeanos” (Fidalgo et al. 1975). The mineralogical composition of these sediments is quartz and alkaline feldspars and volcanic sediments, with the smaller fraction composed of calcite, illite, and montmorillonite (Bonorino et al. 2001). This is a multi-layer system with aquifer-aquitard levels that is regionally considered with a homogeneous transmissivity. In some parts of the coastal area, from 5 m above the sea level until the bottom of the Bahía Blanca estuary, the pampean sediments are covered by sand, limes and clay formations of marine environment denominated ”Maldonado Formation” (Fidalgo 1983), a Pleistocene superior formation also known as post-pampean. Its thickness can reach 15-20 m close to the main channels and decreases towards the continent until it disappears some 4 to 5 km from the coast. This formation has very low hydraulic gradient and a hydrodynamic where vertical movements dominate.

The maximum thickness of the overall aquifer formation is less than 200 m, decreasing towards the hill side of the Sierras Australes (Bonorino et al. 2001). Regionally it is difficult to define its base and the continuity of the different sedimentary layers. This aquifer is considered in close hydraulic connection with the surface water of the valley above due to the rapid changes in the piezometric levels after rain events as well as to the low salinity of its waters. The hydraulic gradients are very different throughout the watershed: the higher values are closer to the hills around 13 per thousand, lowering to the medium area of the watershed to values of the order of 2 to 6 per thousand, and in the south values in some parts to less than 1 per thousand. The permeability of this system is attributed to a secondary porosity from macropores and microfissures and, in agreement with the quoted authors, the estimated values are between 1 and 3 $\mathrm{m\ day^{-1}}$. Transmissivity values vary between 100 and 180 $\mathrm{m^2\ day^{-1}}$.

2.5 Main inland water contribution

The studied area covers the following main set of watercourses: Sauce Chico, Napostá Grande, Maldonado channel (from a natural drainage), Saladillo de García and Dulce, the two latter converging to Maldonado Stream. Sauce Chico and Napostá Grande rivers are the only permanent streams of the Bahía Blanca estuary region (García and García 1964) and form a hierarchized drainage network of third order (Strahler 1952). According to Heffner (2003), the surface water input to the estuary is mainly coming from Napostá Grande and Sauce Chico rivers with respectively 33 and 62% of the total flowing water. The drainage areas of Napostá Grande and Sauce Chico rivers cover 1,237 $\mathrm{km^2}$ and 1,600 $\mathrm{km^2}$ respectively and their surface water flowrate to the estuary is approximately 263,000 $\mathrm{m^3\ day^{-1}}$. Hence, these are the watersheds mainly analyzed in this work. However, due to the importance of the organic matter contribution of Saladillo de García stream to the estuary, the data from its discharge is also appraised. The estimated groundwater discharge to the estuary is about 2,000 $\mathrm{m^3\ day^{-1}}$ representing 0.6% of the total water flowing to the estuary. Sewer discharges and industrial wastewater discharges represent 23.3% and 3.2% of the total flow respectively.

3 MAIN PRESSURES: SOURCES OF DIFFUSE AND POINT POLLUTION

3.1 Urban

The main sources of point pollution in the study area are located in the urban areas of Bahía Blanca district (where 92.3% of its population is in Bahía Blanca city, with 284,776 inhabitants, and 5.4% in the towns of Ingeniero White and General Cerri) (Figure 1) and Coronel Rosales district located southeast (Punta Alta city, with a total population of 57,277 inhabitants, represents main urban centre of the district). At present, the principal sources of urban discharges to the estuary at Bahía Blanca district are the sewage treatment plant of Bahía Blanca city and the sewer of Ingeniero White town. In summer, the Maldonado Resort Area (Balneario Maldonado), close to the Maldonado Channel mouth, can also be considered a pollution source. The sewer system at General Cerri is presently under development. At the present only a small sector of this town has a sewer network which discharges to the Saladillo stream. The main sources of urban pollution of Coronel Rosales district are the sewage discharges of Punta Alta city, done through a deactivated treatment plant.

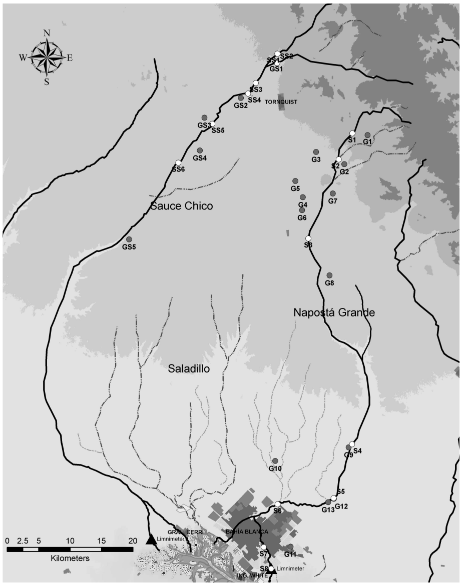

FIGURE 1: *Sampling points for groundwater (G - Napostá and GS - Sauce Chico) and surface water (S - Napostá and SS - Sauce Chico), and limnimeters location in the two main watersheds.*

3.2 Industrial

The industrial discharges concern mainly the harbour area of Bahía Blanca including the petrochemical center. Within Ingeniero White harbour, according to the type of products handled, two areas can be distinguished. The first area is designed for loading grain. The other area is designed to store general merchandise. Also the harbour has a dock for coastal fishing boats and harbour tugs. Finally, in the lower area of the harbour, the Terminal Glencore -Toepfer handles grain and fuel oil for a thermoelectric power plant. At Coronel Rosales district the most important spills are originated by the activities of the Puerto Belgrano Naval Station and the oil tanker operations near Puerto Rosales. According to their receiving bodies, four groups of industrial effluent discharges can be differentiated:

1. Discharges to the unified collector channel of the Petrochemical Pole (Petroquímica Bahía Blanca (PBB) - Polisur; Solvay-Indupa) mainly consisting in water with mercury, brines, chlorinated organic compounds, treatment plant and laboratory discharge waters with muds rich in mercury and zinc, particulate polyvinyl, Hg, HgOH and smoke particles;
2. Discharges to Saladillo de García river (TGS - Gral.Cerri) containing waste water from the treatment plant and laboratory discharges;
3. Direct releases to the Principal Channel (Petrobras; Mega; Profertil; Oil Tanking - EBYTEM) of hydrocarbons, muds with heavy hydrocarbon fractions, treatment plant and laboratory discharge waters, particulate urea and crude oil;
4. Discharges to the cloacae network (Cargill and Air Liquide) with treatment plant and laboratory discharge waters, sunflower pellets and smoke.

3.3 Livestock-agriculture

Besides the urban and industrial sectors referred to above, there is the main socioeconomic activity in the region: a mixed livestock-agriculture production system. In a large part of the study area, raising cattle is the pillar of the productive system. The production is mainly bovine in rotation with cultures climatically adapted and resistant to the soil limitations. Wheat is the main culture, followed by oats and brewing barley to a minor extent. The summer cultures are less significant with sunflower and forager sorghum prevailing. One of the zones with greater cattle-agricultural activity is located at the upper part of Napostá Grande watershed while the horticultural land use in this basin is small and there are only two horticultural producers located in two neighboring areas of Bahía Blanca.

The upper part of Sauce Chico watershed is also occupied by a large livestock development; in the middle basin there are extensive horticultural lands while in the lower basin, near General Cerri and Villarino Viejo towns, there are smaller farms, 75 horticultural farms with 45 producers managing them (Albaladejo et al. 2000). The existing information about the use and type of agrochemical applied in cultivated soils is scarce for pesticides as well as for for fertilizers. Data obtained at the upper part of Napostá Grande watershed through personal communication with producers show that fertilizer application ranges from 40-100 $\mathrm{kg\ culture^{-1}}$ of urea or PDA and pesticides are applied circumstantially for *Lepidoptera*. In the lower part of Sauce Chico basin values that range from 9 to 84 $\mathrm{kg\ yr^{-1}}$ of organophosphor pesticides, carbamates, benzimidazole and pyrethroids, among others, are utilized. Horticultural producers of

Napostá Grande make use of approximately 4 kg yr^{-1} of pesticides (courtesy of Eng. Agr. Jorge Lusto). At the moment, a monitoring programme to assess the agrochemicals used in horticulture is being carried out through the ”Programa de Promoción y Desarrollo del Cinturón Hortícola de Bahía Blanca” implemented by the Bahía Blanca Municipality in collaboration with the Department of Agronomy of the Universidad Nacional del Sur of Bahía Blanca.

4 WATER QUALITY STATUS IN THE MAIN WATERSHEDS

4.1 Methodology

The water composition is a result of a set of factors that influence its quality, e.g. climate, topography, geology, type of soils and vegetation, land cover and land use. The latter causes pressures that are responsible for the specific state of water quality. Surface direct runoff, especially in the case of high intensity rain events, has a composition close to that of the rain, or can result in infiltration through the fractures with some interconnection and appear again at the surface in springs enabling infiltration into the phreatic aquifer. The water composition depends on the paths of circulation, as well as on the time of residence in the aquifer or the river. As a result of this, water quality varies in time and space.

Data of several previous studies in the region were gathered, including some unpublished data from Autoridad del Agua (ADA- Water Authority), the organism that manages the water sources of the Buenos Aires province, and Universidad Nacional del Sur that were kindly made available to this project by their authors. The existing data were complemented with new data collected during recent monitoring campaigns which were outlined by the Instituto Argentino de Oceanografía (IADO) of Argentina and the Laboratório Nacional de Engenharia Civil (LNEC) of Portugal allowing updating the available information in terms of time, space and new elements, and obtaining a diagnosis of the present situation for groundwater and surface water state indicators (Figure 1). The overall data refer mainly to both surface and groundwater of the most important regional watersheds contributing to the estuary: Napostá Grande and Sauce Chico, and are presented in the form of the State indicators (Table 1). Furthermore additional information about other discharges is included.

4.2 Existing information and new data gathered

Table 2 presents a synthesis of the existing records about state indicators and the corresponding dates of sampling. Table 3 and Figure 1 present the statistic results of the new data recently gathered in the ECOMANAGE project. In the next sections, both a hydrologic and a hydrogeologic characterization of the reference situation is made for these watersheds in terms of the state indicators.

TABLE 1: *Quantity and quality "State" indicators proposed for groundwater and surface water in the case of Bahía Blanca estuary.*

Pressure	'State' quantity indicators		'State' quality indicators	
	Groundwater	Surface fresh water	Groundwater	Surface fresh water
Agriculture + population	Piezometric level	Flowrate	pH, E-Coli, NO_3^-, DO BOD, EC, Cl^-	E-Coli, NO_3^-, NH_4^+, PO_4^{3}, BOD, DO, EC, Cl^-, TSS, Si

TABLE 2: *Existing state indicators data for surface and groundwater of Napostá Grande and Sauce Chico watersheds.*

Surface water Source	Date	Indicators	Groundwater Source	Date	Indicators
NAPOSTÁ GRANDE					
Dirección General de Agua y Energía Eléctrica	1952-1970	Flow rate	J.C. Carrica (Hidrogeología - UNS)	1986-1989	Piezometric level pH, EC, Cl^-
C. López and O. Bisiuk (AGOSBA-Autoridad el Agua)	1993-1994 1999-2000	Flow rate BOD, COD, DO, F-Coli, N-TK, NH_4^+, P-tot	F. Limbozzi (UNS-CONICET)	2000-2002	Piezometric level pH, EC, Cl^-
F. Limbozzi (Química Ambiental -UNS)	1992-1994	EC, TDS, Cl^-, NO_3^-, NH_4^+, PO_4^{3-}, Si	F. Limbozzi, J. Marcovecchio (IADO) and T. Leitão (LNEC)	2005-2006	Piezometric level EC, TDS, DO, Cl^-, NO_3^-, NH_4^+, PO_4^{3-}
R.H. Freije and R. Asteasuain (Química Ambiental -UNS Química Marina -IADO)	1997-2006	Flow rate EC, TDS, COD, DO, Cl^-, NO_3^-, NH_4^+, PO_4^{3-}, Si			
F. Limbozzi, J. Marcovecchio (IADO) and T. Leitão (LNEC)	2005-2006	Flow rate EC, TDS, DO, Cl^-, NO_3^-, NH_4^+, PO_4^{3-}			
SAUCE CHICO					
Dirección General de Agua y Energía Eléctrica	1952-1978	Flow rate	E.R. Albouy (Hidrogeología - UNS)	1988-1990	Piezometric level pH, EC, Cl^-
C. López and O. Bisiuk (AGOSBA-Autoridad el Agua)	1993-1994 1999-2000	Flow rate BOD, COD, OD, F-Coli, N-TK, NH_4^+, P-tot	F. Limbozzi, J. Marcovecchio (IADO) and T. Leitão (LNEC)	2005-2006	Piezometric level EC, TDS, DO, Cl^-, NO_3^-, NH_4^+, PO_4^{3-}
R.H. Freije and R. Asteasuain (Química Ambiental -UNS Química Marina -IADO)	1997-2006	Flow rate EC, TDS, COD, OD, Cl^-, NO_3^-, NH_4^+, PO_4^{3-}, Si			
F. Limbozzi, J. Marcovecchio (IADO) and T. Leitão (LNEC)	2005-2006	Flow rate EC, TDS, DO, Cl^-, NO_3^-, NH_4^+, PO_4^{3-}			

TABLE 3: *Synthesis of quality and quantity state indicators for surface and groundwater at Napostá Grande and Sauce Chico watersheds.*

		Minimum	Maximum	Average	Median
TSS	**[$mg.L^{-1}$]**	194	2,673	1,003	951
Cl^-	**[$mg.L^{-1}$]**	15	780	196	155
Cond	**[$\mu S.cm^{-1}$]**	343	4,250	1,174	839
pH	**pH units**	7.3	9.5	8.3	8.3
Turb	**[NTU]**	0	72	14	4
DO	**[$mg.L^{-1}$]**	2.6	12.0	7.2	7.5
DO	**[%]**	27	114	70	66
NO_2^-	**[$mg.L^{-1}$]**	0.001	0.320	0.042	0.016
NO_3^-	**[$mg.L^{-1}$]**	1.23	72.97	15.29	8.82
PO_4^{3-}	**[$mg.L^{-1}$]**	0.019	0.296	0.110	0.075
NH_4^+	**[$mg.L^{-1}$]**	0.00	0.52	0.17	0.16
Water depth	**[m]**	3.4	28.4	12.2	9.6

TABLE 4: *Main flowrate data for Napostá Grande and Sauce Chico rivers.*

Source	Location	Lat - Long	Date	Average Flowrate [$m^3.s^{-1}$]	Minimum Flowrate [$m^3.s^{-1}$]	Maximum Flowrate [$m^3.s^{-1}$]
NAPOSTÁ GRANDE						
Dirección General de Agua y Energía Eléctrica	Cerro del Águila	- 38.20771	1936-1944	0,425	0,06	220,0
		- 62.11302	1988-1990	0,333	0,06	4,6
	Puente Canesa	-38.59059	1963-1968	0,904	0,10	2,5
		- 62.08764	1990	0,967	0,73	1,9
R.H. Freije and R. Asteasuain (UNS-IADO)	Saladero	- 38.76995 - 62.23323	1997-2002	3,030	0,72	13,5
F. Limbozzi and J. Marcovecchio (IADO)	Saladero	- 38.76995 - 62.23323	2005-2007	1,050	0,30	167,1
SAUCE CHICO						
Dirección General de Agua y Energía Eléctrica	Paso Bower	- 38.08953 - 62.28248	1940-1945	1,504	0,31	570,0
R.H. Freije and R. Asteasuain (UNS-IADO)	Villarino Viejo	- 38.72857 - 62.45240	1997-2002	4,530	0,32	28,3
F. Limbozzi and J. Marcovecchio (IADO)	Villarino Viejo	- 38.72857 - 62.45240	2005-2007	1,720	0,03	18,3

4.3 State quantity indicators

4.3.1 Surface freshwater

Table 4 presents a synthesis of the main flowrate overall data available for Sauce Chico and Napostá Grande rivers. The last available records agree with DGAyEE's historical registers (DGAyEE 1950, Carrica 1998) for the lower basin of Napostá Grande river (between Saladero and Puente Canesa flow gauge stations respectively). Equivalent relationships cannot be established in the case of Sauce Chico river because there is no past flowrate record close to its mouth. In both watercourses, data of Freije and Asteasuain correspond to registers measured monthly; therefore they have great divergences with the other records (the staff in the installation of IADO's flow gauge stations of the Hydraulics Laboratory of UNS and the kind cooperation of Eng. Juan Carlos Schefer, member of the Directory of the Water Authority of Buenos Aires province, are acknowledged). The absence of significant flowrate changes, except for seasonal and climatic variations, would indicate that pressures due to urban and agricultural activities have not increased in the last years.

4.3.2 Groundwater

Hydrogeologic observations of Napostá Grande allow inferring that in the piedmont, the phreatic aquifer presents a predominant NE-SW axis where the watercourses have an influent behavior with respect to the watertable. Southward, at the plain, the courses change their character to effluent although at short intervals this condition can change in one or both margins. Probably this is due to the general landscape rejuvenation and the differential erosion of the riverbeds. The groundwater flow at the middle part of the watershed has an N-S predominant path. The dominant type of piezometric surface has convergent radial characteristics towards the water courses and divergent feature in the watershed boundaries with parabolic depression profiles whereas inside the valley within sediments of greater permeability, hyperbolic profiles predominate.

The groundwater flow behavior at the low basin allows subdividing it in three sectors. Southeastward, an irregular piezometric surface's morphology is typical due to the presence of local zones of recharge (sandy bodies) and discharge (topographic gaps) while the northwestern area is a zone topographically depressed where the water table frequently outcrops in the rainy months and the phreatic surface has a radial morphology converging on the drainage paths, with groundwater divides of local order which separate the contributions that the river receives. Inside Bahía Blanca urban zone, the groundwater flow has been modified by engineering works, including the river canalization. Finally, the third sector lies in the area closer to the river mouth, where the water table is shallow and sometimes outcropping and its morphology has been anthropogenically modified. A series of recharge sectors with divergent radial pattern and convergent sectors of local discharges is observed and there is a predominance of vertical movements (evapotranspiration, infiltration and recharge) over the surface and groundwater runoff (Carrica 1998).

At the upper part of Sauce Chico watershed, according to Albouy (1994), the groundwater flux in the water table has a general southward direction and is characterized by a flow of short path with tendency to converge towards the watercourses that drain the aquifer. The watercourses are effluent except in the head of the ephemeral tributaries at the hilly sector, where they are influents. The morphology of the water table is radial with compartments in the watershed boundaries; in the piedmont the flow lines are divergent in the flux axis and the concavity of the piezometric isolines is oriented to the higher part of the terrain. The network of groundwater flow reveals that the hilly sector behaves like a preferential recharge area while the base level, or aquifer discharge level, is constituted by the perennial drainage of the basin.

According to Carrica (1998), in the Napostá Grande basin pluriannual and cyclic phreatic level oscillations have been observed with differences up to 25 m depth in the watershed boundaries while in the discharge zones, closer to the course, no appreciable differences were found. A similar behavior was observed by Albouy in the Sauce Chico basin. According to his work, there is no uniformity in the groundwater level variations for the watershed area, showing larger fluctuations at piedmont sectors.

Data of piezometric levels of Sauce Chico and Napostá Grande watersheds obtained by the IADO-LNEC team during 2006 display no significant variations with respect to those obtained at the end of the 80's. The possible pressures from additional pumping, either attributable to the growth of the population or to agriculture demands, seem to have no significant effect on the piezometry state quantity indicator.

4.4 State quality indicators

4.4.1 Surface water

The average values of quality indicators close to the mouths of the main watercourses were obtained by Dr. R.H. Freije and Lic. R. Asteasuain (UNS-IADO team) from 1997 to 2006 and can be found in Table 5 by their courtesy. The influence of the urban pressures identified in section 3 can be seen in the water quality of Napostá, in the samples taken monthly in the river since 1997, upstream and downstream of the city of Bahía Blanca such as in the current data. While it is possible to observe slight differences for the electrical conductivity before and after the city, the contrast is clear in the nitrate content, with the concentration of this state indicator being higher after the city (although the values are below the guideline of 50 mg l^{-1} defined by WHO (2006) for drinking water).

TABLE 5: *Average values of quality state indicators (collected by Freije and Asteasuain).*

Indicator		Conductivity [µS.cm^{-1}]	TDS [mg.L^{-1}]	DO [mg.L^{-1}]	pH [upH]	Cl^- [mg.L^{-1}]	COD [mg.L^{-1}]	NO_3^- [mg.L^{-1}]	NH_4^+ [mg.L^{-1}]	SiO_4^{4-} [mg.L^{-1}]	PO_4^{3-} [mg.L^{-1}]	Flow rate [m^3.s^{-1}]
Napostá Grande	Before city	1,323	846	9.5	8.8	155	11.1	2.96	0.43	71	0.207	3.52
	After city	1,420	895	9.4	8.6	178	17.0	8.32	0.50	58	0.274	3.03
Sauce Chico	Western branch	959	617	10.1	9.0	105	10.0	2.45	0.41	62	0.078	4.53

For Sauce Chico the existing information is more limited; unpublished data collected by Freije and Asteasuain indicate no significant changes in the pattern of electrical conductivity, nitrate and phosphorous since 1997. The values of electrical conductivity even show a slight tendency to decrease over the past decade. Concerning the nitrogen content in water, the data available allow observing that the nitrate and ammonium levels are low even in the lower basin, where the horticultural activity is intense.

The spatial distribution of the water electrical conductivity median values along the basin for the 90's and 2000 decades is shown in Figure 2. It is possible to detect a general trend for an increase in the water electrical conductivity downgradient. No significant changes are observed comparing the current situation to the results obtained in the Napostá Grande river by Carrica (1998) between 1986 and 1988 and by Limbozzi (1998) in the 90 decade, and at the upper part of Sauce Chico river by Albouy (1994) in the 80's, although the sampling points are not exactly the same.

According to Limbozzi (1998) silicate levels in Napostá Grande river oscillate between 15 and 68 mg l^{-1}. The values start to increase downstream until the middle basin end, but after that area they show a slight decrease probably due to a dilution by means of freshwater flowing from boundary farms. A similar trend can be appreciated in Table 5 before and after the city. Though equivalent information for Sauce Chico river is not available, an analogous behaviour is expected since the sediment composition in both watersheds is similar, with the weathering of its minerals the main source of silicates.

Dissolved oxygen exists in both watersheds in normal concentrations for surface water. Lower saturation percentages correspond to the hilly sectors where greater biological activity takes place; downstream values above 70% are frequent indicating the autodepuration capacity of the watercourses. Despite of the importance of the quality indicators BOD and *E. Coli* to reflect changes in the different scenarios of pressures, mainly those due to urban, agricultural and livestock activities, the existing information about these parameters is scarce and discontinuous. Monitoring for their assessment is a work in progress. Data gathered by Freije and Asteasuain about COD indicate no significant changes in both watercourses during the last decade. However, according to the average values for the obtained series, a significant increase can be observed in this quality indicator for Napostá Grande river downstream from the city, demonstrating the effects of the anthropogenic influence (Table 5). The information collected by the official authorities (Table 2) in sites near the mouth of the main streams, Sauce Chico and Napostá Grande, in the Maldonado Channel and in the Saladillo stream is presented in Table 6. According to these parameters, the main pressures to the estuary would be the sewage treatment plant of Bahía Blanca discharges, followed by Sauce Chico and Saladillo streams contributions.

TABLE 6: *Main streams water quality flowing to the Bahía Blanca estuary (AGOSBA).*

SOURCE	Lat [S]	Long [W]	BOD $mg.L^{-1}$	COD $mg.L^{-1}$	Q $m^3.h^{-1}$	BOD $Kg.d^{-1}$	COD $Kg.d^{-1}$	UNLOADING %
Sauce Chico-RN3	38.702087	62.456778	24	55	5,490	3,162	7,243	15.8
Sauce Chico-Aguará	38.714188	62.412677	60	173	1,250	1,800	5,190	11.3
Saladillo	38.688448	62.388439	75	201	970	1,746	4,679	10.2
Maldonado Channel	38.729022	62.310933	44	112	330	348	887	1.9
Napostá Grande	38.770512	62.232741	26	80	1,510	942	2,899	6.3
Sewer- I. White	38.779488	62.249626	176	457	120	506	1,300	2.8
Sewer- Punta Alta	38.908806	62.063637	101	225	400	970	2,160	4.7
Sewer- Ba Bca	38.764990	62.228290	183	390	2,000	8,780	18,700	40.7
Isaura	38.754264	62.305618	100	295	80	192	566	1.2
Pole Collector Channel	38.762642	62.300887	86	324	290	598	2,255	4.9
ESEBA Neutralization Channel	38.789140	62.249936	65	180	10.4	16	45	0.1

During the years 1993 and 1994, the Sauce Chico river was monitored both in an area contiguous to a papermill, in a resort area named Balneario Parque Norte, and in the discharge zone of the sewage treatment plant of Tornquist town, all sites located in the upper part of the watershed. Although the watershed water quality in this zone has no direct influence on the estuary, it has great importance for several touristic and recreational activities. Data of that monitoring are displayed in the Table 7 representing average values of nine sampling campaigns. Furthermore, during the years 1999 and 2000, the ADA has monitored the water quality of the main courses and the Maldonado channel and also the effluents of the collector channel of Petrochemical Pole and Tornquist sewer discharge.

Previous studies about faecal Coliforms (*Escherichia coli*) in water and sediments of the Bahía Blanca estuary (Cabezalí et al. 2004) made evident an accentuated spatial variability according to the impact of both urban and industrial discharges and the slaughterhouse discharges; however no seasonal variability was found. On the other hand, faecal indicators persist in shallow sediments, raising the risk of contamination. Moreover, bacteria strains from freshwater (Sauce Chico river) are less resistant to antibiotics than others coming from crude sewer unloads.

In the context of industrial pressure, focusing on the state indicators, levels of total and ammoniacal nitrogen, BOD, COD, DO and suspended solids for the liquid effluents of the companies placed in the Industrial Park and the Petrochemical Pole have been appraised. Also pH, electric conductivity and flowrate values were assessed. Data shown reveals that suspended solids discharge from the Polo collector channel and DQO values of TGS and Solvay-Indupa companies occasionally exceed the maximum allowed levels according to provincial legislation, while indicators of organic matter discharge and nitrogen species are regularly below them. Flow rates and electric conductivity are variable according to each company and they have no established limits, while pH values are always within recommended values (MBB 2007). Although the industrial area is only 7.7 km^2 large, the pressures originated in this zone are of great concern for the regional stakeholders, since they can have great influence on the local environment and on fishing practices.

TABLE 7: *Surface water state quality indicators analyzed by AGOSBA at upper part of Sauce Chico watershed.*

Sampling Site	Lat [S]	Long [W]	BOD [mg.L^{-1}]	COD [mg.L^{-1}]	Tot. Coli [NMP/100]	Fec. Coli [NMP/100]
Sauce Chico	38.05202	62.25212	23	61	2,033	10,665
	38.08957	62.28286	40	81	1.59E+08	7.93E+07
Sewer- Tornquist	38.08915	62.23047	71	215	3.34E+04	16,714

4.4.2 Groundwater

The electrical conductivity of the groundwater at Napostá Grande watershed has an average value of 1,174 μS cm^{-1}. Carrica (1998) describes for 80 decade values from 151 up to 7,200 μS cm^{-1}, the latter near the coastal area. Figure 2 shows the values registered in the 90 decade making evident the increase of salinity downgradient. Also, the present situation concerning groundwater values for electrical conductivity is presented for the whole study area. No big changes are observed relating to the values or the trend of higher values close to the estuary, although the sampling points are not exactly the same. The ionic concentrations tend to increase towards the estuary due to the ionic enrichment along the aquifer, from the recharge to the discharge area. The higher salinity and electrical conductivity in the discharge area are also related to the high piezometric levels found, the very low hydraulic gradients as well as the lower sediment permeability, which favors the evapotranspiration and vertical infiltration in comparison to groundwater horizontal flow (Sala et al. 1985).

Concerning chloride ion concentrations of Napostá Grande groundwater, in the 80 decade Carrica (1998) found levels from 8 mg l^{-1} at the hilly zone up to more of 1,000 mg l^{-1} closer to estuary. More recently, for a smaller data series, Limbozzi et al. (2005) reported an average value of 131 mg l^{-1} with the same trend to increase from the hilly zone to the estuary. Latest data ranges from 15 to 780 mg l^{-1} with an average value of 196 mg l^{-1}, sustaining the same trend. According to Comité Técnico Ejecutivo of Municipalidad de Bahía Blanca (CTE), at the coastal zone chloride levels as high as 27,500 mg l^{-1} can be found (MBB 2007).

There are no previous studies for groundwater in relation to nitrate levels in the main watersheds, except for an analysis in the hilly sector of the Sierras Australes where the natural average concentration of nitrate in groundwater was 12.6 mg l^{-1} (Bonorino et al. 1999) and the cited CTE's report on shallow and deep wells where values lower than 27 mg l^{-1} were found. Concerning the data gathered by the IADO-LNEC team, the higher nitrate contents, up to 89.2 mg l^{-1}, are observed in the upper part of the watershed possibly connected to agricultural practices. For Sauce Chico watershed, the evaluated upper and middle parts of the basin show low values (Figure 3).

Additional data of the other species of nitrogen and phosphate levels have been collected in the phreatic aquifer of the region and they are displayed in Table 2 and 3. Furthermore, CTE has carried out a monitoring program on several wells placed inside of Profertil lands with ammonium the only state quality indicator analyzed. The average value obtained was 2,858 mg l^{-1} with a maximum value of 9,300 mg l^{-1}. Low dissolved oxygen levels are usual in both watersheds. Lower saturation percentages correspond to the hilly sectors though the values have no regular pattern in whole area. Groundwater has often low vulnerability to pathogenic contamination due to a barrier effect provided by the overlying soil and its unsaturated zone. Nevertheless, pathogenic contamination may occur through contaminated wells or underground pollution sources, such as latrines and sewer lines or intensive livestock zones; therefore, *E. coli* monitoring in locations connected to pollution sources is currently in progress.

5 SUMMARY AND CONCLUSIONS

As a result of the analysis and interpretation of the state quality indicators, in connection with the sources of pollution (Pressures) in the region, the following conclusions can be drawn. The influence of the pressures identified can be seen in the water quality of Napostá Grande river, namely in the samples taken before and after the city of Bahía Blanca. While it is possible to observe slight differences for the electrical conductivity, the nitrate content is clearly higher after the city. Yet, their values are below the guideline of 50 mg l^{-1} defined by WHO (2006) for drinking water.

For the groundwater in Napostá Grande watershed, there is an increase of salinity downgradient, due to an enrichment of salts from evaporation and the water circulation in the aquifer along the watershed reaching a maximum in the coastal discharge area. Concerning electrical conductivity, it shows no big changes from the 90's to the present situation. However, high values in terms of electrical conductivity conferring to most waters a high risk of salinization for potential agriculture activities. Also the nitrate content is high in the area, with the highest values in the upper part of the watershed, possibly as a result of pressures caused by agriculture practices.

For the water of Sauce Chico river no significant changes are observed in the water pattern since 1997 for electrical conductivity, nitrate and phosphorus. All of them have normal concentrations for surface water. Groundwater in the Sauce Chico watershed shows ionic concentrations that tend to increase towards the estuary due to the ionic enrichment along the aquifer, from the recharge to the discharge area in Bahía Blanca estuary; very low hydraulic gradients and lower sediment permeability favors the evapotranspiration and vertical infiltration in comparison to groundwater horizontal flow.

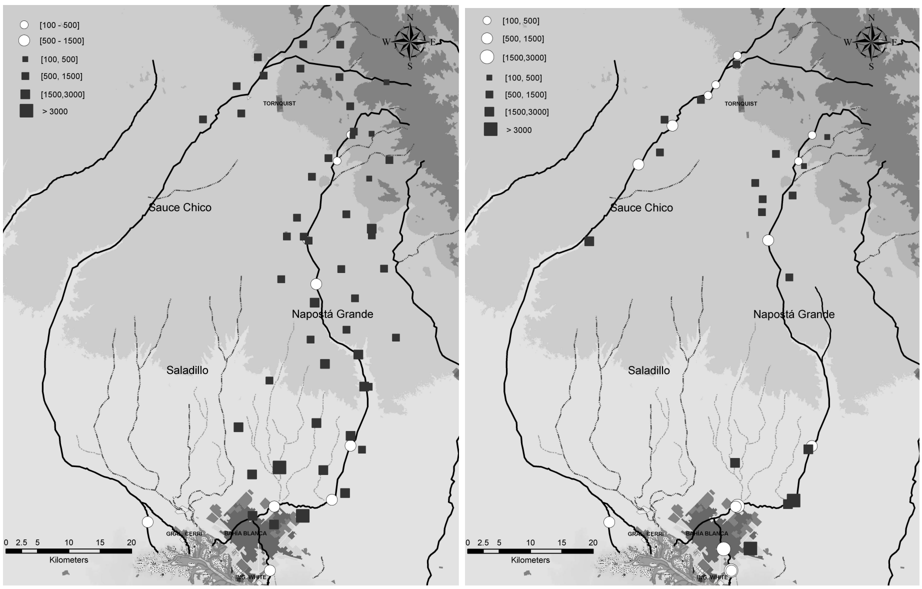

FIGURE 2: *Electrical conductivity values ($\mu S\ cm^{-1}$) for surface (dots) and groundwater (squares): in the 90's (left) and in the decade 2000 (right).*

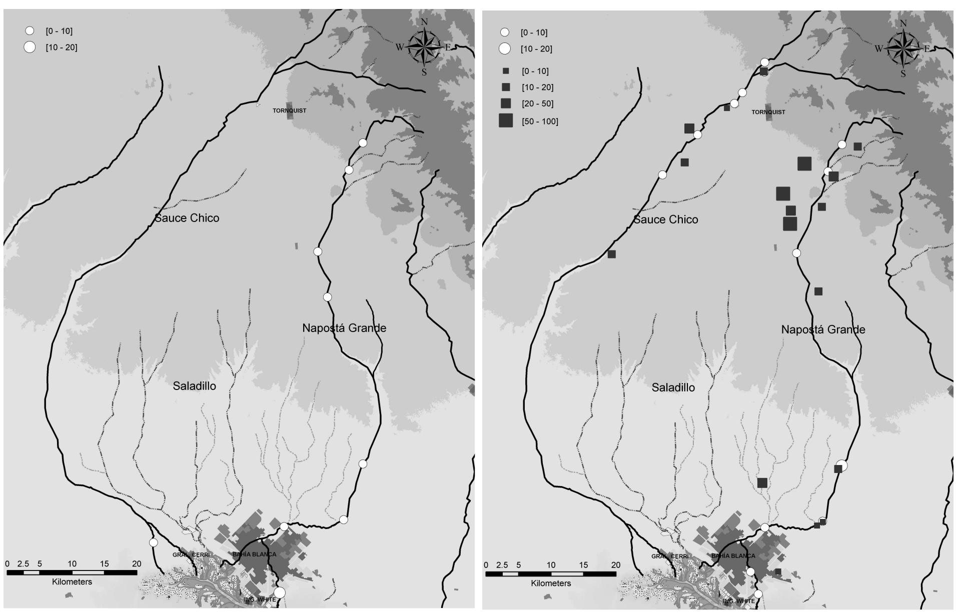

FIGURE 3: *Nitrate values ($mg\ l^{-1}$) for Napostá Grande and Sauce Chico watersheds: surface water in the 90's (left) and surface (dots) and groundwater (squares) in 2006 (right).*

At the present time piezometric levels for both watersheds show no variations with respect to those obtained at the end of the 80 decade. The effects of the pressures identified can, to some extent, be detected in the quality status of both surface and groundwater in the area. Therefore, it is advisable to carry out environmental management measures regarding inland waters protection from those identified sources and to develop monitoring in the area in order to assess the beneficial effects of those measures.

Following this work, the coastal zone is being monitored through four piezometers installed close to Sauce Chico and Maldonado Channel mouths. Also, with the aim of establishing the direct incidence that the watersheds might have in the estuary, it is proposed to continue assessing DOB, *E. Coli* and silicate levels in new and already existing monitoring sites, in their lower parts. Moreover, monitoring of some points upstream and downstream from sites of regional concern such as a paper mill, a private cemetery, a slaughterhouse and a landfill is ongoing at the moment.

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THE ESTUARINE SYSTEM OF THE AYSÉN FJORD

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1 INTRODUCTION

Chile's austral region from 41°S to 55°S corresponds to one of the planet's most complex system of fjords and channels, forming some of the largest estuarine systems of the world (Palma, 2004). In Chile, this system has a length of over 1600 km, and a surface area of nearly 240,000 km^2, with a high degree of geomorphologic and hydrographic complexity.

The coastline of southern Chile (below 41°S) is comprised of a continuous series of deep basins, channels and islands formed by glacial erosion during the Quaternary and tectonic sinking of the Central Chilean Valley. This area extends from Puerto Montt City (42°30'S) to Cape Horn (55°58'S), in a rugged region composed by a myriad of islands, fjords and channels with a relief ranging from 500 to 2000 m (Silva 2002). The bottom topography of the inlets is made irregular by frequent sills, which act as barriers for the circulation of bottom waters, favoring high sedimentation and a depression in dissolved oxygen, although anoxia is never attained (Silva, 2002). These factors, plus highly variable climatic conditions and minimal anthropogenic influence (Silva 2002), produce ecosystems that can be considered structurally and functionally unique (Palma 2004).

The Aysén fjord is located in the middle of this exceptional territory (45°12'S), with a length of 65 km, and an average width of 6.5 km (Fig. 1). The fjord was formed during the Holocene by glacial erosion, it's western part connect with the ocean through the Costa and Moraleda channels, with a median depth of 217 m and a maximum inner depth of 350 m. The fjord mouth opens in a "v" shape that contains a group of small islands. The inner part is almost isolated from direct oceanic influence, with the exception of the ocean driven deep flow (see below); its surface waters receive freshwater discharges from the Condor (50 $m\ s^{-1}$), Cuervo (100 $m\ s^{-1}$) and Aysén (500-1000 $m\ s^{-1}$) rivers (Guzman, 2004). Its main circulation forcings are: coastal ocean semidiurnal tides, local wind stress and river discharges.

The Aysén climate is influenced by polar and sub-polar winds year round, generating the humid (oceanic) characteristic of the western side of the Andes at these latitudes. The annual precipitation pattern shows a strong meridional gradient with 4000 $mm\ yr^{-1}$ on the west side of the fjords to less than 600 $mm\ yr^{-1}$ in the east side, along the border with Argentina. The average atmospheric annual temperature of the area is between 7°C and 9°C, with maximum values in January (≈18°C) and minimum (≈2°C) in July.

The physical, chemical and biological oceanography of the Chilean fjord system has been, and continue to be, the main focus of the CIMAR-Fiordos program, coordinated by the Chilean National Oceanographic Committee (Silva and Palma 2005). This research program organized and deployed 8 scientific cruises to the Chilean fjords between 1995 and 2004, five of which collected information in the area of the Aysén fjord. All CIMAR-Fiordos data can be requested online (free of charge) at the website of the Hydrographic and Oceanographic Service of the Chilean navy (`http://www.shoa.cl`). Most scientific articles related to this area have been published (in spanish) in the scientific journal ”Ciencia y Tecnología del Mar, Chile” (available online at: `http://www.cona.cl`).

2 HYDRODYNAMICS

The overall dynamics of an estuary, and its steady state salinity distribution, depend on two variables: R, the volume of freshwater that enter from the rivers during one tidal period and V, the tidal prism which corresponds to the volume brought into the estuary by the tide and removed over each tidal cycle. Depending upon the relationship between these two variables, estuaries can be grouped into classes (e.g. salt wedge, inverse, intermittent, highly stratified, etc.). Fjords correspond to highly stratified estuaries, where the R/V ratio fluctuates between 0.1 and 1.0. The Aysén fjord is characterized by a slow circulation, with a surface outflow layer, less than 10 m thick, of 0.03 m s^{-1} and an interior compensatory flow of 0.05 m s^{-1} (Cáceres et al. 2002). These characteristics generate residence times inside the fjord in excess of 500 days (Guzmán, 2004). Indeed, other authors have suggested residence time of the order of 10 months at the entrance of the Aysén fjord (Salinas and Hormazábal, 2004). Although the classical estuarine circulation is a two-layer system, Cáceres et al. (2002) have proposed that the Aysén fjord may behave as a three-layer system depending on wind conditions. Coastal tides represent one of the most important forcings in the area. The tidal wave is mixed semidiurnal, with minimum difference ($<$2 m) between high and low tides (Fierro et al. 2000).

The fjord, in terms of conservative variables such as salinity and temperature, can be divided in two layers separated by a halocline (vertical gradient of 6 psu/10m) that in its inner part reaches down to 25 m. Below the halocline salinity increases up to 31.4 psu (Silva et al. 1997, Guzmán and Silva 2002). Temperature in the fjord fluctuates between 9°C and 11°C, but at any given time it may be vertically homogeneous.

3 WATER QUALITY: OXYGEN AND NUTRIENTS DYNAMICS

Fjords' surface waters are well oxygenated through the year (6.0-8.2 ml l^{-1}), with maximum values in spring near the head of the fjord, mainly caused by the seasonal increase in primary production (see below). Consequently, nutrients (nitrate and phosphate) are low through the year with minima during spring. In deeper waters, oxygen doesn't show significant seasonal variability, even though the sub-surface minimum changes from 50 m during summer to more than 100 m in spring (Guzman 2004). Below 100 m dissolved oxygen shows a stable behavior through the year (Guzman and Silva 2002). Nutrients also show seasonal variations in the location of sub-surface maxima following the oxygen dynamics.

Along its main axis the Aysén fjord shows important deep water quality differences. For example, deep water oxygen can be higher than 5 ml l^{-1} at the fjord's mouth and lower than 3 ml l^{-1} at its head (Guzmán and Silva 2002). Nevertheless, deep waters near the head of the fjord have not been reported as anoxic.

Nutrient studies show that some of them have higher concentrations at inland fjords compare to those closer to the ocean. For example, Prado-Fiedler (2000) has shown that ammonium can reach subsurface (10 m) values above 2 μmmol l^{-1} near the head of the Aysén fjord. However, other nutrients such as nitrate and phosphate do not show large horizontal differences but a clear vertical structure. In the case of the Aysén fjord, nitrate and phosphate are located mostly below the halocline with a clear oceanic origin (Guzmán and Silva 2002). This has been corroborated by studies of the Aysén watershed that shows that the Aysén River can be classified as oligotrophic (Oyarzo 2006).

Aysen Fjord's water profiles show very steep vertical gradients for the main nutrients: phosphate, nitrate, nitrite and silicate. While deep waters remain with relative constant values through the year, upper layers ($>$20 m) show large variations between seasons, been mainly modulated by phytoplankton blooms during spring and by freshwater discharges in winter season (Guzmán and Silva 2002).

4 FJORD'S ECOLOGY

Based on available information from the oceanographic cruises Cimar- Fiordo 1, 4 and 7, Aysén Fjord's phytoplankton present different characteristics compared to other channels and fjords in the area, being mainly represented by small size species with low biomass values (Paredes 2005, Avaria et al. 1997). Its composition seems to be affected mainly by water temperature and nutrient concentrations (Avaria et al. 1997). Nonetheless, chlorophyll and size diversity is often modulated by a dynamic combination of several environmental variables (Paredes 2005). The Aysén Fjord present low biodiversity values, with a small increase near the head, where freshwater species like *Rhizosolenia eriensis* and *Melosira pseudogranulata* can be found (Avaria et al. 1999). In general, there is a good relationship between biomass (measured as chlorophyll a), cell density (Avaria et al. 1997) and size diversity (Paredes 2005) in the fjord's head. This suggests that most cells are photosintetically active and most species have similar size (Paredes 2005).

Phytoplankton communities are dominated through the year by microphytoplankton (20 to 200 μm), specially *Skeletonema costatum*, a small diatom with high growth rates attaining high densities during spring season (Avaria et al. 1997, 1999, 2004). After the spring maximum, diatoms abundance suffer a reduction during summer and winter, with smaller phytoplankton species [nano (2-20 μm) and picoplankton (0.2-2 μm)] increase their contribution. This cause the winter reduction of chlorophyll biomass, reaching values almost 10 times lower than spring (Fig. 2). Accordingly, as primary production is determined by the biomass and size structures of phytoplankton in southern fjords, productivity rates also follow this trend, with spring values even 10 times higher than winter (Pizarro et al. 2005a).

The region shows strong seasonal variations in the light climate Daily mean surface values of photosynthetically active radiation (PAR) in Aysén spring season reach 473 μM m^{-2} s^{-1} ($\approx$112.62 W m^{-2} according Lüning 1981), with a 19 m deep euphotic zone at the mouth and 10 m at the head. Summer values are close to 405 405 μM m^{-2} s^{-1} ($\approx$96.43 W m^{-2}) with a euphotic zone of 21 m at the fjord body (Pizarro et al. 2005a). Phytoplankton biomass influence over the optical conditions of the fjord waters could be larger in spring season, when biomass values reach 9.9 mg Chla m^{-2} integrated up to 50 m and with a euphotic zone 28 m deep at the fjord head, and the greater transparency of the water in summer, towards the interior of the fiord reflects already mentioned diminution of the biomass of the phytoplankton (Pizarro et al. 2005b). Moreover, the winter chlorophyll values decrease to 1.6-1.9 mg Chla m^{-2} integrated up to 50 m at the fjord mouth and to 3.5 mg Chla m^{-2} at the head, but the euphotic zone increases (i.e. 46 m at the fjord mouth).

Phytoplankton biomass variability is also explained by the intensity of herbivorous micro zooplankton grazing, which is dominated by copepods and crustaceous larvae, been abundant especially during summer and spring seasons (Avaria et al. 1999). The information about zooplankton in the Aysén Fjord is scanty, with studies focused only in one taxonomic group at a time, without ecosystemic or trophic considerations. Most of the time considering the presence, abundance and distribution of crustaceous larvae (Zama and Cárdenas 1984, Mujica and Medina 1997, Mujica 2003), fisheries (Pequeño 1981, Balbontín and Bernal 1997, 1999), mollusks (Vega et al. 2000) and gelatinous zooplankton (Palma y Rosales 1997, Palma and Aravena 2002). In general, all taxa show low diversity, which is explained by large salinity variations through the water column and high stratification, producing physical barriers on the daily vertical migration of larvae in feeding and anti-predatory behavior. Therefore, species found in the inner parts of the fjord must be adapted to this estuarine environment (Balbontín and Bernal 1997, Palma and Aravena 2002). The most abundant and widely distributed crustaceous decapods larvae belong to the infraorder Brachyura. Another important member of zooplankton communitiy is *Munida subrrugosa* (Mujica and Medina 1997, Mujica 2003), a small galatheid crab founded in large stocks that is an important source of food to marine invertebrates (Mattehews 1932).

Within zooplankton carnivorous, the greater predators of the copepods are quetognats (Alvariño 1985, Stuart and Verheye 1991, Casanova 1999), Aysén Fjord shows large densities but low species richness of quetognats during the summer and spring season, mainly influenced by salinity and temperature (Palma and Rosales 1995). The species of quetognats presents agree with the description their description of organisms strongly influenced by the salinity and the water temperature (Gasca et al. 1996, Marazzo and Nogueira 1996). Aditionally, Aysén Fjord's head has been pointed out as an important reproductive zone for southern fisheries in summer, with lower significance in spring season). Nevertheless fish larvae diversity is bigger than other taxa (Balbontín and Bernal 1999). Microbiological studies on the fjord's waters have focused on bacteria and fungi identification, especially those involved on the nitrogen cycle (Aguilera et al. 2002).

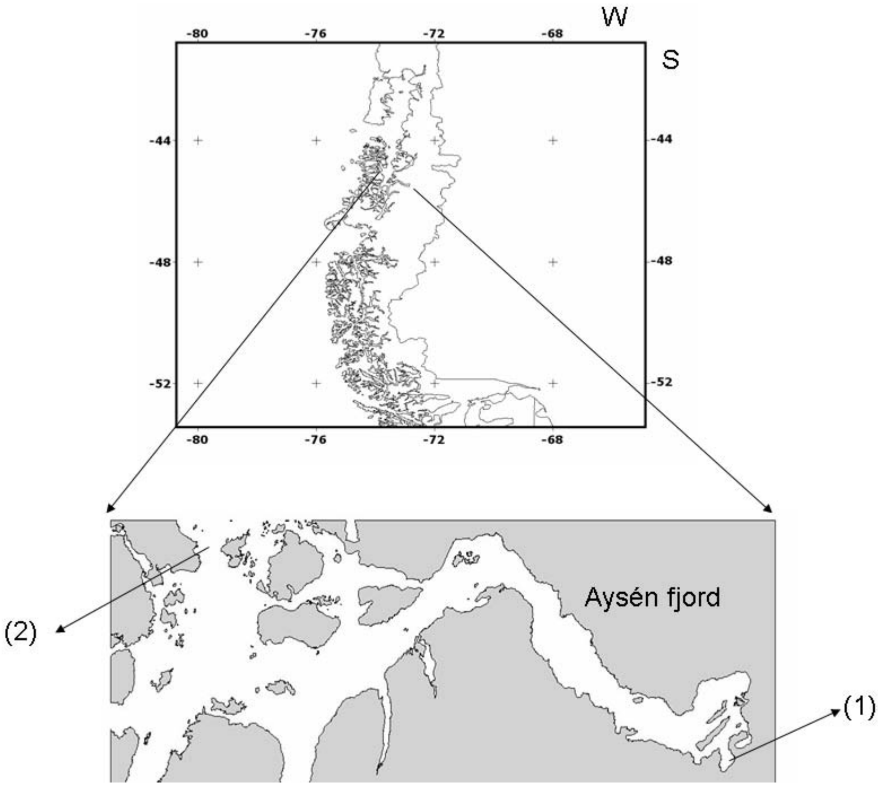

FIGURE 1: *Map of the Chile Southern Coastline, with details for the Aysen Fjord (lower map). (1) Chacabuco bay, (2) Constricción de Meninea.*

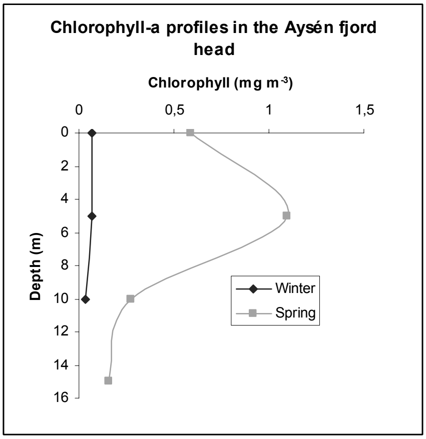

FIGURE 2: *Chlorophyll profiles of the Aysén fjord.*

The main results of these studies show bacteria getting maximum values during spring season, with values almost 100 times larger than winter concentrations for some functional groups. Samples also show that the area of the Fjord's head presents higher bacteria concentrations for almost every functional group -of the nitrogen cycle- compared with the oceanic area (Aguilera et al. 2002).

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THE ECOLOGICAL AND CULTURAL LANDSCAPE OF THE AYSÉN RIVER BASIN

M.M. YARROW AND M.A. TORRES

1 INTRODUCTION

The Aysén River basin is located in southern Chile (between 45^o and 46^o S) and covers an area of 11,456 km^2 (Figure 1). The western part of the basin contains a series of steep, highly dissected subwatersheds. The Aysén basin is somewhat unusual for Chile in that several of the main tributaries have their origin in the dry plateaus to the east of the main Andes range. Given consistent prevailing winds off the southern Pacific, the spatial distribution of precipitation within the basin is highly heterogeneous. The basin's patterns of precipitation, soil genesis, vegetation and human management and extraction all affect the nutrient cycles and export.

The vegetation of this part of the Chilean Patagonia is characterized by temperate evergreen and deciduous forest dominated by Southern Beech (*Nothofagus* sp.). Because over 46% of the basin is still forested, the area is often considered one of the most pristine regions of Chile. However, the landscape has been changed significantly by human activities since the end of the 19th century (see also Delgado and Bachmann, this volume). Colonizers to the area brought with them their cattle and cultural valuations of landscape. Spurred on by their government, these settlers cleared pastures through the deliberate setting of fires, resulting in the loss of approximately 60% of the original old-growth forest throughout the region. Currently, the landscape shows a consistent pattern of settlement and modification to the river valleys.

This chapter provides a description and analysis of the rugged and fascinating landscape found in the Aysén River basin and how historic changes in the basin are connected to the socio-political context of 20th century Chile. We base our analysis on current geographical data but also environmental and social history using a S-DPSIR framework. Given that stakeholder narratives and scenario evaluations have become a central part of ecological modeling for decision support and management purposes (see Marín and Delgado, this volume), it is critical to understand historic patterns of change.

2 GEOLOGY, TOPOGRAPHY, PEGOGENOSIS

The Andes chain in the XI Region of Chile is dominated by the Patagonian Batholith which extends about 1000 km from 41°S to 52°S and is about 200 km wide from east to west. This batholith was formed during different subduction-related volcanic episodes from the Cretaceous to the Neogene (Late Miocene). The Aysén basin is located northeast of the Chile Triple Junction, a point of convergence between the South American plate, the Nazca plate, and the Antarctic plate. Subduction along the margin of the South American plate has given rise to several volcanoes in the vicinity of the Aysén basin which have greatly influenced its geology and soils. It is worth mentioning the Macá Volcano near the headwaters of the Cuervo River and the Hudson Volcano on the southwest border of the Aysén basin. The Hudson has been recently active, with three eruptions in the last 40 years (Gutierrez et al. 2005). This high level of activity might be explained by the relatively rapid rate of tectonic convergence near the Triple Junction during the Holocene (9 mm yr^{-1}) (Stern 2008). The materials originating from volcanic activity in the southern Andean volcanic zone are typically calc-alkaline. A striking feature of the current topography in the Aysén basin (Figure 2) is that the landscape is dominated by glacial and periglacial landforms.

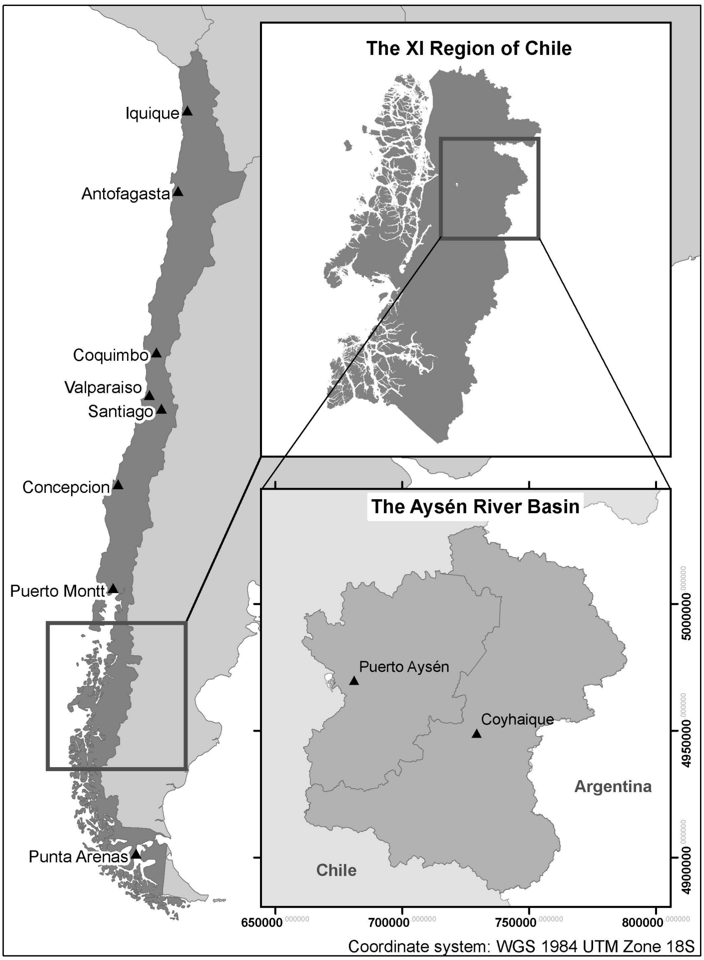

FIGURE 1: *Geographic location of the Fiordo Aysén PHES-system. The red area corresponds to Aysén river watershed.*

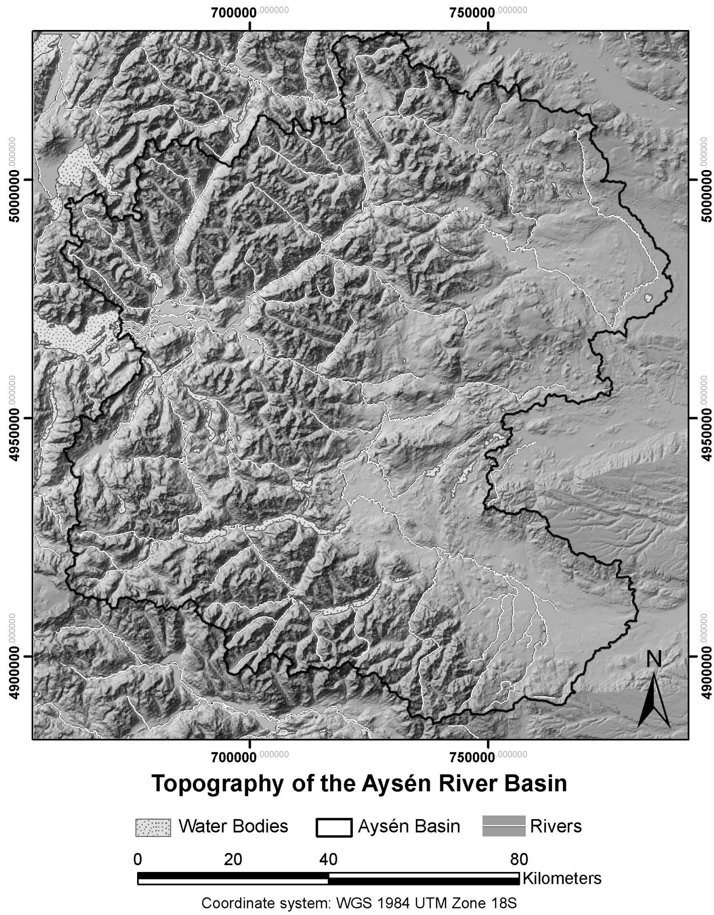

FIGURE 2: *Topography of the Aysén River Basin.*

Glaciation has been such an important process in the basin that it has masked discontinuities between different lithographic blocks (Thomson 2002). Fluvial processes are important in shaping the landscape, especially in the western part of the basin where annual rainfall can reach 5000 mm yr^{-1} (DGA). These processes have created a series of steep-sided, highly dissected valleys to the west of Coyhaique and more open plateau lands to the east. This can be demonstrated by comparing the mean percent slope for the mountainous area in the western part of the basin (45% mean slope to the west of 72°10' W) and in the eastern part (12.3% mean slope to the east of 71°50' W). The river network in the Aysén basin is somewhat unusual for Chile in that several of the main tributaries have their origin in the drier plateaus to the east of the main Andes range then flow west, essentially through the bulk of the Andes range and discharge into the Aysén fjord. This unique topography underlies the heterogeneous spatial distribution of precipitation and the diversity of vegetation communities found in the basin.

Soils of the Aysén basin are fairly young, with slow rates of soil formation. Temperatures are quite cool and, in the western part of the basin, show very moderate seasonal extremes. Puerto Aysén has a mean annual temperature of 9 °C, while in Coyhaique it is just 7.7 °C. This makes for low levels of soil flora activity. Furthermore, the erosion and mass wasting processes that occur in the mountainous western part of the basin also mean that many upland forest soils are thin and undeveloped. Consequently, deep soils are almost exclusively found in the valley bottoms. However, it may have been these deeper soils that have been most affected by fires set by the inhabitants during the early decades of the 20th century, principally in the valleys. Such extensive and long-burning fires have undoubtedly impacted the basin's soils. Although no empirical studies were done at the time, research shows that fire reduces soil organic matter and can alter nutrient cycles in the soil (Alauzis et al. 2004). In the eastern part of the basin where the topography is more open and the vegetation less dense, wind erosion affected many denuded soils after burning. According to a soil survey of the Aysén basin (IREN-CORFO 1979), the majority of soils as classified by textural class have a high percentage of sand. Some minor areas, where low velocity fluvial deposits (such as the area near the outlet of the Aysén river), have finer textures. Other areas associated with wetlands or lakes have hydric and, in some cases, organic (peat) soils. The organic material content of the basin's soils can be quite high, above 20% in many cases (Silva et al. 1999). Soils throughout the Aysén basin have been influenced by volcanism and in some areas this has lead to problems with soil acidity (Hepp 1996).

3 ORIGINAL VEGETATION

According to Gajardo (1994), in the Aysén River basin there are six native vegetation associations. This classification is based on known ecologic, geologic, geomorphologic, climatic and edaphic constraints to the occurrence of each vegetation association. Assuming that these limits were not substantially modified by pre-settlement fires or other events, we can take Gajardo's vegetation associations as a reasonable hypothesis of the composition and distribution of forests in Aysén before colonization. The following are more detailed descriptions of each association found in the basin:

- Aysén deciduous forest: the main species is *Nothofagus pumilio* (Lenga). This forest is relatively homogeneous both in floristic composition and structure. It is found in areas with large precipitation and altitude gradients.
- Puyuhuapi evergreen forest: This forest is found on gentle slopes and valleys both in continental Patagonia and the Chronos archipelago. The most frequent species is *Pilgerodendron uvifera* (Guaitecas cypress).
- Montane evergreen forest: It is found at intermediate altitudes, in mountains areas with relatively gentle slopes and in valleys. In the valleys of the Aysén basin, however, it has been practically eliminated due to ranching. The dominant species is *Nothofagus betuloides* (Magellan Coigüe).

- Patagonian steppe of Aysén: an association that occurs on gentle terrain and where tussock grasses such as (*Stipa neaei*) are predominant and are at times interrupted by shrublike patches of *Nothofagus Antarctica* (Ñirre).
- Deciduous high-elevation shrubland: This association occurs along the vegetation limit of the mountainous western part of the basin. It is composed principally of *Nothofagus Antarctica* (Ñirre) in its shrub-like growth form.
- Periglacial shrubland: This association is formed by shrub and short trees communities that are found near glaciers. The dominant species are *Nothofagus Antarctica* (Ñirre) and *Nothofagus betuloides* (Magellan Coigüe). This association is mostly localized in the southern part of the watershed.

4 HUMAN SETTLEMENT AND LANDSCAPE CHANGE IN THE AYSÉN BASIN

The modification of the landscape in Aysén, as in the rest of Chile, began to accelerate with colonization. During the 19th Century, after a series of struggles for independence from Spain, the newly liberated countries of Latin America became preoccupied with establishing their administrative boundaries. In Chile, the area south of the 38th parallel had been only lightly explored by the time of independence in 1810. The first Chilean colonizers to arrive in Aysén might be considered repatriates - they were from communities that were expelled from the Argentine Patagonia during the conflict over the political boundary at the end of the 19th century.

In the first decades of the 20th century, the Chilean government began to actively encourage the colonization of its southernmost territories by providing land grants to companies with the condition that they bring Chilean or foreign workers. This might be seen as a desperate act on the part of the government to secure its national sovereignty, given that the population of the entire region in the first decade of the 20th century was on the order of a few hundred people. Timber and cattle companies that were thus attracted to the Aysén region began to rapidly clear land in order to construct their own facilities, towns for their employees and create pastures for cattle and sheep. The tool used to clear land was fire, and vast stretches of the forested landscape were intentionally burned (Ortega and Brüning 2004, Otero 2006). During this period of colonization, no government policies dealt with land management or appropriate techniques for natural resources extraction. The idea that nature existed to be tamed by humans was culturally dominant at the time. Forests were actually seen as an obstacle to the growth of an economy based on ranching and agriculture. In the most isolated areas of southern Chile, the government would only grant land if the colonists were able to remove 70% of the forest in a period of 5 years (Otero 2006). This type of government incentive and the desire of large cattle companies to extent their production created the conditions for large-scale environmental change. Between 1933 and 1952, close to 2.8 million hectares of forest were burned in the Aysén region; this included more than 50% of the region's lenga (*Nothofagus pumilio*) forests (Quintanilla 2007).

Despite such an effort to prepare the land for ranching, the majority of the concessions granted by the government went broke due to an overestimation of the capacity of the soil to sustain productive pasture lands. Another unexpected consequence of clearing the forests was the resultant water and wind erosion - estimated at 5 tons per hectare per year (Otero 2006), that removed thousands of tons of soil from a zone characterized by slow pedogenesis. This erosion impacted rivers and lakes of the Aysén basin. An example is the sedimentation of the port that provided the livelihood for the former regional capital - Puerto Aysén. The port was forced to move to Puerto Chacabuco and the city of Coyhaique eventually became the regional capital and most populous and prosperous city in the region (Figure 3). Erosion also took its toll on cattle companies.

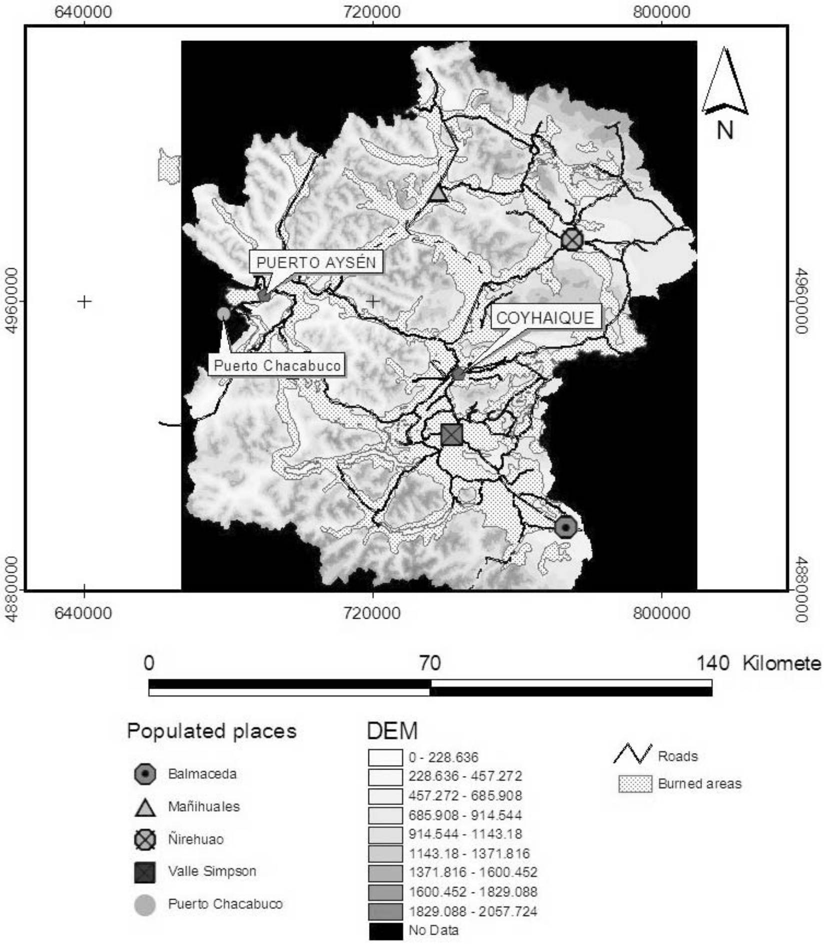

FIGURE 3: *This map can be seen the location of the most important populated places in the basin. Also, you can see how these places and roads built to connect them have been installed in the valleys and lower areas, which coincide with the burned areas.*

After leaving the region for greener pastures, their lands were spontaneously occupied by Argentine settlers who have since raised cattle and sheep in subsistence manner. This brief history of the colonization of the Aysén River basin shows how the political and cultural context of the day determined the way settlers viewed and 'managed' their land. Ultimately, the impact of colonization by fire was so large that it is clearly seen in the basin's current landscape composition and configuration (MINVU 2005).

A conceptual model often used to study environmental problems is the DPSIR model (Mateus and Campuzano, this volume). One of the key aspects of this model is its holistic approach based on the assumption that economic activities and societal behavior affect environmental health. The model simplifies this complex relationship in order to promote an understanding of the origin and consequences of environmental problems and how societies respond to them (Gobin et al 2004). Although the model has been widely used, the social variables that it incorporates are not always clear or complete - only social indicators from a natural science perspective are included, i.e. simplifying society to its economy. The model also makes no reference to temporal scales. Given that change in the landscape configuration of the Aysén River basin has been driven by socio-economic factors occurring in a particular historic context, we used a transformed version of the DPSIR model to describe the principal physical-ecological and socio-economic transformations that occurred in the Aysén basin during its colonization: the S-DPSIR (Social-DPSIR) or SEES model (Society - Economy - Environment - Society).

According to the SEES model, landscape change drivers not only include the patterns of production and consumption but also the social structure that contains them. Thus, for the Aysén River basin, the following drivers were identified (Figure 4): (1) Historical: Chilean independence and territorial struggles; (2) Socio-political: Colonization of frontier areas; (3) Organizational: the most viable productive activities, ranching and agriculture; (4) Cultural: Valuation of the natural World.

The main pressure identified is the use of natural resources, specifically the intention to arrive at a profitable use of the landscape through ranching and agriculture. The "State" of the environment, after drivers and pressures had acted, was a heavily modified landscape. Over 30% of the Aysén basin was burned, erosion became a serious problem on the hillsides, and the sedimentation of the rivers, lakes and estuaries occurred. Impacts were both ecological and socio-economic: (A) Ecological impacts: changes in the physical-chemical and ecological characteristics of the basin and in the ecosystem processes and functions; (B) Socio-economic impacts: changes in the mentality of the inhabitants and their valuation of the forests. Large economic losses after cattle companies went broke.

Responses, on the other hand, occurred at distinct levels and can be divided according to the historic moment that they occurred: (A) Immediate responses: change in the economic activities carried out by the inhabitants of the basin, change in the spatial distribution of the primary populated areas, failure of the cattle companies and their subsequent abandonment

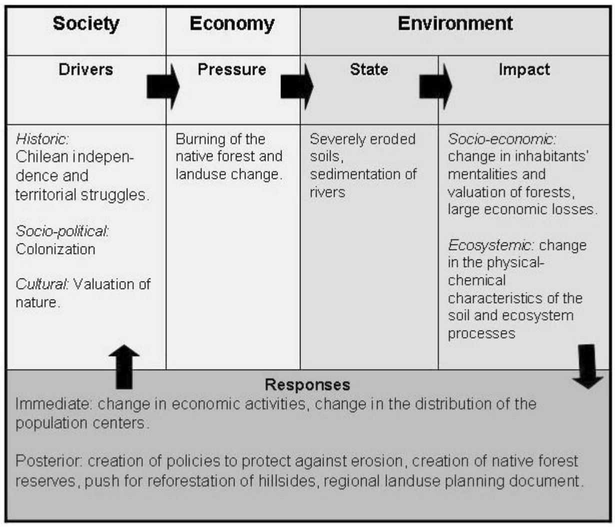

FIGURE 4: *Conceptual model of historic change in the structure of the landscape in the Aysén river basin based on the S-DPSIR framework.*

of the land; (B) Posterior (after 1938): the creation of policies to protect against soil erosion, the creation of reserves for the conservation of the native forest, and a push to reforest the hillsides. Only during 2004, these ideas were incorporated into a regional land use planning document with a goal of sustainable distributing different land uses throughout the region.

This history of the colonization and transformation of the Aysén watershed is best understood in the political context of the day and in terms of the cultural identity and worldview of the first European and Chilean settlers. This history also shows how the drivers of change (generated by society) return as negative feedback when analysis or reflections about consequences of management actions are not incorporated into the decision-making process. Landscape changes occurred in the Aysén basin can be seen by comparing the maps of the original vegetation associations and the current land use (Figure 5).

5 CURRENT CONFIGURATION OF THE LANDSCAPE

When the fires in Aysén had burned out, it was clear that the landscape had been dramatically changed. Valleys had been converted into extensive grasslands with occasional patches of bushes or small trees. The remaining forests were located almost exclusively on the sides of the mountains and the high altitude zones of the basin. The fires created a particular landscape that persists until today - the presence of burned trees trunks scattered throughout the pastures of the basin. These trunks impede a more active management of these pastures because large machinery cannot effectively operate and thus the regional government is sponsoring the removal of these trunks (Figure 6).

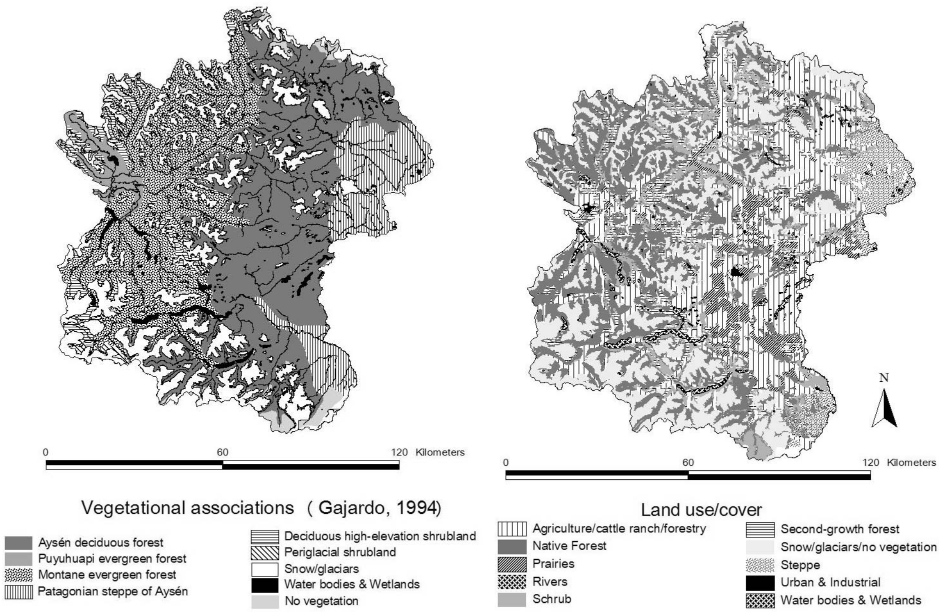

FIGURE 5: *(a) vegetation association proposed by Gajardo (1994), we take this vegetation associations as a reasonable hypothesis of the composition and distribution of forests in Aysén before colonization. (b) Actual land use cover. Comparing this figure with the number 3 we can see that areas having greater intervention are in the valleys and areas with low slope.*

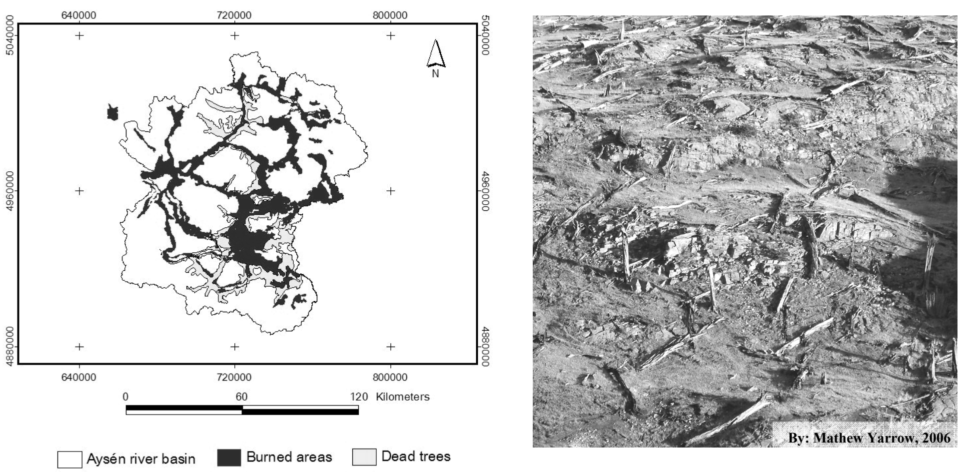

FIGURE 6: *Map of the extension of the areas with dead trunks in the Aysén basin (close to 10% of the surface area) and a picture of how these landscapes look.*

5.1 Landuse within Aysén basin

The cartographic data set used for this analysis was obtained from the national inventory of native forests (1997) (Scale 1:50,000). From the information on current land use (Figure 5, right), it is clear that the main economic activities are physically located in the valleys and steppe. As a result, the riparian and lowland forests of the Aysén basin have been significantly altered. The area of most extensive land cover change is located in the eastern part of the basin, principally in the upper Simpson River, the Coyhaique River and the Emperador Guillermo River valleys. When the percent coverage of the different land uses is calculated for burned and un-burned areas, an important difference in the amount of native vegetation coverage is detected (Figure 7). Currently, 46.4% of the watershed corresponds to forests. However, only 13.5% of the watershed is protected as part of the national system of wild protected areas. The rest of the forests can be harvested after the regional governmental office approves a management plan. The main vegetation types in the Aysén basin correspond to native forest associations:

- *Lenga forest*: represents 63.4% of the native forest cover in the watershed.
- *Evergreen forest*: As a mono-specific forest, this represents 4.2% of the forest watershed; when associated with other species the evergreen forest reaches 32.2%.
- *Puyuhuapi evergreen forest*. This forest represents only 0.2% of the forest area within the watershed.

Grasslands and pastures, used mostly for cattle, comprise 29.3% of the watershed surface followed by areas with snow and glaciers (14%) and areas with no vegetation (7.3%). A smaller fraction of the watershed corresponds to water bodies (1.6%), wetlands (1.2%), urban areas (0.1%) and agriculture land (0.1%). Figure 8 shows the allocation of the gross domestic product by the year 2002 in the Aysén region. The contribution of agriculture, cattle farming and forestry is only 3.9% (Chilecalifica program 2004).

5.2 Landscape analysis

When using landscape metrics to quantify landscape pattern, it is essential to have a central question in mind to help guide decisions about the scale and type of metrics used (Lausch and Herzog 2002, McGarigal and Marks 1994). In the context of ECOMANAGE, a central objective was to link the hydrodynamics, nutrient cycles, and human activities in the river basins. Underlying the landscape patterns of vegetation are precipitation gradients, geomorphology and soils. These patterns can be captured in certain landscape indices. In order to study the configuration of the landscape, analysis plots were established along east-west transects separated by 25 km. To aid comparisons, plots were centered on isohyets intersecting the transects (Figure 9). The seventeen landscape plots are 10 km per side; sixteen were used in the analysis, while the seventeenth was placed to capture a specific area in the east basin.

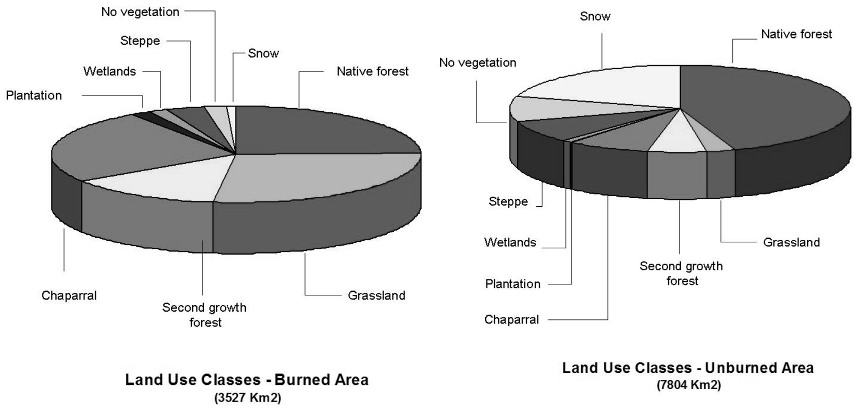

FIGURE 7: *Comparison of the proportion of land use classes in the burned and unburned areas of the Aysén basin.*

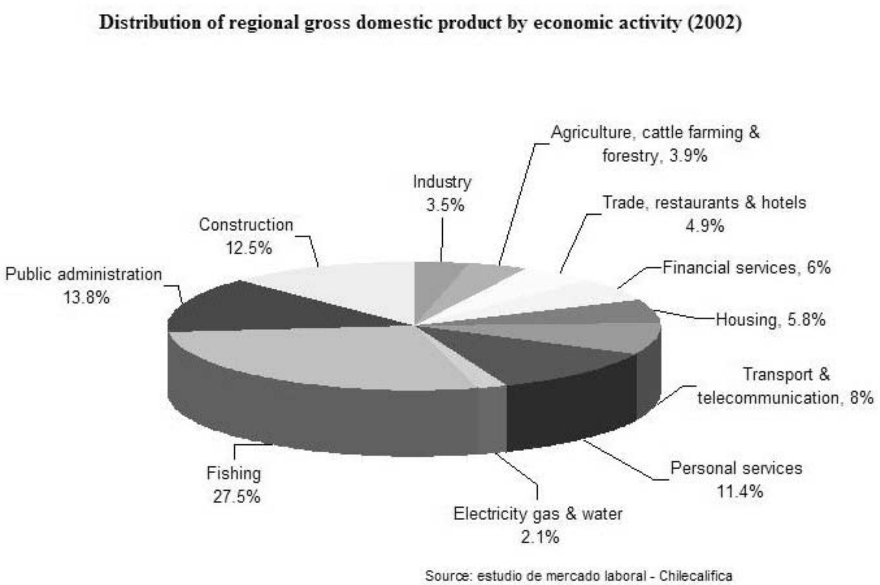

FIGURE 8: *Gross domestic product in the Aysén region, 2002.*

A GIS layer that contained information about forest structure was used in the following analysis. Land cover classes such as urban, agriculture, pasture were lumped together and considered the 'matrix'. Each plot was converted to an ArcGrid format with a resolution of 250x250 pixels (40m) and introduced into the FRAGSTATS 3.3 software. The coverage of native forest in the plots between the isohyets 1000-2000 mm H_2O yr^{-1} is quite low, reflecting the extensive human use of this area. The area between 600-1000 mm H_2O yr^{-1} shows more forest cover because it includes areas that are drier, and often colder that the area between isohyets 2000-3000 mm H_2O yr^{-1}, and therefore not as adequate for ranching activities. Further analysis revealed some notable patterns. First, and not surprisingly, the number of patches of forest in the landscapes is highly correlated to the edge density (r = 0.799, n = 16, $p < 0.0002$). Second, there is a significant negative correlation between the percent of each plot burned by forest fires and the percent of forest cover (r = -0.61, n = 16, $p < 0.0123$). Other noteworthy findings are listed below:

- There is a significant and positive correlation between the length of roads and the area of forest burned within 16 plots established in the Aysén basin. (Linear model: y = 18.57 + 1.482x; $p < 0.0034$; $r^2 = 0.44$);
- Valleys, at a basin scale, tend to concentrate human disturbance and activity, effectively fragmenting the basin into several large mountain islands mostly covered by forests, and snow/glaciers;
- The eastern side of the Aysén basin, with less dramatic topography, has generally experienced more human impact than the western side.

The above analysis points to several ideas worthy of further investigation. First, valleys of the Aysén basin tend to be the most altered areas. This indicates a spatial proximity of major human landuse and the fluvial network. The steep topography of the basin creates high flow rates and short hydrological trajectories. This fact, in conjunction with low water temperatures, suggests that rivers act more as conduits of nutrient than major processors. Thus, the connectivity of nutrient cycles in the Aysén basin and fjord is estimated to be quite high. Another corollary of the spatial proximity of human land use and the river network is that riparian areas have been some of the most altered systems within the Aysén basin.

6 NITROGEN AND PHOSPHORUS CYCLES

The isolated nature of the Aysén River basin extends to its nutrient cycles. Here, prevailing winds blow in from the south Pacific, passing over mostly uninhabited islands, fjords, and forested mountains before arriving at the basin. Thus, exogenous pollution from upwind areas (a common occurrence in the northern hemisphere) is virtually non-existent in the basin (Godoy et al. 2003, Oyarzún et al. 2004). Given that external input via dry and wet deposition is low, forests in the basin would be expected to have high internal nutrient cycling resulting in low nutrient loading to rivers. Currently, the water quality of the basin's rivers does not present any substantial problems and their waters have been classified as oligotrophic (Oyarzo 2006). However, the creation of pastures by burning and the subsequent introduction of cattle and sheep into many areas (especially valleys), have altered watershed dynamics in ways that are not yet fully understood.

Because extensive studies of nutrient cycles are not currently available for the Aysén basin, we used information from similar areas, especially Chiloé Island to generate a hypothesis of what is likely occurring in Aysén. The dissolved inorganic N (NO_3 and NH_4) in wet deposition to forested ecosystems in Puyehue Park averaged 0.059 $mg\,l^{-1}$ and contrasts sharply to DIN concentrations of 0.49 $mg\,l^{-1}$ in New England USA (Oyarzun et al. 2004, Campbell et al. 2000). While the *Nothofagus* genus, the dominant trees in the Aysén, does not have symbiotic N-fixation, non-symbiotic N-fixation takes place in the woody debris and litter on the forest floor. Pérez and colleagues (2003a) report that nonsymbiotic fixation in Chiloé ranges between 1.5-3.6 $kg\,N\,ha^{-1}\,yr^{-1}$. In general, natural N inputs to the Aysén basin are low.

Net N mineralization has also been studied in Patagonia; values between 12 to 37 $kg\,N\,ha^{-1}\,yr^{-1}$ have been reported, significantly less than forested systems with anthropogenic N inputs (Pérez et al. 1998, Pérez et al. 2003a). Pérez and colleagues (2003a) show that denitrification in temperate forests of Chiloé Island are very low: roughly 0.2 $kg\,N\,ha^{-1}\,yr^{-1}$. Although measured data are not available for the Aysén Basin, it is hypothesized that denitrification maybe higher in the western fringe where precipitation can surpass 4m yr-1, or in specific patches of wetland soils in the eastern part of the basin. Denitrification is likely limited by the fact that inorganic N is taken up rapidly by microbes and vegetation (Perakis and Hedin 2001, Perakis and Hedin 2005). Due to low available N, turnover rates of canopy foliage in Patagonian forest tend to be low: on the order of 5 years for broad leaf evergreen forests and 15 years for conifer-dominated forests (Pérez et al. 2003b).

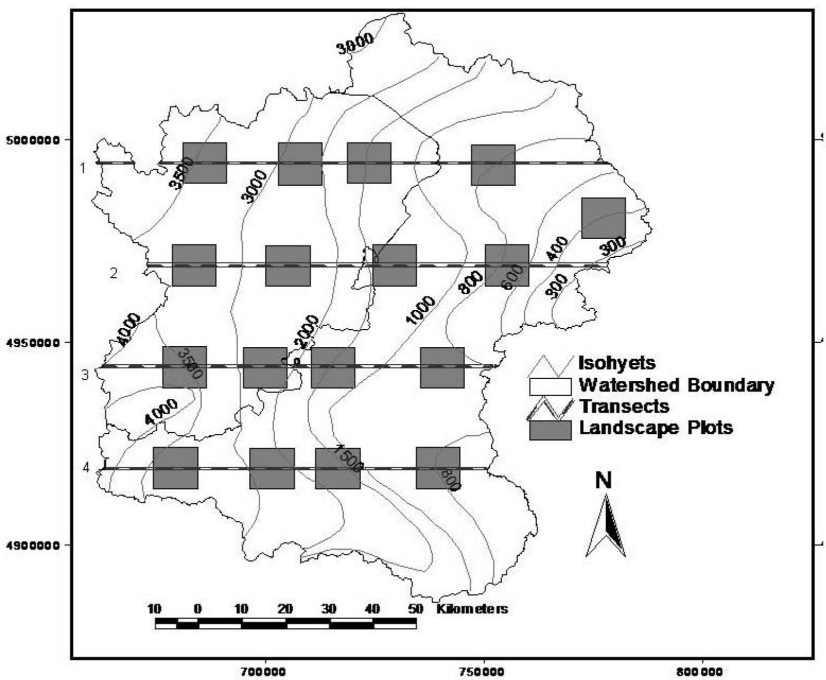

FIGURE 9: *Sampling design used in landscape pattern analysis, Isohyets in* $mm\,H_2O\,yr^{-1}$. *Transect number at left to the watershed.*

Dissolved organic nitrogen makes up the bulk of exported N in forested watersheds and appears to be controlled by hydrological factors instead of by microbial processes or plant uptake (Perakis and Hedin 2001). Unfortunately, the importance of DON in the basin's rivers cannot be evaluated because this variable is not measured by the local governmental agencies. What is known about the phosphorus cycle in *Nothofagus* forests come from outside the Aysén basin. In Chiloé one study found that the amount labile P in the soil was fairly even among different vegetation communities (Thomas et al. 1999). Several studies have found that labile P in forest soils in Patagonia is greater than the annual demand for P by the vegetation. For example, in an evergreen *Nothofagus*-dominated forest in Chiloé, the annual P uptake was 21.7 kg ha^{-1} while the available soil P measured 103 kg ha^{-1} (Vann et al. 2002). However, in the Aysén basin, P is a limiting factor in the pastures. One study reported in Hepp (1996) showed that pasture production increased 35% when P was added in the autumn (and all other nutrients where added in excess) and 43% when P was added in the spring. Additional information about the modeling of nutrient cycles in the Aysén basin can be found in Part C of this book.

7 DISCUSSION AND CONCLUSION

The history of the Aysén River basin is a textbook case of how decisions taken in a specific political, economic and cultural context can affect all aspects of a physical-ecological-social system (PHES-system) and become the catalyst of permanent changes. Six to seven decades since intentionally set fires wiped out most of the original forests in the valleys of the Aysén basin, the overwhelming majority of these areas remain deforested. Although, ranching is practiced on many of these deforested lands, effectively suppressing forest re-growth, erosion and other changes to the soils during this time appear to have made this denudation of the valleys an irreversible change. However, cultural change has occurred since the original colonization of the basin and this underlies a new set of changes that affect the landscape in different ways. For example, measures are now being taken to protect the remaining native forest, such as the creation of public and private reserves. This has allowed the regeneration of some forested areas and kept erosion in check. Today, as land available for forestry in regions to the north becomes scarce, reforestation with exotic species such (e.g. *Pseudotsuga menziessi* and *Pinus ponderosa*) has taken place. Such activity is justified in the public sphere by claims that erosion will be reduced. It is also an indication that as new productive activities become profitable in this still isolated region, the landscape will continue to be changed, creating new patterns.

Another important aspect that can strongly determine landscape structure is the regional planning documents that define the desirable development scenarios (Lausch and Herzog 2002, Herzog et al. 2001, SERPLAC 2005). This political tool attempts to determine at a regional scale the way the landscape will be organized and structured in the future. One of the main planning goals is to distribute productive activities through the territory so that secondary effects of exploiting particular resources, such as water pollution, do not negatively affect other economic activities. Although the lessons of past disaster in Aysén have to some extent been learned by the public planning agencies, the complexity of PHES-systems and the increasing needs of a growing population for a diverse array of ecosystem services make it very difficult to estimate the consequences of landuse decisions. This observation underlines why it is important to (1) take the social and environmental history of a PHES-system into consideration in decision-making process and (2) connect various parts of complex systems through modeling and scenario analysis. Decisions will ultimately be political in nature, but they should be based on the best possible input from social actors and scientists. This will go a long way toward avoiding the same dramatic pitfalls of past decisions taken in the Aysén basin.

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SOCIO-ECONOMY OF THE AYSÉN AREA

L.E. DELGADO AND P. BACHMANN

1 GENERAL CHARACTERIZATION OF AYSÉN SOCIO-ECONOMY

Human settlements in the Aysén region are known since the XVI and XVII centuries, when the area was discovered by Hernando de Magallanes on 1520, who called this area the ”Trapananda province” (CONAF-SERNATUR 2005). The area was then inhabited by two aboriginal groups: Tehuelches (or Patagones) and Alacalufes (or Kawéshkar). The former located in the ”pampas” and the later in coastal zones. Contemporaneously, those two groups left no descendents in the Aysén population.

The current Chilean political-administrative division of the territory was generated in the year 1979. The Aysén region was divided in four provinces, with Coyhaique and Aysén being the most developed among the four. The late immigration processes can be divided in two periods: (1) *the colonization from the end of the 19th century through the first half of the 20th century*. During this period there were two waves of immigration. The first, spontaneous in nature, included settlers from Argentina and the achipelago of Chiloé; the second had its origin in land concessions given to commercial enterprises and private citizens by the Chilean government with the objective of incorporating these far-flung regions into the national economy. (2) *Growth of commercial activity in the second half of the 20th century*. The commercial activities of this period have been primarily based on the exploitation of the region’s natural resources and include: fishing and aquaculture, mining, livestock production and ranching, sawmills and agriculture.

Aysén is the third political region in terms of surface area and at the same time the less populated. Its population was of only 197 inhabitants in 1907 and of 91,492 in 2002 (INE 2002). Its average density today reaches only 0.8 inhabitants km^{-2}. Eighty percent of that population lives in urban areas, mostly in Puerto Aysén and Coyhaique counties, located within the Aysen river watershed (Fig. 1 in Yarrow and Torres, this volume). Both counties show important differences in terms of demographic and socio-economic variables. Consequently, each county has been is analyzed separately. The capital city, Coyhaique, was founded in 1929 to provide support to the settlers who had been arriving since the end of the 19th century to raise livestock. Puerto Aysén was established in 1914 as a small port and as regional capital. However, at that time the habilitation of new territory for colonization was drove by a ”fire strategy” resulting in more than 3 x 106 hectares of native *Nothofagus* forest completely burned (Fig. 3 in Yarrow and Torres, this volume). Between the multiple environmental impacts caused by the fires, erosion and run-off of huge amounts of soil through the Aysén watershed provoked the embankment of the main river, making necessary the relocation of the main port and regional capital, from Puerto Aysén to Puerto Chacabuco, and from Puerto Aysén to Coyhaique (1976) respectively. At the front of these administrative changes, caused by the ecological disaster occurring hundreds of kilometers away in the upper region of the watershed, Puerto Aysén started a socio-economic deterioration process that is perceived until today (Ortega and Brunning 2000). Puerto Chacabuco, 14 km from Puerto Aysén, is now the major port in the fjord for tourist routes, and sustaining the major concentration of salmon and seafood processing factories. The majority of the salmon concessions are located along the fjord.

2 DEMOGRAPHY

2.1 Aysén County

The population of Aysén County in 1992 was of 19,090 habitants with 53% of males. In the decade between 1992 and 2002, total Aysén population increased by about 3000 people, reaching 11,853 residents (Fig. 1A). Figure 1B shows the percentages of male and females in rural and urban areas in Aysén County. The percentage of male residents is greater than the percentage of female residents in both areas, but more notably in the rural areas. This tendency is mainly due to the type of economic activity carried out in Aysén (e.g. small-scale, artisan, fishing and aquaculture), that require physical labor customarily done by men. Camps on islands and remote coastal areas, which are frequently visited by fishermen, have been dubbed by the local residents as 'islands of men'. Age class distribution in Aysén County shows the following patterns: (A) The age group between 15 and 19 years in rural areas is significantly reduced compared to neighboring age classes. This pattern is not seen the urban areas and it is likely to represent the emigration of young people to find work in other areas; (B) Contrary to what is observed in the rural areas, the urban area presents a sex ratio close to 1:1, with some age classes having more women than men.

2.2 Coyhaique County

The county of Coyhaique has a larger population than Aysén. In 1992, the total population was 43,209 inhabitants, this figure grew by about 7,000 people to a total population of just over 50,000 in 2002 (Fig. 1C). In contrast to Aysén County, the sex ratio is closer to a 1:1. Urban population is divided equally between sexes; however, in rural areas the male population is greater than the female population (Fig. 1D). The rural population in Coyhaique County shows the following patterns with respect to age and sex: (A) The age group between 15 and 19 years is primarily composed by men; (B) Male: female ratio is similar to rural areas of Aysén, but distinct from the urban zone, where the female and male populations are similar in size.

3 REGIONAL MIGRATION

The Aysén region-like other outlying regions of Chile-has among the highest rates of immigration and emigration. The rate of regional annual immigration is 2.3%. Available date shows that those females ages 15 to 19 from both Coyhaique and Aysén emigrate from the region looking for better educational and employment opportunities. According to the 1992 Census, a total of 9,067 people in region declared themselves to be immigrants from other regions of Chile or other countries. A total of 2,782 people had emigrated from the X Region (bordering the region of Aysén to the North) with 2,220 people originating in the Metropolitan Region (Santiago) and 1,425 from other countries.

4 EDUCATION

The education levels measured by the Census in the Aysén region are as follows: kindergarten and preschool, elementary and middle school (primary school), high school, humanities (referred as high school level before ∼1965), vocational high school, technical school (1-3 years post-high school), professional school (2-4 years post high-school), university ($>$ 5 years), and the special category ”never attended an educational establishment”. Kindergarten and preschool level corresponds to ages 0-6; primary or elementary and middle level to ages 6-13; normal high school and vocational high school to ages 14-18.

4.1 Aysén County

According to the 2002 Census, fifty-one percent of the total population of Aysén had achieved primary or elementary/middle school level, 27% had finished high school (31% if humanities and vocational high school are included). In 2002 only 9% of the population had a higher education (technical, professional, or university). This might be partly due to the lack of universities or professional institutes in the region as well as the income distribution of the county of Aysén.

4.2 Coyhaique County

According to the 2002 Census, 46% percent of the total population of Coyhaique had achieved primary or elementary/middle school level, 29% had finished high school (34% if humanities and vocational high school are included). Thirteen percent of the population of Coyhaique has a higher education degree (technical, professional, or university). In the last three years, there has been an increase in the number of private and state-funded establishments of high education in the region. This is likely to reduce the emigration of certain age groups and an overall improvement in quality of life indicators.

5 ECONOMY

5.1 Employment

According to the results of the 1992 census, the economically active population of the XI region of Chile was of 27,839 people. The sectors that employed most people were agriculture (4,185 people) and commerce (3,377 people), concentrating between them 27% of the region’s workforce.

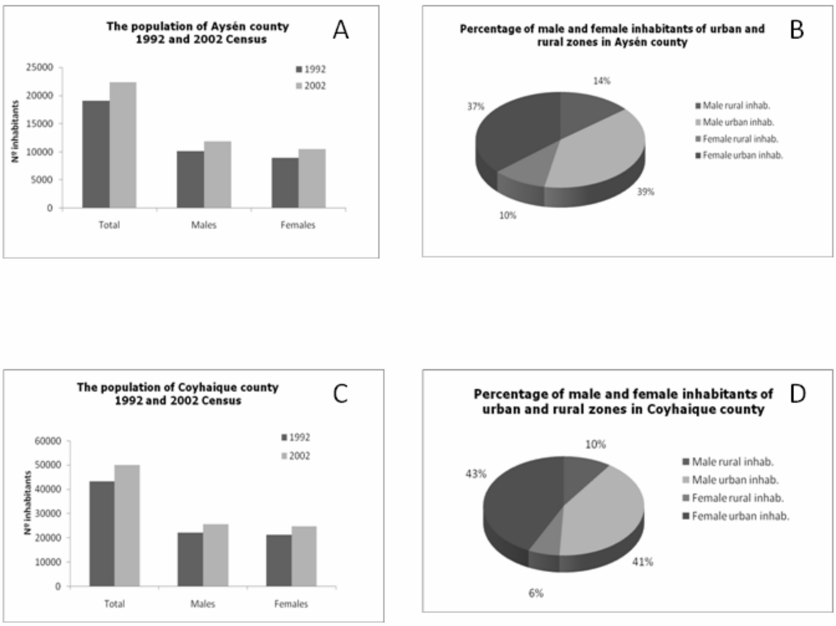

FIGURE 1: *Main demographic characteristics of the Aysén fjord population. 1A: The population of Aysén county in 1992 and 2002, total population and by sex. 1B: Percentage of male and female inhabitants of urban and rural zones in Aysén county (2002 Census). 1C: The population of Coyhaique county in 1992 and 2002, total population and by sex. 1D: Percentage of male and female inhabitants of urban and rural zones in Coyhaique county (2002 Census).*

5.1.1 Aysén County

Of the 8,733 people of working age ($<$15 years) in Aysén County (2002 Census), 75% are salaried employees, 17% unskilled workers, 13% are workers involved in agriculture, fishing, livestock raising, and 11 % skilled labor (machine and plant operators). However, only 3% of the workforce is classified as employer or manager. This fits with the relatively low educational levels and the income distribution in the county.

5.1.2 Coyhaique County

In Coyhaique County, there are 18,970 people of working age, of whom 73% are salaried workers. As in Aysén only about 3% of the workforce are bosses/managers or employers. The category with the largest percentage of the workforce is 'unskilled worker', followed by technicians and mid-level professional (representing 13% of the population). The next group is made up of officials, machine operators, craftsmen, and workers in other trades. Service workers and salespeople make up 10% of the workforce, while office employees correspond to 9%.

5.2 Distribution of the workforce among economic activities

The counties of Aysén and Coyhaique have very different local economies and provide different job opportunities to their residents. Aysén County's economy is mainly based on aquaculture and fishing and the services related to these activities. However, 19% of Aysén County's rural population works in land-based productive activities (agriculture, cattle ranching, and forestry). Coyhaique County's economy, on the other hand, is essentially service-oriented, especially public administration and services related to the city's status as regional capital. In the rural areas of Coyhaique County, 57% of the populace works in farming, ranching, and mining.

5.3 Recent evolution of the economy

From a long-term perspective, the most notable change in the economy of the Aysén region has been a shift from forestry, agriculture, and cattle grazing to fishing and aquaculture. Figure 2 shows that in 2002 fishing generated close to 25% of the GDP. Another remarkable aspect is that transportation and communication have risen in the past decade-a trend that is likely related to the growth and requirements of the aquaculture industry. Although services currently contribute less percentage-wise to the GDP than in the early 1990s, they continue to be a fundamental sector in the regional economy. In fact, the economy of the regional capital of Coyhaique is fundamentally based on the service sector. The category that includes commerce, restaurants and hotels is also important at the regional level and is directly related to tourism.

5.4 Economic projections by sectors

5.4.1 Aquaculture

Industrial salmon farming in Chile has grown spectacularly over the last 15 years. Indeed, between 1990 and 2005 the value of exports grew by almost 1,000%. Fishing and aquaculture represent more than 13% of the gross regional product and provide employment to more than 18% of Aysén labour force. Indicators of this are the statistics on the number of investment initiatives registered with the SEIA (the Environmental Impact Evaluation System administrated by the National Environmental Commission or CONAMA). In 2002, the registered initiatives in aquaculture and fishing represented a potential investment of 466 million dollars; in 2003 the registered initiatives represented 153 million dollars.

The majority of these initiatives correspond to the final 'fattening' phase of salmon production that would occur in the saline waters of Aysén Fjord. These initiatives indicate that aquaculture production in the XI region is in a process of expansion: by 2010 it is projected that the XI region (and Aysén County in particular) will produce 42% of the national salmon production (from a 20% today). This growth would be the result of an expected investment of 1400 million dollars in the region.

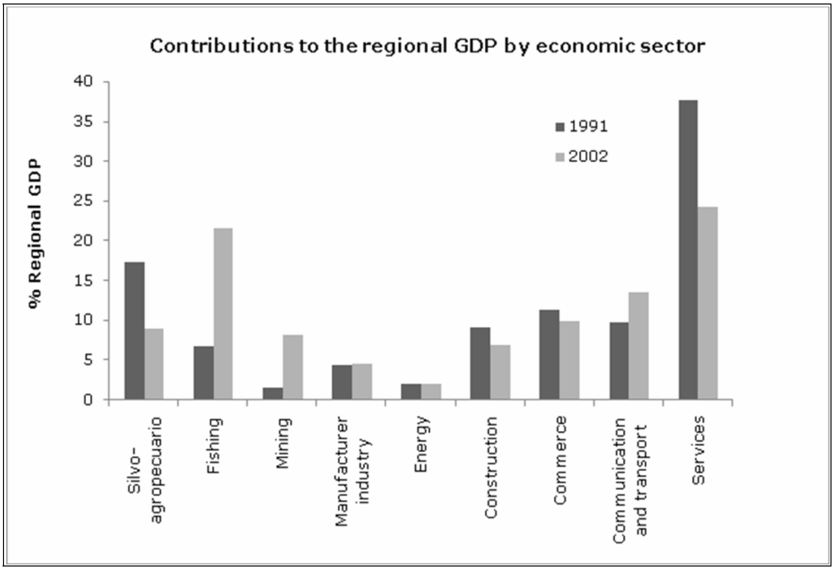

FIGURE 2: *Contributions to the regional GDP by economic sector (Silvoagropecuario = agriculture, forestry, cattle ranching).*

The production of salmon species for export occurs in several distinct phases, each requiring specific environmental conditions or ecosystem services. First, fish hatcheries involved in smolt production require contaminant-free fresh water with temperatures below 12 ^{o}C. The raising of salmon from smolt to adults require estuarine or protected coastal zones where salmons can adapt to salt water conditions and achieve a size sufficient for harvesting. The conditions exist in the Aysén watershed and Fjord to support the growth of both of these phases. Finally, with an increase in production and a desire within the industry to improve the quality of its products, it will be necessary to increase the concentration and production of processing plants. This trend in salmon production is creating a significant cluster of salmon production and processing enterprises in the area in and around Puerto Aysén and Puerto Chacabuco.

5.4.2 Tourism

The regional tourism industry has shown positive growth recently in terms of both the number of tourists arriving in the region and the number of tourist-related business and available hotel beds. In general terms, Aysén county and Puerto Chacabuco in particular, serve as a port of entry for tourists arriving by boat. Balmaceda, a small village come 60 km east from Coyhaique, has the only airport in the watershed serviced by major carriers. Most of the services and established tourist attractions are located in the Eastern part of the watershed, with Coyhaique as the main tourist hub. Aysén County doesn't lack scenery or attractions for tourists, but they are less developed than Coyhaique in terms of infrastructure and the access is limited. Regional planners are hoping to change this in the future. In both counties there are numerous lakes and rivers suitable for sport and recreational fishing (trout and salmon are the main target species). Different kinds of boating activities are available, including rafting and kayaking.

5.4.3 Agribusiness and Forestry

This sector has traditionally been small-scale and oriented toward local markets, although this is gradually changing. The cold climate restricts agriculture to certain areas of the watershed. Forestry activities include the production of firewood for home heating during winter. Harvested firewood consists of native southern beech species (*Nothofagus* spp.). The production of lumber is based on the introduction of non-native conifer species. Although this represents only 0.6% of the national lumber and pulp production, the number of hectares planted with pine is increasing. Cattle ranching have only recently been gaining access to markets outside of the region and country. Although about 1 million hectares are currently used for this activity, the production has been limited by soil erosion and nutrient leaching. Several governmental agencies have programs to boost cattle production in the watershed.

6 CONCLUSIONS

Currently, Aysén is confronted with the enormous challenge of improving the quality of life through economic growth and generation of jobs, maintaining the cultural and ecological integrity of the region. Its fast growth and immigration rates and the ubiquity of national news and cultural media have effectively pushed the local cultural patrimony into a secondary role. On the other hand, in 2005 Aysén Regional Government drafted the Regional Land Use Plan (*Plan de Ordenamiento Territorial*; PROT 2005) of Aysén Region in which the future development scenarios for the region were identified. The plan foresees that salmon farming will receive a strong impulse. The coastal area of Aysén region is expected to hold 70% of the future expansion of Chilean salmon aquaculture. Hydropower is the other strategic sector expected to rapidly develop in the next decade. Industry and commerce are classified as activities in expansion, while primary natural resources exploitation activities, like cattle, mining, agriculture and fishing as activities in decline. The main basis (almost 90%) for the development and subsistence of this region is still the utilization of natural resources. Consequently, there is a lot a pressure for the commercial use of the natural elements (e.g. native forest as wood, prairies for grazing) of the Aysen watershed. Indeed, salmon farming both within the watershed (production of juveniles) and within the fjord is the second largest in the country. On top of those activities, eco-tourism and artisan fisheries also contribute to the utilization of natural resources. It is easy to anticipate, given the variety of commercial activities and uses of resources, the emergence of conflicts of interest among social actors. However, although integrated watershed management is urgent, including the effective participation of all social actors involved, it has not yet been implemented.

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PART D

SITE APPLICATIONS: INTEGRATING THE COMPONENTS

LAND COVER ANALYSIS OF ECOMANAGE STUDY AREAS AS BASIS FOR DPSIR FRAMEWORK APPLICATIONS

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1 INTRODUCTION

Considering that the SEA (Strategic Environmental Assessment) is based on the DPSIR framework (Therivel 2004, Dalal-Clayton and Sadler 2005) it is possible to say that the land cover pattern of a given area is the result (STATE) of the choices made between different appropriate alternatives of land use considered good enough to reach an income growth (DRIVING FORCES). Nowadays, the choice among the many land use alternatives are almost always and everywhere mediated and/or supported by local, national and international policies (Nijkamp and Vindigni 2003) although the final decision is always in the hand of the land owners. As a matter of fact, after the conference Earth Summit held in Rio de Janeiro (Brazil) in 1992, the implementation of the Agenda 21 for sustainable development is considered in almost all the countries of the world (Filho 2002) at all administrative levels. It is obvious that sustainable development depends on the choices of land use; therefore there is a need of appropriate tools to support spatial decisions for ”sustainable land use” planning.

Land cover maps are the most important tools among all. These types of maps are able to show land cover patterns (spatial patterns) of specific areas at a given time and may offer more information on the state of the environment when an appropriate ecological description is available for each land cover type. The ecological description may be based on several environmental quality indicators; one of the most relevant is vegetation cover since it conditions all the ecosystem functionality. The same land use can produce different land cover types depending on several factors such as the type of ecosystems involved and their reaction to impacts, the way in which a land use is implemented and the intensity of land use. At this level land cover maps are valuable tools used to measure the divergence between the current land use locations and the most suitable ones as suggested by Dragan et al. (2008). Therefore, to define the land cover type and to measure the vegetation cover within each land cover type, remote sensing data are commonly used within GIS (Geographic Information System) technology (see Malczewski 2004 for a critical review on the use of GIS for planning purposes).

This chapter provides a summary of land cover analysis carried out at the three study areas of ECOMANAGE: Santos Estuary (Brazil), Bahía Blanca Estuary (Argentina), and Fiordo Aysén (Chile). Here land cover is analysed under the perspectives of the DPSIR framework to understand the environmental situation of the different land components of the three estuarine systems. Land cover maps have been produced for each area both for implementing soil erosion models in order to estimate the rate of sediments that can flow in the water, and to establish a monitoring system to register land cover change as consequence of socio-economic and climate change.

2 MATERIALS AND METHODS

LANDSAT images were the main source of land cover analysis to produce land cover maps and to calculate the Normalized Differential Vegetation Index (NDVI) for each land cover type and its corrections according to Brown et al. (2000) and Chen et al. (2002). Based on the considerable number of images available and their spatial and spectral resolution acceptable for an accurate vegetation cover classification (i.e. a variety of bands covering almost all electromagnetic spectrum from visible to infrared, the ones corresponding to photosynthetic activity), Landsat images were the best option for land cover analysis. Another important aspect taken into consideration and very much significant to support this choice was the fact that LANDSAT images can be downloaded free of charge at `http://glcf.umiacs.umd.edu/index.shtml`.

To obtain the land cover maps of the three study areas, supervised classification was applied based on the maximum likelihood algorithm (ENVI 2003). Training samples were selected during field campaigns (Brazil during April 2005 and March 2006, Argentina and Chile during March 2006) with the help of GPS technology. Maps of the areas, bibliographical data, aerial photographs, and other information provided by the partners were also essential for the definition of training samples and for field and laboratory verification. Specific land cover classes were defined for each study area based on the main land uses relevant to the economic activities of these areas (DRIVING FORCES).

Among the several parameters that can be obtained from remote sensing data (spectral bands) the NDVI is one of the most important, furthermore recognized as a very sensitive indicator of vegetation cover and land degradation (Dregne 1983, Tucker et al. 1983, Box et al. 1989, Price 1992, Davenport and Nicholson 1993, De Jong 1994, Hill et al. 1994, Lacaze 1994, Pickup and Chewings 1994, Gamon et al. 1995, Duncan 1997, Peñuelas and Filella 1998, Purevdorj et al. 1998, Hurcom and Harrison 1998, Lyon et al. 1998, Azzali and Menenti 2000, Hill and Peter 1996, Thornes 1996). The NDVI is widely used since it is easily obtained for different time and seasonal periods from images that can be downloaded free of charge from internet (NOAA, LANDSAT, MODIS, etc). However, its use in tropical areas has some drawbacks that can be overcome by some indices that ”correct” the traditional NDVI (Brown et al. 2000, Chen et al. 2002). Therefore for the three sites of the ECOMANAGE project, the leaf area index (LAI) was preferred.

The LAI was obtained by indirect ground measurements with two instruments, the LAI-2000 of Licor (Anonymous 1992) and the hemispheric photographs (Baret and Weiss 2004). The measured LAI was used as the independent variable and the corrected NDVI obtained from the LANDSAT images as the dependent variable in the inversion of the empirical relationship as suggested by Chen et al. (2002) to obtain the LAI maps from the corrected NDVI maps. As a result, a-dimensional character of vegetation cover (m^2 of leaves over m^2 of soil) that has a more structural meaning than NDVI and that is less affected by the high temporal variation of NDVI is obtained.

3 SHORT DESCRIPTION OF THE THREE AREAS

3.1 Santos Bay

The study area is part of Baixada Santista, located in South America in the southeastern part of Brazil within the southern limit of the tropical zone. The mean temperature is about 22 °C, and the average rainfall about 2500 mm yr^{-1}. The area belongs to São Paulo State, which is one of the most important states of Brazil with a total area of 248,176.7 km^2, a population of 40,404,010 inh (estimated for May 2006), representing 21.5% of Brazil's population with a density of 149.2 inh km^2. The study are, Santos, is approximately at 70 km distant from the city of São Paulo, the capital of the state and one of the most crowded cities of the world. São Paulo state is responsible for approximately one-third of the Brazilian GDP (Gross Domestic Product). The state's GDP (Purchasing Power Parity-PPP) consists of 550 billion dollars, making it also one of the biggest economies in Latin America.

The economy is based on machinery, automobile and aviation industries, services, financial companies, commerce, textiles, orange cultivation, sugar cane and coffee production, among the most important ones. Baixada Santista has a total area of 2373 km^2, Santos about 2000 km^2. The economy of Santos is based on its port, tourism, some important industries (i.e. copper, chemical), commerce, banks and other services. The port is the most important in Brazil and at the same time the biggest one in Latin America: it traded over 72 millions tons in 2006. It faces large industrial complexes and shipping centres that handle a large portion of the world's coffee exports; as well as a number of other Brazilian exports including oranges, bananas and cotton. The population of Santos is about 480,000 concentrated almost exclusively in the urban areas.

The estuarine system is characterized by a flat area surrounded by scarps of the plateau of Serra do Mar. The sedimentation in the estuary was always high requiring continuous dredging activities. Taking into consideration the ecological point of view the most important characteristics of the area are: the "Mata Atlantica" tropical forest that covers almost completely the scarps and the extensive mangrove forests covering almost all the flat humid areas. Currently, the Brazilian legislation considers mangroves and "Mata Atlantica" as permanent preservation areas, as a result much more control and attention is given to these natural ecosystems, protecting them from urban and industrial development.

3.2 Bahía Blanca

Bahía Blanca is located in the eastern side of Argentina, in Buenos Aires Province, due to its position, Bahía Blanca belongs to a Temperate Climatic Zone, with average yearly temperatures between 14 °C and 20 °C and thermal seasons very well differentiated. Bahía Blanca is characterized by hot summers and cold winters, while spring and autumn seasons are mildly and pleasant. Winds are, generally, moderated with an average of 25 km h^{-1}, however during spring and summer (December, January and February) wind speed can increase up to

60 km h^{-1} and more, coming from the north and north-west in summer and from the south and south-east in winter. The yearly rainfall range fluctuates between 500 mm and 600 mm, with high monthly variability. Rains are more frequent and abundant at the end of spring and beginning of summer.

The economy of the area is mainly based on the intense agricultural and cattle activities that strongly influenced the land cover pattern and the reduction of natural vegetation cover. However, also industries like oil refineries (ESSO, Shell, Petrobràs), chemical (INDUPA, Dow Quimica) and oil tanking activities and artisan fisheries are highly relevant for the area. Bahía Blanca is an important transhipping and commercial centre thanks to its location between the important regions of the ”Pampas” and ”Patagonia”. It handles the large export trade of Buenos Aires and neighbour provinces. Bahía Blanca estuary system hosts several sea ports especially on the north-eastern shore of the bay. The most important are: Puerto Ingeniero White, mainly used for grains and containers, it concentrates a large percentage of Argentina's exportation of cereal and oil, and Puerto Galván specialized in sunflower, soy oil and chemicals such as urea. The population is about 340,000 inh for an area of 6000 km^2. Urbanization level is quite high, about 98.5% of the population is concentrated in urban areas.

Completely flat areas characterize the estuarine system as well as significant intertidal flats all along the estuary, consequently producing a very high tidal excursion (almost 4 m). From the ecological point of view the coastal humid area is quite different from the one of Santos, Bahía Blanca has large extensions of salt marshes and sand vegetation.

3.3 Fiordo Aysén

Aysén, the eleventh administrative region is located in southern Chile, between 45° and 46° S and from 71° E to the Pacífic ocean. The limits of this region are: the lakes in the north, Argentina in the east, the Strait of Magellan and Chilean Antarctica to the south, and the Pacific Ocean to the west. The region has fresh oceanic and humid climate. There are an insular part and a continental part, with a total territory of 108.000 km^2. It is the least populated of the thirteen regions of Chile (91492 inh) with a density of 0.8 inh km^2. The study area includes the watershed of Fiordo Aysén (Aysén). The total area of the watershed is 11456 km^2. The annual precipitation pattern shows a strong meridional gradient between 4000 mm yr^{-1} on the west side of the basin to less than 600 mm yr^{-1} on the east side, along the border with Argentina (Delgado and Yarrow 2005).

The economy that was mainly based on forestry, breeding and agriculture, is today shifting towards an economy based on aquaculture (salmon breeding) and tourism. This will have a positive effect on forest cover that was drastically reduced during the past decades by a strong deforestation. From the ecological point of view the watershed of Aysén is very heterogeneous; including cold steppe, intermediate humid temperate, humid boreal, humid temperate, glaciers-snow and tundra.

4 RESULTS: A SCHEMATIC COMPARISON OF THE STUDY AREAS ON THE BASIS OF DPSIR FRAMEWORK

Delimitation of study areas was based on the limits of the three watersheds related to the corresponding estuarine systems. The areas of Santos, Bahía Blanca, and Aysén are: 2,000 km^2, 6,000 km^2 and 11,000 km^2 respectively. The comparison between the three areas in terms of land cover types and LAI were summarized in Table 1. The accuracy of the three land cover classifications range between 88 to 91%, the lowest for Bahía Blanca and the highest for Santos. The main reason for this difference is related to the fact that Santos has a more uniform and extensive vegetation cover of forest (on scarps) and mangroves (at sea level), whereas the land cover of Bahía Blanca is very fragmented by ”patches” of different types of crops, pastures, estuarine vegetation and the estuary itself which are much more difficult to classify.

The uncertainty of mapping Aysén's land cover is due to the heterogeneity of the landscape and the high variation in the digital elevation model (from 0 to 2600 m). The comparison of the three study areas in terms of DPSIR framework is presented in Table 2. The scale of evaluation ranges from 0 to 5, where 5 represent the maximum intensity. The areas may be considered quite different in terms of the driving forces and pressures. The fact that Santos's economy is evidently oriented to port and urban activities (commerce, services) and industries has lead to an intensive land use of flat areas for urbanisation, industrial settlements and port infrastructures and to a relatively good preservation of forest on the scarps and on highlands.

TABLE 1: *Percentage of land cover types in the three study areas and average (avg) and standard deviation (std) of their leaf area index (LAI). The maximum values for each variable of the two sets of data are indicated in bold to show the differences between the three areas.*

Land cover types	Santos	Bahía Blanca	Aysén	LAI Santos avg	std	LAI B. Blanca avg	std	LAI Aysén avg	std
Urban areas	**0.15**	0.023	0.001	0.4	0.6	0.8	**1.5**	**1.2**	**1.5**
Industrial areas	**0.03**	0.001	1E-04	0.1	0.3	0.1	0.6	?	?
Agricultural areas and grasslands	0	**0.226**	0.219	0	0	**5.5**	1.5	2.7	**2**
Pastures and prairies	0	**0.392**	0.146	0	0	**2.4**	1.0	1.2	0.87
Bare soil or very sparse vegetation	0.015	**0.236**	0.097	**1.3**	**2.6**	0.3	1.1	0.2	0.8
Mata atlantica initial secondary stage	**0.203**	0	0	**5.5**	**1.1**	0.0	0.0	0.0	0.0
Mata atlantica advanced stage	**0.411**	0	0	**6.1**	**1.1**	0.0	0.0	0.0	0.0
Degraded vegetation	**0.081**	0.025	0	**4.6**	1.8	1.7	0.9	2.4	**2.0**
Shrublands								3.84	1.94
Open forests	0	0	**0.187**	0.0	0.0	0.0	0.0	**4.4**	**3.5**
Broad leaved evergreen	0	0	**0.18**	0.0	0.0	0.0	0.0	**4.1**	**2**
Dense decidous mixed forest and coniferous	0	0	**0.17**	0.0	0.0	0.0	0.0	**5.8**	**3**
Intertidal flats	0	**0.023**	0	0.0	0.0	0.0	**0.3**	0.0	0.0
Saltmarshes (spartina, sarcococornia)	0	**0.049**	0	0.0	0.0	**0.2**	**0.7**	0.0	0.0
Sand bars, sand banks, sand pl	0	**0.025**	0	0.0	0.0	**0.1**	**0.5**	0.0	0.0
Mangrove	**0.083**	0	0	**3.3**	**0.8**	0.0	0.0	0.0	0.0
Sparse mangrove	**0.027**	0	0	**1.0**	**0.6**	0.0	0.0	0.0	0.0

TABLE 2: *Comparison between the three study areas in terms of DPSIR related only to vegetation cover as state variable. The values range between 0 and 5 (maximal intensity). The scores of Driving forces (D) reflect a judgement of intensity based of the economy of the areas, the scores on Pressures (P) reflect the data of population growth, the scores on vegetation cover (S) reflect the data in Table 1, the scores of impacts (I): loss of vegetation cover reflects the distance from the maximal natural vegetation cover, the risk of soil erosion, land slides and sedimentation reflect the combination of the land cover with the digital elevation model. As a result it is expected that low vegetation cover and steep slopes have a higher risk of soil erosion and slides, and represent a condition to produce more sedimentation in the estuarine systems.*

	DPSIR comparison			
	Santos	**Bahía**	**Aysén**	
Urbanisation	5	4	1	D
industrialisation	4	5	2	D
agriculture	0	5	3	D
breeding	0	2	4	D
forestry	0	0	3	D
mining	0	0	3	D
infrastructures	5	5	4	D
commerce	5	5	3	D
tourism	3	2	3	D
port activities	5	5	2	D
fisheries	0	2	4	D
aquaculture	0	0	5	D
population	5	3	1	P
space demand	5	4	1	P
dredging	5	?	0	P
vegetation cover	3	2	4	S
loss of vegetation cover (deforestation)	2	3	4	I
Risk of soil erosion	3	1	3	I
risk of land slides	3	0	3	I
sedimentation	3	3	2	I
conservation initiatives - % protected areas	30	?	13	R

The pressure of urbanisation is so high that in some parts of the scarps, near industrial sites, some "favelas" are developing, thus increasing the risk of slides. Furthermore, several investigations (CETESB 1980, 1986, 1989, 1990, Domingos et al. 1998, Fialho 1997, Klump et al. 1994, 1996, 2002, Moraes et al. 2000, Pinto 2000, Silva Filho 1987, Szabo et al. 2003) have shown negative effects from air pollution on the forest remnants surrounding the industrial settlements.

According to LAI data collected by ECOMANAGE there is a significant decrement of the value, which is below the average up to a distance of 1,5 Km from industrial sites (Napolitano and Feoli in prep.). On flat areas facing the sea forests were completely destroyed while mangroves suffered a strong reduction. Nowadays, mangroves are still at risk due to soil reclamation for port infrastructures, urbanisation and dredging, despite having been declared protected areas. In the watersheds of Bahía Blanca estuarine system the original vegetation, mainly prairies with sparse trees and shrubs, has practically disappeared. Crops occupy more than 20% of the watershed areas, while controlled pastures (over the original prairies) reach about 40% of the area. The space for urban development is higher than in Santos study area, so the pressure for space is lower in Bahía Blanca, which is reflected by a higher LAI in urban areas (Table 2).

The intensive agricultural and pasture activities might increase the risk of high nutrient loads in the estuary with eutrophication consequences. The fact that bare soil has a high cover value depends mainly on crops ploughing activities. Generally, agricultural activities leave soil unprotected for some period of time and exposed to wind erosion during that period. The geomorphology of Aysén's watershed is more complicated than the geomorphology of Santos and Bahía Blanca watersheds. The watershed is much larger than the other two and shows several valleys and mountain ranges with strong variability in elevation causing a high variability of vegetation cover distributed in vegetation belts. The land cover reflects an economy mainly based on forestry and cattle and sheep farming. The area suffered a strong deforestation process that has increased the risk of soil erosion, slides and sediment flow towards the sea. However, nowadays Aysén's economy is shifting towards aquaculture (salmon farming) and tourism which alleviates the pressure on forests.

5 CONCLUSIONS

The interaction between economy, geomorphology and environmental and ecological features of the three study areas has strongly influenced their land cover pattern. Considering the water condition of the three estuarine systems it is possible to say that this pattern has been influenced mainly by urban and industrial pollutants in Santos, by industrial and agricultural pollutants in Bahía Blanca and by sediments and aquaculture pollutants in Aysén. Planning activities, as RESPONSES in the DPSIR framework are required. These RESPONSES should be based on policies that will reduce the risk of losing ecosystem functions of the estuarine systems, hence maintaining a high level of the ecosystem services in the watersheds of the study areas. Protection measures of the natural ecosystems are applied in Santos and Aysén, however a development policy which integrates the ecosystem functionality of the watersheds with the ecosystem functionality of the estuarine systems is needed for a better integration between the economy related to land use and the economy related to the use of coastal waters and ecosystems.

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GROUNDWATER ASSESSMENT OF SANTOS ESTUARY

M.A. MANCUSO, J.P. LOBO-FERREIRA AND J.L. ALBUQUERQUE FILHO

1 INTRODUCTION

Groundwater studies were developed for a better understanding the fresh water dynamic on the basin. A groundwater flow model was developed to simulate the contribution of fresh water from the sedimentary aquifer to Santos Estuary, based on Anderson and Woessner (1992) protocol. The conceptualization was based on the physical system, its geometry, geology and hydrogeology. The conceptual model also comprises groundwater recharge estimation and water budget understanding. For groundwater studies the subject area was the land that contributes in surface and underground with fresh water to Santos Estuary. Only the western part of the Santo Amaro Island and Cabuçu basin contributes with fresh water to the Santos estuary. The water flows directly to the Bertioga channel, which separates the Santo Amaro Island from mainland. So the study area is about 835 km^2 large.

2 GEOLOGY

The geological description is based on CETEC (1999) and the geological map based on the Suguio and Martin (1978, in DAEE, 1979, scale 1:100.000), to delimitate the Cenozoic sediments, and Perrota et al. (2004) (scale 1:750.000), IPT (1981) (1:500.000) and DAEE (1979) for the geological characterization of this formations (Figure 1). The geology of the area may be grouped into the basement formations (Paleozoic and Pre-Cambrian) and cenozoic cover (alluvium, marine and mixed sediments, undifferentiated continental sediments and Cananéia formation) (Figure 1). The basement formations are the result of several tectonic phases, responsible for deformation, faulting, foliation, besides metamorphism and magmatic processes, combined with variations of sea level and climate. Inland, several faulting and epeirogenesis have produced the escarpment of the actual Serra do Mar.

In the Cenozoic, the main events may be summarized in topography modelling, tropical humid climate, sea transgression and deposition of the sedimentary sequences. The geological description of the Cenozoic formations is summarized in Table 1. The sedimentary cenozoic coverage is formed of unconsolidated sediments, located in the plain and low areas of the Coastal Plain and in the foot of the hills. They are represented by four geological units: Qa - alluvium sediments; Qm - marine and mixed sediments; Qi - undifferentiated continental sediments; and Qc - Cananéia formation.

The Cananéia formation (Qc) is composed of old marine sandy deposits (thin sands) with sparse clayey layers, often limonited, with average thickness of 30 m. Externally to the Cananéia formation, extensive portions of Marine and Mixed Sediments (Qm), include sands from beaches, marine deposits locally subject to fluvial and/or eolic action, sandy-silted-clayey terms from fluvio-marine-lacustrine deposition and mangroves deposits. The thickness of these sediments (sandy to clayey, mud with high content of biodetritical organics from mangrove) may be more than 50 m deep. The detrital deposits mainly located in the basal portion of the hills and half hillside (Undifferentiated continental sediments - Qi), are mainly composed of immature sediments, poorly sorted and often coarse material from colluvium's material. They are formed of gravels, sands and clays in variable proportions sometimes comprising numerous blocks.

The alluvium sediments (Qa) comprise unconsolidated sands of variable granulometry, as well as clay and fluvial gravels, also found in terraces. In the Paleozoic and Pre-Cambrian rocks the soils cover the basement, having been produced from the weathering of bedrock, and whose granulometry, mineralogy and thickness vary accordingly with the basement rock lithology.

The characterization of the bottom of the sedimentary aquifer and the sediments stratification is based on the following data:

- geological maps: Suguio and Martin (1978), Perrota et al. (2004) and IPT (1981);
- 4 geological cross sections (DAEE, 1979);
- 42 well logs (DAEE, 1979);
- 10 geophysical logs (DAEE, 1979).

According to the collected information, the bottom of the aquifer (top of crystalline rock) varies from -25 m close to the hard rock, disappearing where the hard rock formations outcrop, and increasing up to -230 m in the southwest part of the basin, close to the ocean and up to -125 southeast of São Vicente Island. In the Santos Estuary channel the bottom of the aquifer varies mostly from -50 to -75 m, with the higher levels north of the island are up to -25 m (Figure 2). The sediment thickness in Praia Grande, close to the ocean, is mostly around 100 m, increasing to 230 m southwest. North of Santos the thickness decreases to 30 m but southeast it is around 150 m (Figure 2).

3 HYDROGEOLOGY

The geological approach was used to study the aquifer geometry and hydraulic characterization. In Santos basin there are two main groundwater systems, the fracture and the sedimentary aquifer. In this case study, considering the purpose of the hydrogeological studies on the identification of the fresh water discharge to the estuary, we focus on the sedimentary aquifer system. The hydraulic characterisation of this aquifer media was based on the 3D geological map (Figure 3), elaborated on the basis of Suguio and Martin (1978), Perrota et al. (2004), IPT (1981), well logs, geophysical logs, geological cross sections and hydrogeological data from DAEE (1979). The hydraulic conductivity (K) parameter for each geological formation of the sedimentary aquifer was a function of the lithology and, also, based on recovery test of monitoring wells performed by Consultoria Paulista (2004) and MKR (2003).

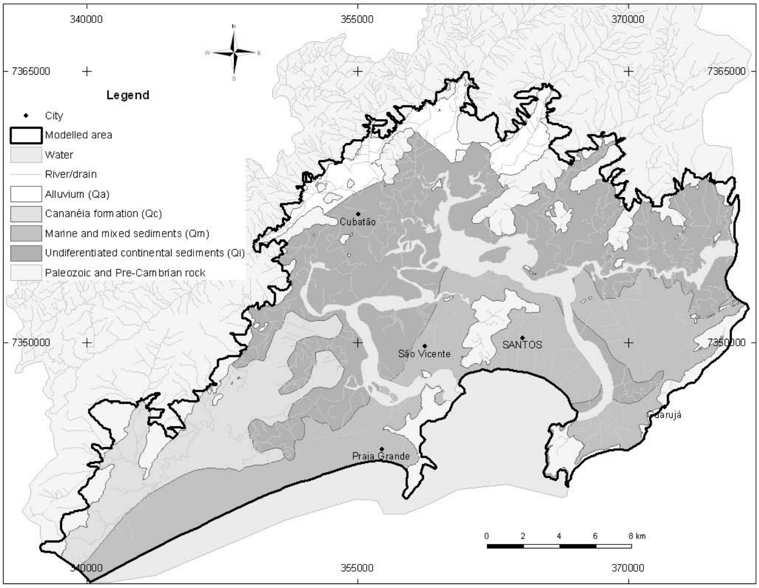

FIGURE 1: *Geology of the sedimentary formation of the Land Santos Estuary area (Source: Perrota et al. 2004, Suguio and Martin 1978).*

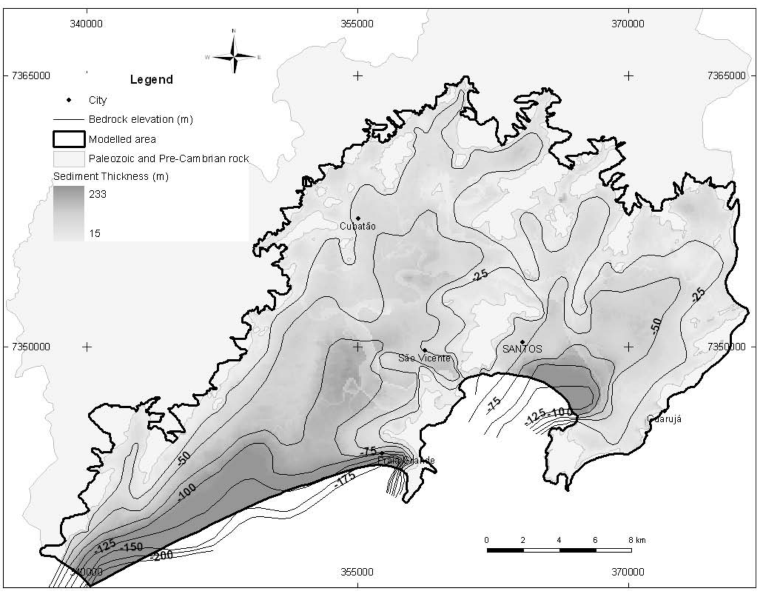

FIGURE 2: *Bedrock elevation and thickness of the sediments in the porous aquifer (Source: Bedrock elevation modified from DAEE 1979).*

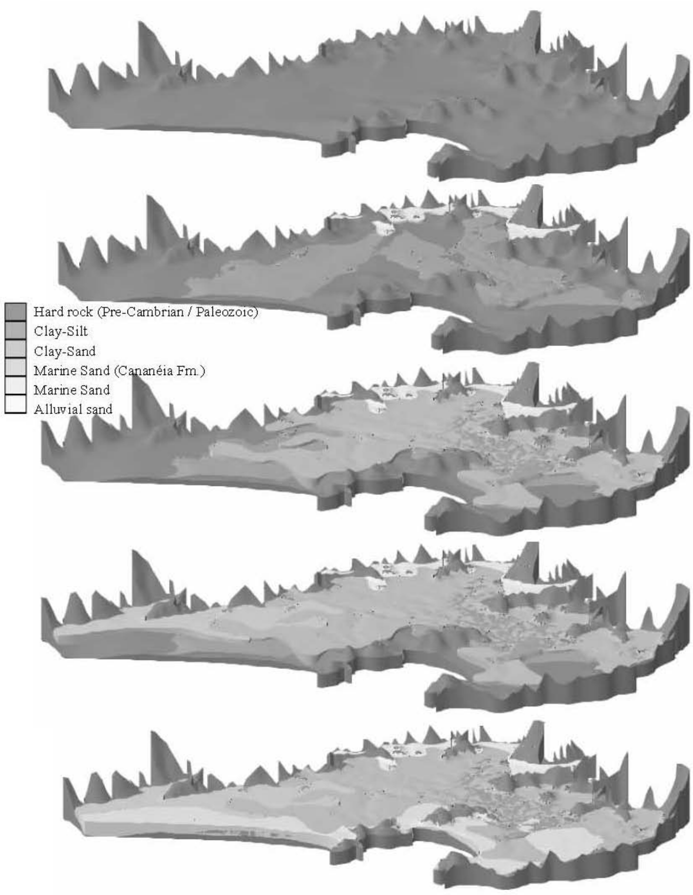

FIGURE 3: *Steps of a 3D view of the sedimentary formations in the porous media (Source: after DAEE 1979, Suguio and Martin 1978, Perrota et al. 2004, IPT 1981).*

For the formations the following hydraulic conductivity values have been assigned:

- Alluvium (Qa) (alluvial sand): k_x and k_y = 1 m d^{-1} (1.16 x 10^{-3} cm s^{-1}), k_z = 0.01 m d^{-1} (1.16 x 10^{-5} cm s^{-1});
- Cananéia formation (Qc) and Marine and mixed sediments (Qm) (marine sand and sandy-siltic-clayey terms and mangroves deposits): k_x and k_y = 0.5 m d^{-1} (5.79 x 10^{-4} cm s^{-1}), k_z = 0.005 m d^{-1} (5.79 x 10^{-6} cm s^{-1});
- Undifferentiated continental sediments (Qi) (sand-clay and silt also mangroves deposits): k_x and k_x = 0.5 m d^{-1} (5.79 x 10^{-4} cm s^{-1}), k_z = 0.005 m d^{-1} (5.79 x 10^{-6} cm s^{-1});
- Weathering rocks from Pre-Cambrian / Paleozoic formations was assigned: k_x and k_y = 0.009 m d^{-1} (1.16 x 10^{-5} cm s^{-1}), k_z = 0.0001 m d^{-1} (1.16 x 10^{-7} cm s^{-1}).

Groundwater levels have been monitored for the mapping of the regional piezometry. Bibliography from DAEE (1979), Consultoria Paulista (2004) and MKR (2003) helped on this approach. Groundwater levels are close to the surface and vary from 0 m to 5 m elevation in Praia Grande and up to 8 m in the western part of São Vicente Island.

3.1 Evaluation of groundwater discharge to the Estuary

Groundwater studies were developed for a better understanding of the fresh water dynamics in the basin. A groundwater mathematical model was elaborated to simulate the hydrodynamics of the sedimentary aquifer system. The area for modelling was defined in ArcView, using a geological map to delimit the Cenozoic sediments (Suguio and Martin 1978 in DAEE 1979) and the geological characterization (Perrota et al. 2004, IPT 1981). A groundwater flow model was developed to simulate a sedimentary aquifer, based on the Anderson and Woessner (1992) protocol. The conceptualization was based on the physical system, its geometry, geology and hydrogeology. The conceptual model also comprises groundwater recharge estimation and water budget understanding.

As a boundary condition for the sedimentary aquifer, on the crystalline interface, it was assumed that the soil covering the basement rocks, produced by the weathering, is capable of retaining water from precipitation. That water, as recharge, flows to rivers and on the interface between the aquifers, to the sedimentary formation. As a result the modelled area was the sedimentary aquifer plus small basins from the crystalline aquifer (Figure 4). To delimit the crystalline basins without surface drainage a topographic map was used, with elevation curves from 20 to 1160 m (IGGSP 1971, 1972), besides a drainage map (stream coverage) for this formation. The data was used to produce a flow direction map and from the results a new grid was created outlining all drainage basins within the analysis window.

3.1.1 Recharge

The recharge was calculated based on the mass balance between water coming in, going out and being stored in the water system. This balance was made for the Cubatão watershed by DAEE (1979). The results were used as the basis for estimating recharges on the others watersheds of the modelled area. For modelling purpose the total recharge was estimated as 28.2% of precipitation (DAEE 1979), considering urban areas and mangrove. The mean recharge value was then 0.0019 $\mathrm{m\,d^{-1}}$ as an initial modelling condition. Recharge values for each watershed (Table 2) were considered as an input for groundwater model calibration in the study area, besides the influence of urban areas and mangroves on the recharge rate.

3.1.2 Groundwater mathematical model

The total modelled area is 448 $\mathrm{km^2}$ (406 $\mathrm{km^2}$ of land area approximately) and comprises part of the following watersheds: Piaçabuçu, Boturoca, Cubatão, Mogi, Quilombo, Jurubatuba, Cabuçu, Santo Amaro Island and São Vicente Island. The flow model was developed considering flow as a steady state, with lakes and the channel considered as constant head cells. Rivers were simulated as drains. A two layer model of the porous aquifer was elaborated, with the first being an unconfined layer and the second a confined/unconfined layer. This division intended to evaluate the real groundwater flow rate discharge to the estuary, considering a total penetrating channel on the first layer. The first layer has a mean thickness of 25 m, and the second layer has variable thickness according to the bedrock elevation.

The 3D model grid starts at the World Coordinates X = 337,080 m, Y = 7,336,560 m and Z = -230 m, ending at X = 377,080 m, Y = 7,365,560 m and Z = 1,070 m. The model has two layers subdivided into 116 rows by 160 columns. Each cell is 250 m by 250 m along the xx and yy directions. The zz direction varies according to the layer to which each cell belongs and to the x and y coordinates of that cell. The thickness for the first layer was calculated as 30% of total thickness of the sediments, between topographic level and hard rock, specifying a minimum thickness of 20 m. Total thickness was calculated from topographic grid and hard rock top elevation grid (-230 m), base for the sedimentary aquifer system (Figure 5). Surface water from rivers, drains, Santos estuary and the Atlantic Ocean were represented as drains and constant head on Modflow software. On the first layer (Figure 6) were represented the rivers, drains and channels, as a partially penetrating water system. The Atlantic Ocean has been represented as a constant head and a full interception system for groundwater flow. Considering the purpose of this work, groundwater and sea water interaction was not simulated . The constant head boundary at 0 m allowed quantifying mean groundwater discharge from the sedimentary aquifer.

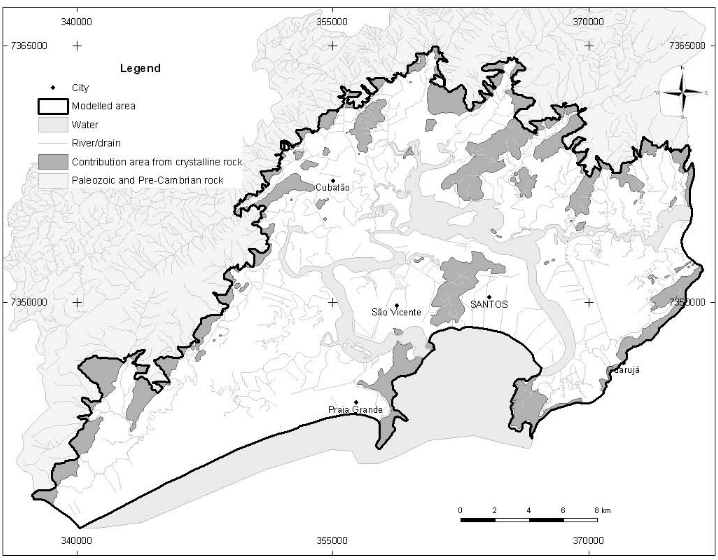

FIGURE 4: *Sedimentary aquifer and basins from crystalline aquifer that contribute with groundwater flow to the porous aquifer (Source: Cenozoic formation limit after Suguio and Martin 1978).*

3.2 Model calibration results and conclusions

Model calibration target was the piezometric surface to be compatible with DAEE (1979). Table 3 shows range values used for calibration. The objective of the model calibration is to obtain piezometric surface compatible with observed heads, or the piezometric surface of DAEE (1979). In a general way, at Santos Estuary sedimentary aquifer groundwater flows locally towards the rivers, and the drains opened according to the urban development. Regionally groundwater flows towards the sea as may be observed from piezometric maps shown in Figure 7.

Water budget shows that flows in and out from the system are almost the same, with a discrepancy of 1%. Results show that total discharge from groundwater is 877 x 10^3 $m^3\,d^{-1}$, 208 x 10^3 $m^3\,d^{-1}$ (24%) flows to the Estuary and 493 x 10^3 $m^3\,d^{-1}$ (56%) to the drains, part of groundwater flows to the Atlantic Ocean (20%) (Table 4). Total discharge from precipitation through the sedimentary aquifer to the Estuary is about 8%. Flow depends on the watershed location, area and land use, but mean flow from groundwater to the Estuary boundary is around 1 $m^2\,m^{-1}\,d^{-1}$. Table 4 and Figure 8 show the groundwater discharge from the sedimentary aquifer to the Estuary System.

According to the groundwater model developed in this work, the discharge from the sedimentary aquifer directly to the Estuary is approximately 8% of the precipitation. Discharge from fractures of the crystalline aquifer (hard rock) has not been considered. From the results it can be concluded that groundwater discharge to Santos Estuary depends on the watershed location, area and land use. Flow discharge is lower on a well drainage basin, because groundwater contributions are mainly to the drains. This base flow discharge is counted on the rivers flow rate. There is a high fresh water discharge to the Atlantic Ocean, from Praia Grande and Santos area. Lower values are related with crystalline rock outcrop and small islands. Monthly recharge values for each watershed were considered as an input for model simulation in the study area, as well as the influence of urban areas and mangroves on the recharge rate.

One can observe differences between groundwater recharge rates on forest land use for the Cubatão watershed, and groundwater recharge on urban areas, as calculated for Santos and São Vicente. The computation didn't consider losses from water supply pipes. From the simulation it can be conclude that Boturoca watershed, São Vicente and Santo Amaro Island are the main contributors of fresh water from the sedimentary aquifer to the estuary. Lower discharges to the estuary were observed in drains and, afterwards as surface water, to the estuary. It's important to collect new groundwater data to improve this first modelling approach. The physiography of the area, the land use and mainly the large amount of rainfall, increasing the availability of water, at least in quantity terms, may be responsible for the non-existence of groundwater studies.

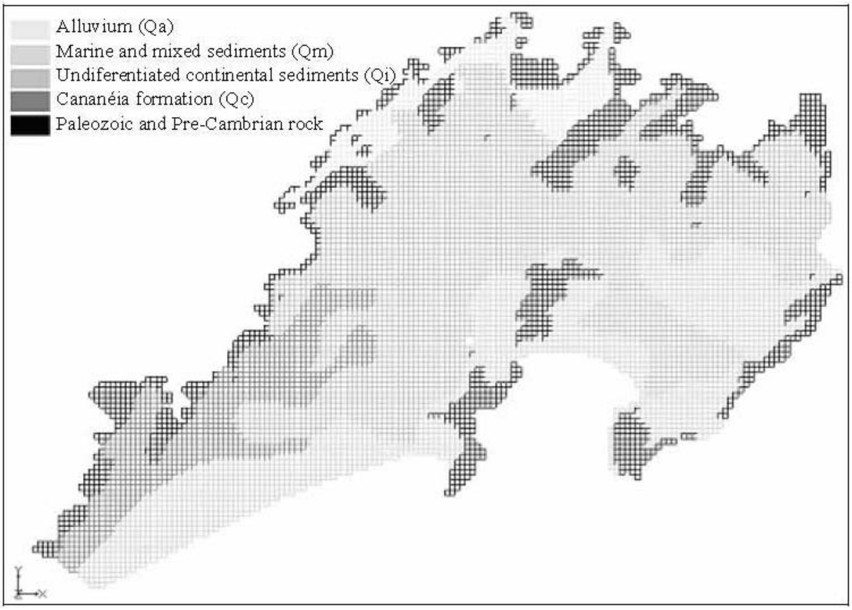

FIGURE 5: *3D model grid for the sedimentary aquifer at Santos Estuary System.*

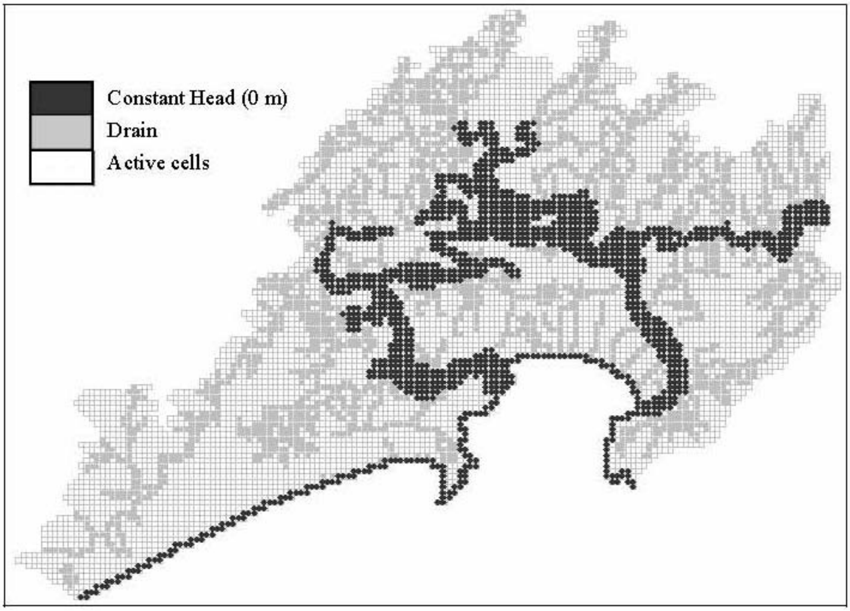

FIGURE 6: *Source/Sink for layer 1 shows rivers as drains, and constant head cells at ocean boundary and at Santos Estuary channel.*

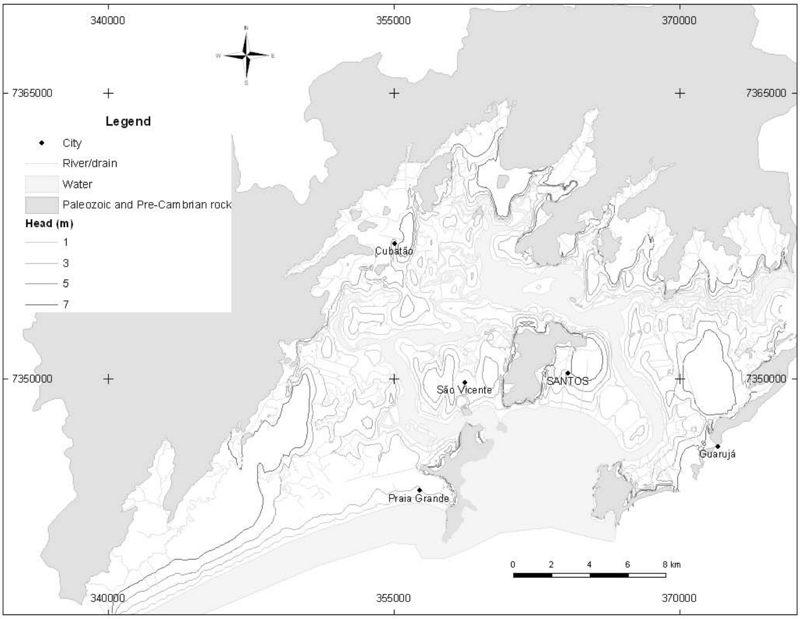

FIGURE 7: *Piezometric head for Land Santos Estuary sedimentary aquifer resulted from Modflow model.*

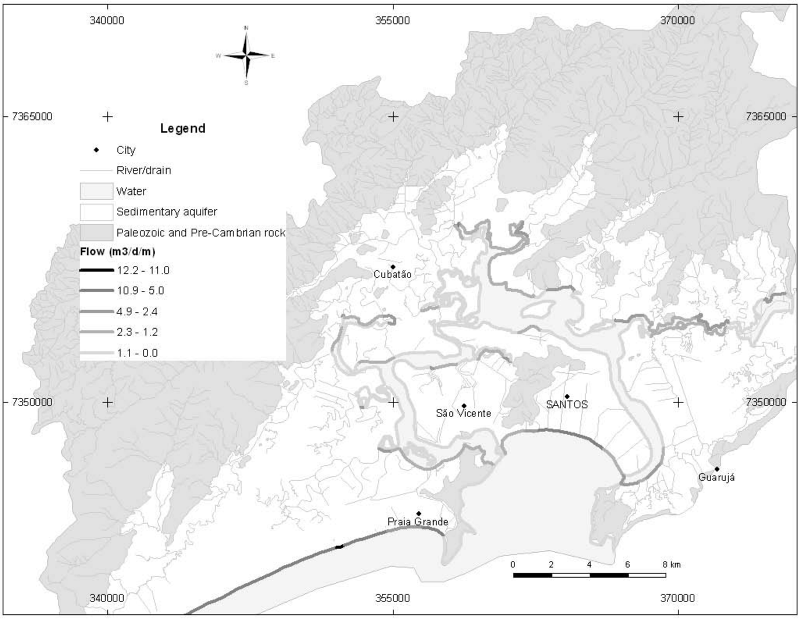

FIGURE 8: *Groundwater flow discharge from the sedimentary aquifer to Santos Estuary.*

TABLE 1: *Geology of the sediment formations of Land Santos Estuary area (Source: Geological Map of São Paulo State, in IPT 1981).*

Period (Age)	Geological formation	Lithology
Cenozoic	Qa - Alluvium	Unconsolidated sands of variable texture, associated clays and gravels
	Qm – Marine and mixed sediments	Sands, marine, sandy-siltic-clayey terms and mangroves deposits
	Qi – Undifferentiated continental sediments	Continental deposits
	Qc – Cananéia formation	Unconsolidated thin marine sands

TABLE 2: *Recharge in Santos Estuary watersheds.*

Watershed	Modelled Area (km^2)	P (mm $year^{-1}$)	Potential Recharge[2] ($m^3\ d^{-1}$)
Boturoca stream	128.1	2640	261,282
Cubatão stream	21.0	2863	46,451
Piaçabuçu stream	57.1	2405	106,098
S. Vicente island	58.5	2291	103,547
Mogi stream	20.5	2704	42,827
Santo Amaro island	68.6	2376	125,929
Cabuçu stream	28.4	2771	60,801
Jurubatuba stream	36.3	2539	71,207
Quilombo stream	25.0	2619	50,586
Islands	4.9	2410	
Total	448.3		868,728

[2] 28,2% from precipitation, considering urban areas and mangrove (DAEE 1979)

TABLE 3: *Range values used to calibrate the model.*

Parameter	Initial Condition
Hydraulic Conductivity (k)	k_{xx} and k_{yy} = 0.009 m d^{-1} to 1 m d^{-1} ; k_{zz} = 0.005 m d^{-1} to 0.0001 m d^{-1}
Drain Conductance	0,2 to 2 L (m)
Recharge	1,8 x 10^{-3} m d^{-1} to 2,2 x 10^{-3} m d^{-1}
Initial head	0 m to 10 m

TABLE 4: *Estimated groundwater discharge to the Estuary System resulting from a year groundwater simulation.*

Watershed	Discharge Boundary (m)	Discharge Flow ($m^3 d^{-1}$)			
		Estuary	Drains	Atlantic Ocean	Total
Boturoca stream	43,971	-43,816	-147,098	-58,805	-249,719
Cubatão stream	5,804	-4,600	-45,890	-	-50,490
Piaçabuçu stream	10,697	-13,910	-38,002	-68,044	-119,956
S. Vicente island	40,878	-40,092	-23,798	-45,648	-109,538
Mogi stream	10,449	-8,580	-30,283	-	-38,863
Santo Amaro island (*)	25,202	-25,825	-74,350	-2,826	-103,001
Cabuçu stream (*)	13,866	-24,164	-43,302	-	-67,466
Jurubatuba stream	14,183	-21,717	-56,260	-	-77,977
Quilombo stream	9,224	-19,941	-33,604	-	-53,544
Islands	29,544	-5,036		-	-5,036
Santos Estuary - Ocean				-1,531	-1,531
Total		-207,681	-492,587	-176,855	-877,122

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CONTAMINANT TRANSPORT IN THE SEDIMENTARY AQUIFER OF ALEMOA

C.B.B. LEITE, C.Z. TOPOROVSKI AND M.A. MANCUSO

ECOMANAGE project aims to push the capacity of assisting managers to join horizontally knowledge from ecological and socio-economic disciplines. The three key aspects of ECOMANAGE are (1) the consideration that a coastal zone depends on local pressures, but also on pressures originated in the drainage basin, transported mostly by rivers and by groundwater, (2) that socio-economic activities are the driving forces of those pressures and that their impacts on the ecosystem have feedback on socio-economics and (3) the impacts depend on physical characteristics of the ecosystem that together with the loads determine its ecological state. The project is destined to develop a management integrated system for the coastal zone and apply it in the Santos Estuary - SP - Brazil. The research was developed using mathematical models to quantify the pollution loads transported by groundwater from Alemoa's dump to the Santos Estuary.

Alemoa quarter is one of the oldest used areas of Santos city. It's located at the northwest part of the island. This area is inside the limits of Santos Port, where today is located an important industrial complex, with patios and deposits of containers used to transport loads to and from the port. Alemoa area was used for more than 50 years as a disposal for solid waste from the operation of the Santos Port. Also, domestic waste was illegally dumped in the area. Beyond these activities, the area was used to store and move explosives on the access bridge and material dragged during the installation activities of Petrobras´ terminal.

This work presents the first approach for assessing the evaluation of contaminant discharge from the illegal waste dumped at Alemoa area into Santos Estuary, based on existing data. It presents the conceptual model, the mathematical model and the computed results that allowed the quantification and mapping of the contaminant flow discharge. The physical data of the Alemoa area was presented in terms of geology, hydrogeology, altimetry, evaporation and precipitation. The total modeled area is 0,917 km^2 and comprises part of the São Vicente Island (Figure 1).

The model was developed using the software: Ground Water Modeling System - GMS®, with the modules MAP, MODFLOW, PEST, MODPATH, and MT3D. MODFLOW was first published by McDonald and Harbaugh (1988). The flow model was developed considering flow as a steady state, with lakes and the channel considered as constant head cells. Rivers were simulated as drains. The conceptualization was based on the physical system, its geometry, geology and hydrogeology. The conceptual model also comprises groundwater recharge estimation and water budget understanding.

The area for modeling was defined in AutoCAD®, using lithologics and hydrogeologics information of 35 boreholes. As a boundary condition for the sedimentary aquifer, it was considered the crystalline interface at -25 m deep. To delimitate this area was used a topographic map (AGEM, 1:1 000). These data was used to produce a grid with 10 m x 10 m cell representing the elevation surface. Considering the lithology two layers were defined, both layers on porous aquifer. The bottom elevation of the first layer is -3.5 m and the bottom elevation of the second is -25 m, so the first layer has a thickness of 4 to 10 m.

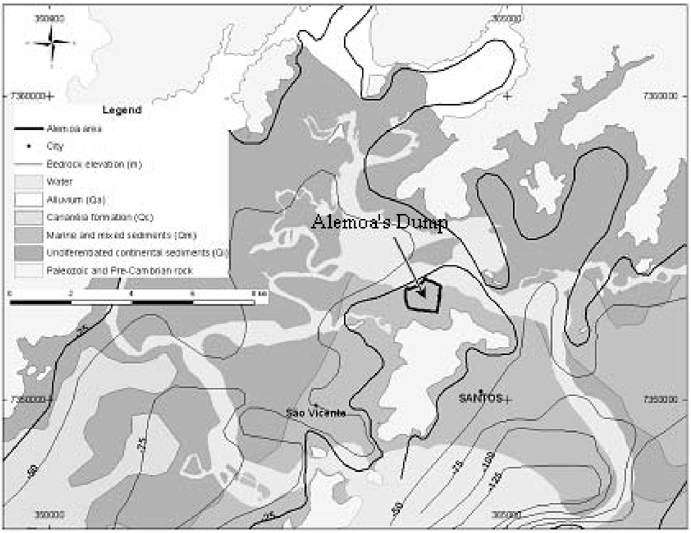


FIGURE 1: *Alemoa's dump in Santos Estuary.*

The mathematical model used to represent the contamination plume was MT3D. The model was published by Zheng and Wang (1999) and it is used to calculate the transport of a contamination plume by resolving transport equations. Four transport models were built to determinate the contaminations plume of benzene, cadmium, lead and toluene. The income data used by MT3D model had followed the same parameters used for Modflow model. For the determination of the contamination plume were considered the concentrations of contaminants obtained in each of the 35 boreholes (observation wells) as a constant load launched into the aquifer.

Considering these data, the contaminant transport model allowed the simulation of advection and dispersion in the aquifer sedimentary system. The final aim was to quantify the diffuse pollution loads discharging to the Santos Estuary. Figures 2 to 5 present the contamination plumes in Alemoa area in a period of 100 years, and Table 1 presents the contaminant discharge to Santos Estuary.

From the results, presented in Figures 2 to 5 and Table 1, it can be concluded that the contaminant discharge of the Alemoa's dump are mainly directed to drains, and then to Santos Estuary. Also, underground discharge direct to the Santos Estuary is lower. The total contaminant discharge of Alemoa's dump to the Santos Estuary, considering a period of 100 years, was calculated in 175,048.91 mg to benzene, 53,310.1 mg to toluene, 824,704.21 mg to lead and 64,602.64 mg to cadmium.

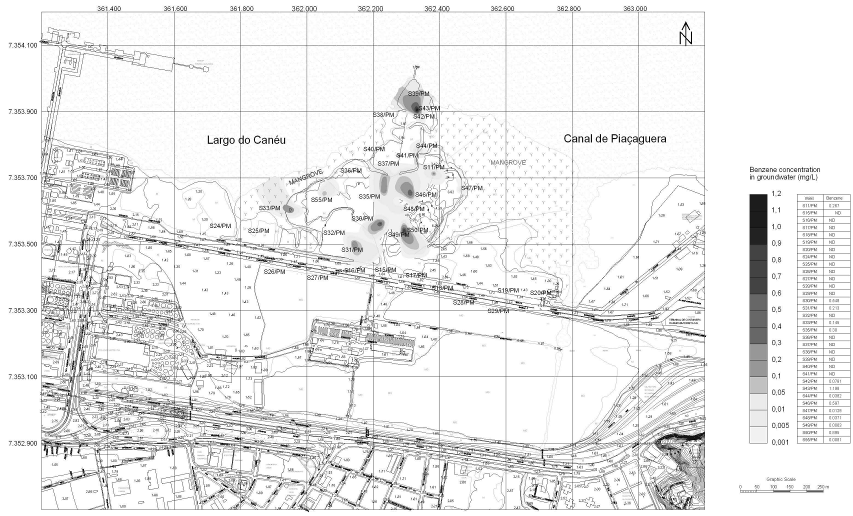


FIGURE 2: *Contamination plume for Benzene (C_6H_6).*

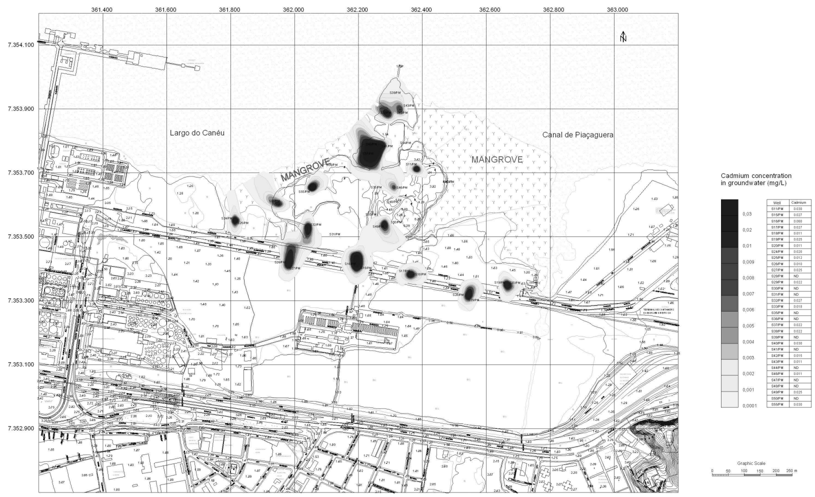
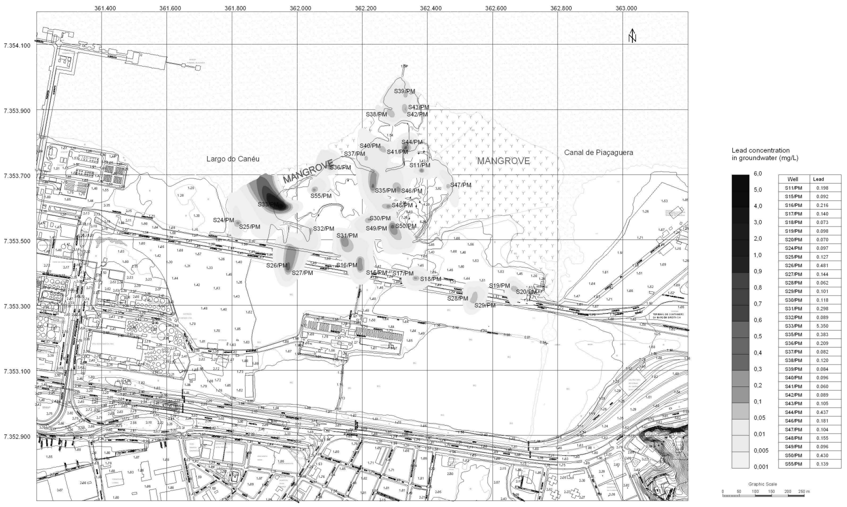


FIGURE 3: *Contamination plume for Cadmium (Cd).*

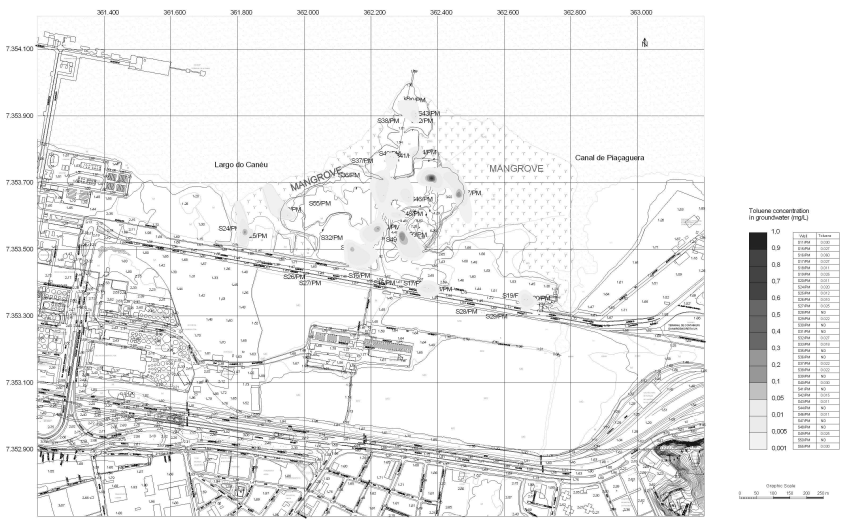
TABLE 1: *Contaminant discharge to Santos Estuary.*

Compounds	Drains (mg)	Drains (mg y^{-1})	Santos Estuary (mg)	Santos Estuary (mg y^{-1})	Total Discharge (mg)	Total Discharge (mg y^{-1})
Benzene (C_6H_6)	130,443.00	1304.43	44,605.91	446.06	175,048.91	1750.49
Cadmium (Cd)	59,247.54	592.48	5,355.097	53.55	64,602.637	646.03
Lead (Pb)	494,854.80	4948.55	329,849.90	3298.50	824,704.70	8247.05
Toluene (C_7H_8)	51,095.49	510.95	2,214.61	22.15	53,310.10	553.10



Well	Lead
S11/PM	0.198
S15/PM	0.092
S16/PM	0.216
S17/PM	0.140
S18/PM	0.073
S19/PM	0.098
S20/PM	0.070
S24/PM	0.097
S25/PM	0.127
S26/PM	0.481
S27/PM	0.144
S28/PM	0.062
S29/PM	0.101
S30/PM	0.118
S31/PM	0.298
S32/PM	0.089
S33/PM	5.350
S35/PM	0.383
S36/PM	0.209
S37/PM	0.082
S38/PM	0.120
S39/PM	0.084
S40/PM	0.096
S41/PM	0.060
S42/PM	0.089
S43/PM	0.105
S44/PM	0.437
S46/PM	0.181
S47/PM	0.104
S48/PM	0.155
S49/PM	0.096
S50/PM	0.430
S55/PM	0.139

FIGURE 4: *Contamination plume for Lead (Pb).*



Well	Toluene
S11/PM	0.030
S15/PM	0.027
S16/PM	0.060
S17/PM	0.027
S18/PM	0.011
S19/PM	0.025
S20/PM	0.011
S24/PM	0.020
S25/PM	0.012
S26/PM	0.010
S27/PM	0.025
S28/PM	ND
S29/PM	0.022
S30/PM	ND
S31/PM	ND
S32/PM	0.027
S33/PM	0.018
S35/PM	ND
S36/PM	ND
S37/PM	0.022
S38/PM	0.022
S39/PM	ND
S40/PM	0.030
S41/PM	ND
S42/PM	0.015
S43/PM	0.011
S44/PM	ND
S46/PM	0.011
S47/PM	ND
S48/PM	ND
S49/PM	0.025
S50/PM	ND
S55/PM	0.030

FIGURE 5: *Contamination plume for Toluene (C_7H_8).*

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LOAD AND FLOW ESTIMATION IN SANTOS WATERSHEDS

P. CHAMBEL-LEITÃO, A.F.P. SAMPAIO AND P. ALMEIDA

1 INTRODUCTION

The basin of the Santos estuary, due to its high rate of urbanization has a very diversified use, designed especially for public and industrial services, reception of domestic and industrial effluents, fishing for sustenance, recreation, navigation and generation of electricity (CETESB, 2005). The estuary has an important basin designed for the supply of one of the great industrial centers of the country, the largest port of Latin America and a population of over a million people distributed over five towns that make up the estuary of Santos. Historically some rivers received in the past a huge load of industrial effluents without any treatment from petrochemical, chemical and fertilizer plants as well as from a steelworks. In the last decade, the quality of the effluents improved significantly thanks to regulatory actions taken by environmental institutions which forced industries to invest in treatment systems, but some rivers still suffer high rates of BOD. Besides, the basin suffers large pressures from irregular human occupation on the river banks and from urban discharges without treatment.

This paper presents the application of SWAT to estimate de effluent flows to the estuary and the application of HARP-NUT guidelines to estimate the loads of nutrients to the estuary. Both the flows and nutrients estimated were used as boundary conditions to the MOHID estuary model.

2 THE SWAT MODEL

During the ECOMANAGE project a Landsat satellite image of April 30, 2000 was used for the elaboration of vegetation cover maps for the Santos estuary watersheds. The results obtained were then used as the Landcover input for the SWAT model. The soil map included in CETEC (1999) was used as input for SWAT. Information referring to the analytic, grain and chemistry characteristics associated with these types of soils were collected from Rossi (1999). The topography used was obtained by NASA in the Shuttle Radar Topographic Mission (SRTM) and that was available for download from the Internet.

The precipitation used was from Paranapiacaba (code: E3-037) Latitude -46.28 and Longitude -23.77 which lies at an elevation of 820 m. This station was chosen for four reasons: i) it has an almost complete time series; ii) it has a long period of data which includes the flow measures data (from 01-01-1936 to 30-06-1998); iii) it has an average annual precipitation of 2000 mm which, according to the average annual precipitation maps, is the average precipitation in the region; iv) it is the most representative for Quilombo river which is the watershed with most measurements of flow. Fluviometric, meteorological, and water quality data was made available through BcDaee2000. There are 3 fluviometric stations in the watersheds around

Santos estuary (Figure 1). Unfortunately, only data for a short historical period are available for Cubatão and Moji stations (three and two years respectively), while for Quilombo we have 15 years of data. The main conclusion was that Cubatão measured and modelled flow had the best fit (Table 1). On the other hand Quilombo modelled flows were higher than the measured ones. This might be due to an under estimation of evapotranspiration or an excess input of precipitation to the model.

Evapotranspiration estimated by the model is around 20-30% which is similar to values estimated by Lopes (2001). To obtain the measured values, 50% evapotranspiration would have to be calculated with the model. But this wouldn't be consistent with flow measures for Cubatão and Moji which indicate evapotranspiration of 20-30%. On the other hand precipitation is expected to be representative for Quilombo river because it's the closest to Quilombo and it has an average annual precipitation of 2000 mm which, according to the average annual precipitation maps, is the average precipitation in the region. This means that probably the flow measurements (which in fact are based on river level measurements) in Quilombo were underestimates. Finally, Moji has few values and half of them are not concordant with the model while the other half shows a very good fit.

3 HARP-NUT GUIDELINES

The application of the OSPAR HARP Guidelines to the Santos Basin has helped to elucidate some of the gaps in the data available for this basin. Nevertheless, we have proceeded to apply the guidelines, indicating in the following text the assumptions made for each guideline. Based on CETESB monitoring stations, four monitored areas were evaluated using OSPAR HARP Guidelines. The name of the water quality station is shown in Table 2 as well as the area of the associated drainage basin. The distribution of these drainage basins is displayed in Figure 1. The remaining area that drains to the estuary and is not included in these monitored watersheds is called the unmonitored area, and its delineation is shown in Figure 2.

No data was found on aquaculture in the area of Santos Estuary. Due to the high degree of industrialization of the region and to the level of pollution in the area it is expected that there is no aquaculture in the area. Loads estimated for Henry Borden using CETESB data were 446 ton N yr^{-1} and 43 ton P yr^{-1}. For the drainage area of the monitoring station Cuba27 (in the upper part of Cubatão watershed), no industries were identified, and the diffuse loads coming from this area were used as an input to the Cuba39 monitored area.

The following industrial discharges are included in the drainage area of monitoring station Cuba39: Ultrafertil-CCB, Ripasa, Petrobras-RPBC, CBE, Dow Química, Petrocoque and Carbocloro. From those, there were available data for outflow and BOD only for Ultrafertil-CCB, Ripasa, Petrobras-RPBC, Petrocoque and Carbocloro; additionally, there were data on Nitrogen and Phosphorus and outflow for Ultrafértil - CCB. The Henry Borden Power Plant also discharges in the Cuba39 monitoring station, and was also considered an industry.

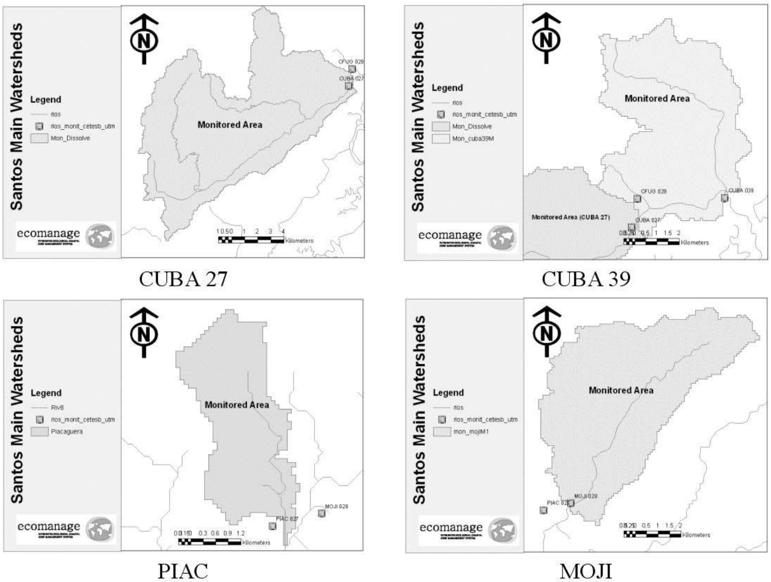

FIGURE 1: *Drainage area of CETESB monitoring stations.*

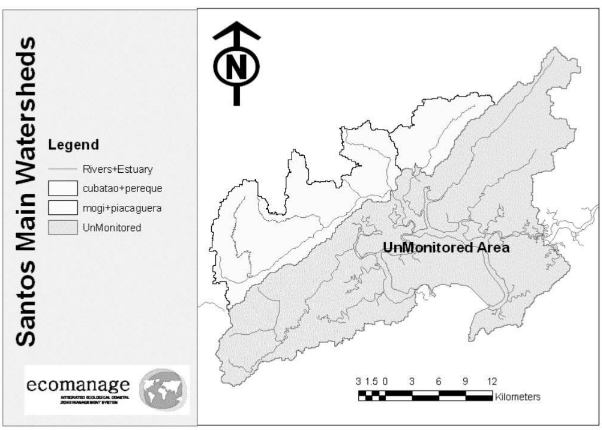

FIGURE 2: *Unmonitored area draining to the estuary.*

TABLE 1: *Comparison of observed and modelled average monthly flows.*

	Calibration period	Observed flow [$m^3 s^{-1}$]	Modeled flow [$m^3 s^{-1}$]	R2	Model Efficiency
Cubatão	Oct '66–Jul '69	10.4	11.6	0.68	0.67
Moji	Aug '72 –Sep'74	2.35	2.61	0.34	0.19
Quilombo	Oct '71–Nov '87	3.02	4.50	0.63	0.01
Dailly					
Cubatão	Oct '66–Jul '69	10.4	11.6	0.56	0.55
Moji	Aug '72 –Sep'74	2.35	2.61	0.33	-0.97
Quilombo	Oct '71–Nov '87	3.02	4.50	0.33	-0.67

TABLE 2: *Monitored area delineated based in monitoring stations.*

Watershed	Code	Area [km^2]
Cubatão	CUBA 27	135
Cubatão	CUBA 39	45
Moji	MOJI	46
Piaçaguera	PIAC	9

For Moji four main industries were identified in the drainage basin: Ultrafertil-CPG, Manah, Serrana which are part of Bunge Fertilizantes S/A. Only for the first industry was found data on BOD, nitrogen and outflow. Even with those values the estimated loads originating from industry are almost irrelevant. However in a load oriented approach it is shown that loads in the rivers are very high and that they are probably originating from these industries. According to Santos (2001), it is estimated that in Cubatão, in the state of São Paulo, there are about 20 million tons of phosphorus cast, dumped into two open-air piles referring to the production of Ultrafértil (Moji, Cubatão, Estuary) and Copebrás (Piaçaguera).

According to data from 2006 provided by CETESB, Cosipa is responsible for most of industrial load with 352.71 ton N yr^{-1} of nitrogen and Ultrafértil CPG is responsible for most of the industrial phosphorus load (35.85 ton P yr^{-1}). For guideline 4 data from effluents were inferred from statistics on the population from the Instituto Brasileiro de Geografia e Estatística (IBGE 2000). Population values were obtained at the level of the smallest administrative unit: ”Bairro”. Data available included population, total households and households connected to public sewerage. Households include also slums improvised houses.

First the percentage was estimated of each ”Bairro” inside each watershed. Assuming that the households are uniformly distributed in the ”Bairro”, population, total households and households connected to public sewerage were multiplied by that fraction, obtaining in this way the amounts inside each watershed. Then the average size of a household was estimated. Multiplying this value by the number of households connected to public sewerage, the number of people connected to public sewerage was estimated. Assuming that there was no treatment of the wastewater, and that each person produced 12 g N day^{-1} and 2.5 g P day^{-1}, an annual value of N and P in sewerage was estimated. These per capita values are proposed by guideline 4. As a first approximation (taking in consideration we do not know the percentage of the population in the unmonitored area which is connected to the outfall) it was assumed that all discharge directly into the estuary.

Using a similar approach as the one mentioned in guideline 4 (Sewage Treatment Works and Sewerage) it was estimated the value of nutrient loads from households not connected to public sewerage. The only difference from the previous approach was that the per capita loads were: 8.5 g N and 1.2 g P per person and per day. These values are proposed by guideline 5. These values are smaller than the ones in untreated public sewerage basically because the sewage that is not connected to the public collectors can take more time to reach the water bodies allowing for retention during the travel time. This hypothesis is not correct for example for systems discharging directly to the estuary or the main rivers.

The Harp guideline 6 provides example methodologies from several signatory countries as to how to quantify nitrogen and phosphorus from diffuse anthropogenic sources and natural background losses. The use of the SWAT model is one of the proposed methodologies and was the one used in this approach. Loads in the river (”load oriented approach”) were estimated based on 5 years of measurements of CETESB in the monitoring places defined in Figure 1. For the period of water quality data, average monthly values of SWAT flows were used, because there were no measured flows in that period. As seen before SWAT was calibrated based on historical measurements in the watershed gauge stations.

MOJI and PIAC present N/P ratios of about two, which is high considering that the urban effluents have a N/P ratio of 5. This is probably related with the industrial effluents released with high concentrations of phosphorus in these sub basins. Retention is a function of temperature, physical characteristics of rivers and lakes, such as residence time (lakes) and specific runoff, hydraulic load and bottom characteristics (rivers). Many of these parameters are difficult to measure, and therefore difficult to implement in calculation procedures. In general, nitrogen retention is more influenced by biological processes than the phosphorus retention, whereas the phosphorus retention is more influenced by sedimentation processes than the nitrogen retention.

In order to arrive at nutrient retention estimates, the German approach described in OSPAR guideline 9 was used. In order to estimate the surface area of water in the monitored watersheds an empirical relation suggested in guideline 9 was used. This relation allows estimating the area of the watershed occupied by rivers. It was assumed that the area of lakes in this watershed was irrelevant. Additionally flow results from SWAT were used to calculate retention using the equations described in this guideline (the same flow results as the ones used in Guideline 7). The parameters in said equation were derived empirically from 100 river basins in Europe. The guideline does not indicate the variability and uncertainty associated with these parameters. Keeping in mind the steep topography, these are fast-flowing rivers and thus it is likely that retention is relatively low.

The application of the HARP Guidelines shows that Load Oriented Approach and Source Oriented Approach gave similar values for CUBA39 station, while for MOJI and PIÇA stations the values were completely different (Table 3). The main reason for the major discrepancies in MOJI and PIAÇA are related to the high measured loads in the rivers in combination with the absence of effluent concentrations from the industries in the watersheds. Table 4 shows that urban effluents have an important contribution to loads of nutrients. However this value is probably overestimated, because part of the population is directly connected to the outfall.

4 DISCUSSION AND CONCLUSIONS

SWAT model was calibrated to generate flows for Santos watersheds. This allowed to produce two types of flow as boundary conditions for the estuary model: i) the main discharges of fresh water if possible associated with the water quality gauge stations ii) the total discharge of fresh water to the estuary. The difference between both these flows is that the drainage area associated with the first one is roughly half of the total drainage area of the estuary. This kind of result is important to improve estuary model results. In fact it only makes sense to impose a discharge in a region where the tide does not influence river flow. Only there we can guarantee a point discharge. Below that point water arrives at the estuary (in high tide) in a diffuse way. It does not make sense to define in the model thousands of diffuse discharges. It is also controversial to combine all the diffuse discharges into only one point discharge. However all these scenarios can be tested with estuary model.

TABLE 3: *Results of the application of HARP Guidelines for the Santos basin.*

	Santos Basin	GL	Total Nitrogen		(tons yr^{-1})		Total Phosphorus (tons yr^{-1})			
	Discharges and losses of N and P		C27	C39	Moji	Pica	C27	C39	Moji	Pica
+	Upstream inputs			408				33		
+	Losses from aquaculture	2	0	0	0	0	0	0	0	0
+	Discharge from industry	3	0	535	0	1	0	101	0	0
+	Discharges from sewage treatment plants	4	5	42	0	0	1	9	0	0
+	Losses households not connected WWTP	5	54	17	3	0	7	2	0	0
+	Diffuse nitrogen losses	6	169	93	100	23	10	6	3	1
=	Sum of all losses/discharges (from Source Orientated Approach)		227	1080	103	24	18	152	3	1
-	Nutrient retention in surface waters	9	15	11	31	3	1	1	5	1
=	Total estimated nutrients at the monitoring point (Source Orientated Approach)		212	1069	73	21	17	151	-2	0
	To be compared with:									
	Total (from Load Orientated Approach)	7	408	1558	997	177	33	184	245	220

TABLE 4: *Results of the application of HARP Guidelines to the unmonitored area of the Santos basin - Source oriented approach.*

	Santos Basin	**GL**	**Total N (tons yr^{-1})**	**Total P (tons yr^{-1})**
	Discharges and losses of N and P		UnMonitored	
+	Upstream inputs		0	0
+	Losses from aquaculture	2	0	0
+	Discharge from industry	3	19	4
+	Discharges from sewage treatment plants	4	4674	974
+	Losses households not connected WWTP	5	0	0
+	Diffuse nitrogen losses	6	753	63
=	Sum of all losses/discharges (from Source Orientated Approach)		5446	1040

In terms of load in the estuary we can discern 4 different main sources: i) load from the rivers in the industrial area; ii) Load from Cosipa; iii) Load from urban origin; iv) Diffuse load from forested watersheds estimated with SWAT. The first source is located in the watershed monitored area, while the remaining three sources are located in the unmonitored area.

Average monthly concentrations were estimated from CETESB measurements of concentration in the river. These values were compared with the known loads in the respective watershed. For Cubatão load, the origins of the loads were justified. For Moji and Piaçaguera the origin of the loads in the river was not justified, but they probably originated from the fertilizer industries located on these watersheds, from which there are no data available on effluents.

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AN ECOLOGICAL MODEL APPLICATION TO THE SANTOS ESTUARY, BRAZIL: TESTING AND VALIDATION

M. MATEUS, A.F.P. SAMPAIO AND S. MATEUS

1 WHY MODEL ECOLOGICAL PROCESSES IN SANTOS ESTUARY?

Over the past decades an intense research program has been carried out at the Santos Estuary to study the ecological dynamics of the system. Much of this work has been field data acquisition in monitoring programs and subsequent interpretation. Modelling approaches have been scarce and generally devoted to the study of hydrodynamics in the bay area or sediment transport in the system (Harari and de Camargo 2003). Until now there has been a lack of integrative modelling approaches in the Santos estuary. This is the first attempt at simulating the hydrodynamics of the system coupled to some of its ecological processes. Since the Santos estuary is a highly impacted system (Braga et al. 2000, Medeiros and Bicego 2004a, b, Abessa et al. 2005, Cesar et al. 2007), the need for such a tool is emphasized by the necessity to link human pressures known in the area with the ecological state of the system according to the DPSIR framework (Mateus and Campuzano, this volume). Among the most relevant aspects that such a tool can address, there are: (1) The insights that it can provide into the major physical-ecological interactions in the estuary; (2) Assessing the contribution of isolated Pressures such as the sewage outfall or industrial effluent to the State of the system; (3) Provide a numerical tool to help in the management of the system (Response) by allowing the testing of different scenarios of human Pressures.

This work presents the application of a coupled physical-ecological model within the MOHID system (Leitão et al., this volume) to the Santos estuary. The ecological model, the MOHID WQ model, is an adaptation of the NPZ modelling paradigm (Fasham et al. 1990). Detailed description of the model implementation is provided, with particular emphasis on the characterization of the forcing and modelling options. Model results are compared against field data to assess model performance and validate the results. This model application relies on the assumption that the model correctly reproduces the hydrodynamics of the Santos estuary.

In the DPSIR framework, this work describes the state of the system based on some ecological and water quality parameters. The model is forced with values previously quantified for the Pressures, namely, nutrient and organic matter concentrations in the rivers, the submarine outfall in the bay, the absent sewage drainage networks in the quarters and direct raw sewage inputs from slum quarters around the margins of the estuary. These Pressures, which fall in one of the major groups of estuarine and coastal area pressures, namely pollution, including urban, industrial, agricultural and aquaculture discharges (Borja et al. 2006), express the forcing from the major drivers: human occupation and industry. The outcome of this numerical study is a benchmark model application which can be further developed to study different scenarios of development or actions prompted by responses. Also, the results presented here will enable the quantification of changes induced by responses, when compared with the results from the model runs for different scenarios.

2 MODEL IMPLEMENTATION

The ecological model is coupled with the hydrodynamic model previously described, and so the same assumptions for the physical features of the system apply here. These are: (1) the water-column is homogeneous, i.e., non-stratified over the modelled domain (2D horizontal); (2) the hydrodynamics in the bay are controlled by the tide and the water flow from the main channels, i.e., there are no shelf water currents. The model domain is the same as for the hydrodynamic application and to reduce the run time, the coarse grid was adopted for the present study. The external conditions include river discharges, forcing functions (for example irradiance, air temperature), and boundary conditions (concentrations of each state variable on the open Atlantic boundary). Whenever possible, these values have been taken from available field data. The ecological model runs with a time step of 1 hr. A constant value for each property was defined as the open boundary condition (Table 1).

2.1 Atmospheric forcing

Climatological irradiance levels were calculated by the model for the domain geographical coordinates from the solar constant corrected for cloudiness with mean monthly cloud cover data. Air temperature and relative humidity were also used to force the model over the entire domain (Figure 1), with monthly values taken from field observations made by CODESP meteorological station at Alemoa during 1997 and cloud cover was taken from field observations made at the Ilha da Moela meteorological station at Guarujá during 1999.

2.2 River inputs

The model considers six river inputs inside the model domain. When considering the ecological dynamics of the system, river discharges also need to be characterized by values of nutrients, organic matter components, and other biogenic constituents. For simulations with the ecological model, the discharge of Cubatão+Henry Borden and Mogi+Piaçaguera are characterized by mean monthly concentrations of ammonium, nitrate, nitrite, phosphate and dissolved oxygen (Figure 2) calculated from data collected by CETESB between 2000 and 2005. The COSIPA industrial effluent is characterized by high nutrient loads and a significant flow for an effluent of this kind. According to Gragnani (1996) most of Moji/Piçaguera flow is caught by Cosipa and dumped in the Cosipa channel. Therefore we have modelled a joint discharge of Moji/Piaçaguera and the Cosipa effluent. The Cosipa, contribution is a flow of 2.38 $m^3\ s^{-1}$ and a concentration of 23 $mg\ l^{-1}$ of ammonium and DON, and 4.6 $mg\ l^{-1}$ of both phosphate and DOP. These values have been estimated from field observations and available monitoring studies from Cetesb-Cubatão. The remaining values as well as the values assumed for the other discharges are summarized in Table 2.

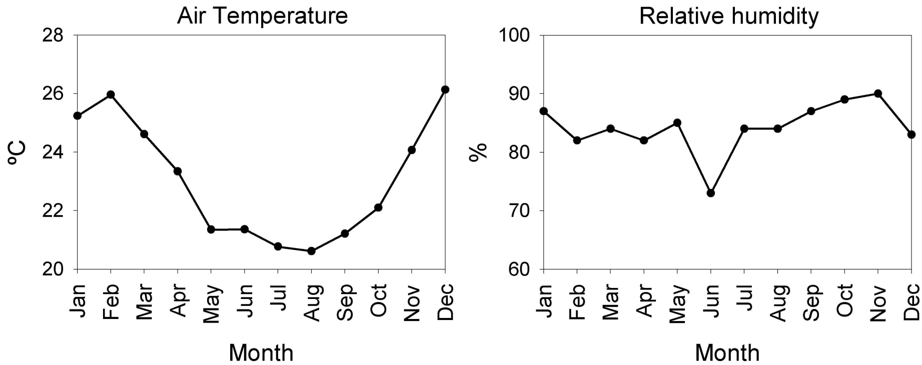

FIGURE 1: *Monthly mean values for air temperature and relative humidity used to force the model.*

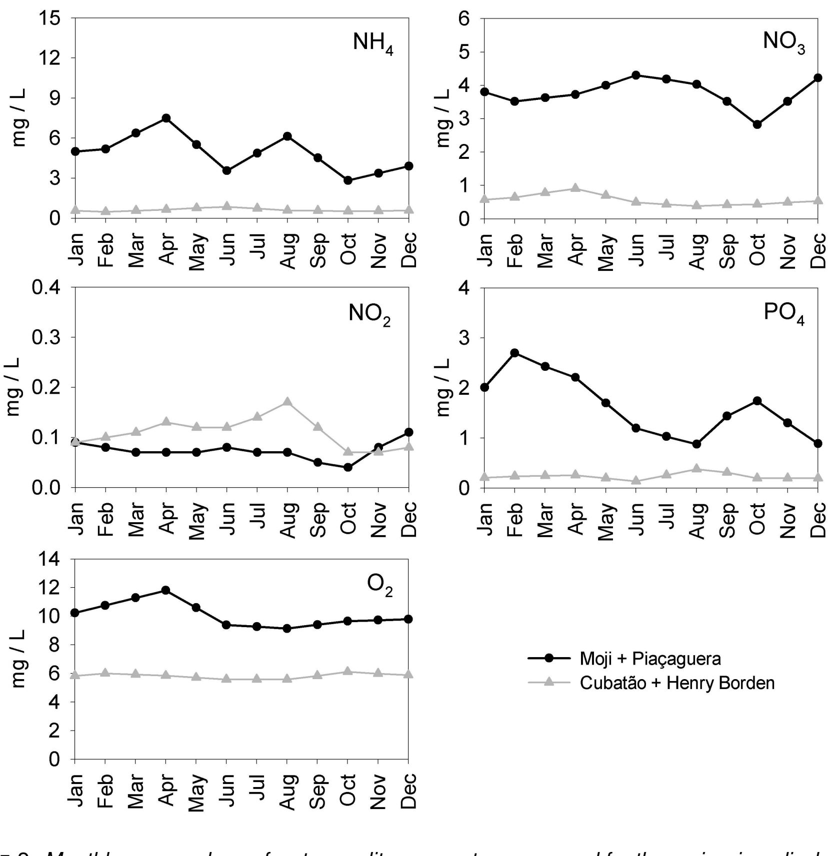

FIGURE 2: *Monthly mean values of water quality parameters assumed for the major river discharges: the joint discharges of Moji + Piaçaguera and Cubatão + Henry Borden.*

TABLE 1: *Initial conditions, river inputs and boundary conditions defined for each variable in the model simulations (unless noted otherwise).*

Properties	Units	Initial conditions	Boundary conditions
Nitrate	$mg\ N\ l^{-1}$	0.09	0.095
Nitrite		0.003	0.001
Ammonium		0.002	0.002
Refractory DON		0.2	0.02
Labile DON		0.02	0.002
PON		0.37	0.0009
Inorganic phosphorus	$mg\ P\ l^{-1}$	0.03	0.03
Refractory DOP		0.028	0.002
Labile DOP		0.0028	0.00028
POP		0.05	0.005
Oxygen	$mg\ O_2\ l^{-1}$	8	9.56
Phytoplankton	$mg\ C\ l^{-1}$	0.05	0.02
Zooplankton		0.03	0.01
Temperature	ºC	20	20
Salinity	PSU	20	36
Cohesive sediments	$mg\ l^{-1}$	100	25

2.3 Sewage nutrient input

As already mentioned, the Santos estuarine system is heavily impacted by human occupation and activities. A significant part of the anthropogenic pressure is in the form of diffuse sewage discharges in several places scattered inside the estuary (Braga et al. 2000, CETESB 2001). Most of these discharges are associated with the slum quarters near (and sometimes over) the water line, and often with absent or provisional sewage drainage systems. Together with the faecal contamination, the discharges into the system also have an associated load of nutrients, both in a mineral and an organic form. To account for this source of eutrophication, the model considers 29 sewage discharge points inside the estuary (Figure 3, left). These comprise the submarine outfall, sewage treatment plants and direct inputs from slum quarters. Each discharge is characterized by flow rate, associated nutrients and dissolved organic matter concentrations. The flow and concentration of sewage treatment plants were calculated from data obtained from SABESP (São Paulo State Basic Sanitation Company) and direct discharges were estimated from the population number calculated for each district by the Brazilian Institute of Geography and Statistics (IBGE 2000), in the municipal data on people living in slums and in measured values of water consumption by neighborhoods (Table 2). This number stands as an estimate based on the households not connected to the municipal sewage system.

Reference flow and concentrations for areas where data were not available were taken from Metcalf and Eddy (2005) values for high-strength wastewater (Table 3). The partition between the different pools of nutrients (organic and mineral) was also based on Metcalf and Eddy

(2005). All discharges are characterized by a cohesive sediment concentration of 120 mg l^{-1}, a salinity of 0.5 and a temperature of 24 °C. Together with the sewage discharges originating from the poor or non-existent sewage system in some areas of Santos, the submarine outfall in the bay is another large source of nutrients. The effluent characteristics used in this simulation are presented in Figure 4.

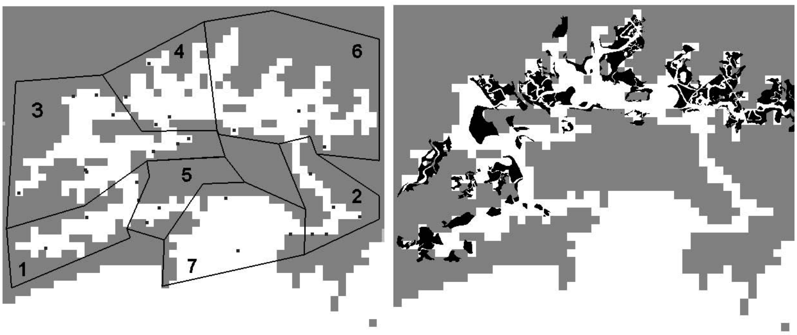

FIGURE 3: *Left: Sewage discharge points associated with the outfall, sewage treatment plants, and the slum quarters and absent sewage drainage networks. The model domain is divided in seven integration boxes to allow the calculation of the exchange fluxes of properties inside the estuarine area between the boxes. Legend for the boxes: São Vicente Channel and Piaçabuçu river (1), Santos channel (2), Boturoca and Mariana river and Pompeba large (3), Largo do Canéu or Cubatão area (4), Lower São Vicente Channel (5), Bagres and Barnabé Island area (6), and Santos bay (7). Right: Mangrove areas used in the model to account for the shading effect.*

2.4 Mangrove shading effect

Considering the large mangrove areas inside the estuaries, the model accounts for the shading effect of mangrove trees. Because a considerable fraction of the mangrove forest is in permanently flooded areas, they strongly reduce the amount of light reaching the water surface. The spatial distribution of mangrove areas (Figure 3, right) was taken from aerial survey photographs and satellite images. This model feature is based on three simple assumptions: (i) the shading effect is quantified as a 50% reduction of the irradiance that reaches the surface, (ii) the mangrove is a climax community, implying that the shading effect is constant, and (iii) there are no net nutrient fluxes between mangrove and water column. The latter assumption is based on studies made in the nearby mangrove forest at Cananeia, where it was observed that the mangrove forest does not export significant amounts of nitrogen to the adjacent lagoon (Carmouze et al. 1998). Apparently, the simultaneous processes of release and incorporation of ammonium explains this occurrence. The explicit modelling of mangrove dynamics would require a complex biochemical reaction model. This, in turn, would make the modelling exercise computationally heavy and compromise the simplicity that management models aim for.

TABLE 2: *Sewage discharge areas and characterization of the respective effluent. Discharges include the outfall, sewage treatment plants (STP), slum quarters and quarters beyond sewage drainage. Values calculated according to measured mean concentrations of N and P of the effluent marked with *. The remaining values were estimated.*

Sewage discharge area	Inhabitants	Flow ($m^3\ s^{-1}$)	N (kg day^{-1})	P (kg day^{-1})
Quietude and Vila Nova	36006	0.057	310.4	44.8
Antartica and Ponte Nova	38120	0.060	356.6	57.8
Sítio Campo	10533	0.011	89.5	12.6
Jd Rio Branco, Mangue Seco and Quarentenário	12356	0.011	148.3	30.9
STP Humaitá*	23018	0.040	97.1	17.5
Trevo*	21522	0.020	258.3	53.8
STP Samaritá	26940	0.040	78.7	8.1
Vale verde and Vl Esperança	7792	0.007	66.2	9.4
Vl Esperança e Vl Natal	8557	0.008	72.7	10.3
Nova Republica e Bolsão	5583	0.005	67.0	14.0
São José e Vila Nova	9555	0.009	81.2	11.5
Vila Pescadores	8340	0.008	100.1	20.9
Vila Criadores	800	0.001	9.6	2.0
Caneleira/Butantã/ Jd São Manoel/Alemoa	9751	0.009	117.0	24.4
Vila Gilda / Sá Catarina / Joquey (Sambaiatuba)	34239	0.032	410.9	85.6
Pompeba / Piçarros / Caxeta and Joquey	13950	0.013	167.4	34.9
Vl Pantanal	3600	0.003	43.2	9.0
Explanada do Barreiros	9215	0.009	78.3	11.1
México 70	28980	0.027	347.8	72.5
Bitaru (Rio d`avó)	15300	0.018	183.6	38.3
Morro José Menino	1233	0.001	14.8	3.1
Ilha diana	175	0.000	2.1	0.4
V. Carvalho (Caixao/ Acaraú/Sta Madalena) and V. Carvalho c/ rede	90242	0.084	1082.9	225.6
Prainha	7552	0.007	90.6	18.9
V. Carvalho (Conceiçãozinha)	7774	0.007	93.3	19.4
Sto Antonio / Sta Clara / Engenho / Flores / Cachoeira / Mangue Seco e Primavera	20782	0.019	249.4	52.0
Sta Cruz	5018	0.003	60.2	12.5
Goes	300	0.000	3.6	0.8
Sta rosa e Vl Ligia	7359	0.007	88.3	18.4
STP Cubatao *	31493	0.200	546.5	113.8
Santos outfall*	520950	2.500	12313.7	1883.5
TOTAL	1,017,035	3.22	16,151	2,673

2.5 The role of benthonic OM mineralization

A simple benthic model is coupled to the pelagic model to account for nutrient diagenesis in the sediment. A fixed mineralization rate of 0.1 d^{-1} is assumed and PON and POP are converted to ammonium and phosphate, respectively, and released to the water column. The stoichiometric balance of oxygen consumption is considered in this process.

2.6 Spin-up

The model is run for a period of one year (July 2004 - July 2005) as a spin up period. Given the control of suspended sediment concentration on the ambient light, special attention was paid to the description of cohesive sediments, both in the water column and in the sediments of the estuary. The methodology adopted for modelling cohesive sediment dynamics consisted of starting the simulations with a layer of 10 kg m^{-2} of cohesive sediments in the entire domain and a constant concentration of 100 mg l^{-1} in the water column. Based on previous modelling experiments, a one year spin-up was considered a reasonable time to achieve a reasonable cohesive sediment pattern inside the estuary with deposition and erosion areas already defined.

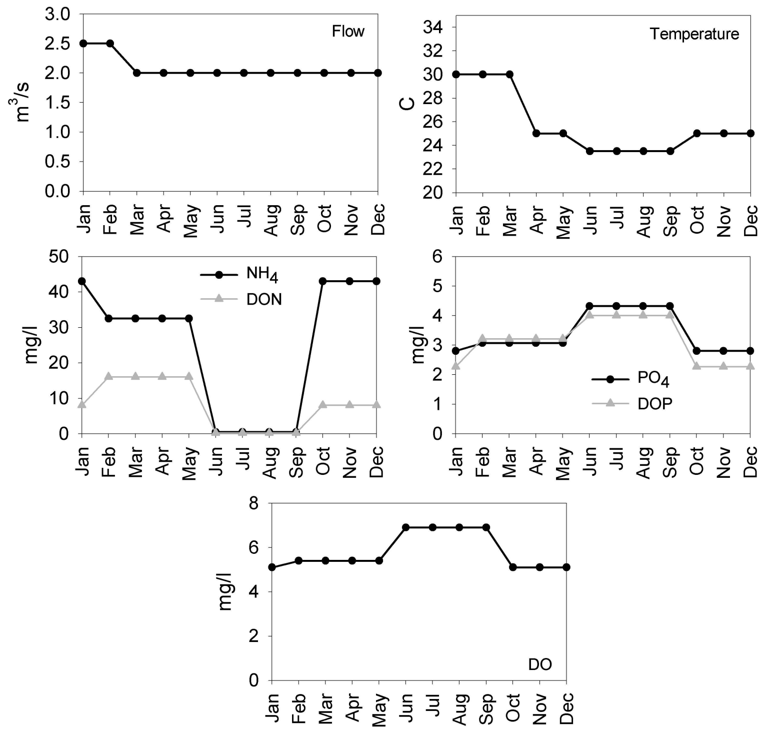

FIGURE 4: *Monthly mean values used to characterize the outfall effluent discharge in Santos bay.*

2.7 Model calibration and validation

Models can be calibrated with field data by adjusting model parameter values until an acceptable simulation is achieved, i.e., reproduce the main features of the modelled system as reflected in the data. Once this is achieved, another simulation is performed and validated with an independent set of data (Thomann and Muller 1982, 1987). If the second simulation is also acceptable then the model is considered valid. This approach has been used in this model application.

The model has been previously calibrated with available data (Braga et al. 2000, Bosquilha 2002, Lima 2003). Due to the scarcity of available field data, the calibration process focused on reproducing the major features of the system and the range of variability as inferred from in situ data. Once this was achieved, the model was then validated with data from two monitoring campaigns in the estuary made during the Ecomanage project. The monitoring program was designed to quantify the spatial. differences in the estuary (eight stations distributed all over the estuary) as well as the seasonal differences by running a campaign in winter (August 2005) and summer (March 2006). The model was validated using the field data for temperature, dissolved oxygen, nutrients (ammonia and phosphate), and chlorophyll.

3 MODEL RESULTS

Any ecological model, irrespective of the level of detail and complexity, produces a considerable volume of results that may vary in relevance depending on the objectives of the model implementation. For the sake of clarity, we have focused on a few state variables (phytoplankton biomass, oxygen concentration, nitrate, phosphate and organic matter),on rates in the form of mass fluxes and on some dimensionless variables such as limitation factors To reduce the bulk of results, only some stations will be discussed. These have been chosen such as to represent distinct areas of the estuary. Also, the model domain has been aggregated into seven regions, or boxes (Figure 3, box labels in the legend). Integration boxes try to capture areas in the system with similar conditions (both biotic and abiotic). This concept is used to allow the assessment of the temporal evolution over larger areas of the system and to calculate net mass fluxes between adjacent boxes, averaging out temporal and spatial variability. These results are then used to plot annual budgets.

3.1 Temperature

Model results for water temperature show the typical seasonal pattern of tropical estuarine systems, namely, high temperatures during the entire year with an increase in the austral summer months (Figure 5 and Figure 6), reaching 32 ^{o}C in some areas. A spatial pattern can also be seen in the results (Figure 6), with higher temperatures in shallow areas inside the estuary, and lower temperatures in the coastal areas. Only the influence of the Cubatão River and Henry Borden is noticeable by the cool water plume associated with the discharge. Spatial differences are not noticeable when comparing the stations (Figure 5) because they are located in the main (deeper) channels where the residence time of water is significantly lower than in the inner, more stagnant and shallower areas.

3.2 Cohesive sediments

The concentration of cohesive sediments in the water column shows a large spatial and temporal variation. The significant difference between the inner areas and the mouth of the estuary is evident in Figure 5, with concentration at P5 (innermost station) peaking above 100 mg l^{-1}, while barely reaching 20 mg l^{-1} at P1. Higher values are observed during the rainy summer months (Figure 6), while in the drier winter months lower concentrations are observed in the entire system. The timing and level of the higher values of suspended sediment in the water column agree with field data where SST maximum values of around 180 mg l^{-1} are reported in January (Moser et al. 2002).

As the major source of sediment input into the system, river discharges have a significant influence on the horizontal patterns. Sediment concentrations are usually higher in the inner parts of the estuary, where rivers are located (Figure 6). This is particularly evident in summer months near Cubatão River, the largest contributor of sediments to the system. Sediment resuspension also contributes to the concentration in the water column, especially in shallow areas where the currents are more intense, i.e., higher bottom stress. This contribution is seen in the marked spring and neap tide cycles in the model results (time series).

3.3 Light

Suspended matter determines the underwater light climate in the water column by reducing down welling light. As a consequence, available light for photosynthesis is mostly controlled by the amount of cohesive sediments in the water column. This implies that the general pattern of irradiance in the water-column closely follows the pattern of cohesive sediment distribution, as shown in Figure 6. Despite the higher concentration of sediments in the water column during summer, irradiance levels are still higher because the irradiance reaching the surface is also higher during this period. In some places the values are >250 watt m^{-2}. Because the model is a 2D application, light is integrated over the entire water column, meaning that in deeper areas less light is available. This explains why in some areas the irradiance is quite low, even when cohesive sediment concentration is also low, and vice-versa. This effect is seen in the bay area (lower irradiance, lower sediment concentrations) and in the inner shallow areas (higher irradiance, higher sediment concentrations), both with values ranging from 0 and 100 watt m^{-2}.

3.4 Nutrients

Model output for ammonium and phosphate concentrations at all stations is plotted in Figure 7. The results show a similar spatial and temporal pattern characterized by: (1) slight seasonal fluctuations, (2) a marked fortnightly frequency induced by the spring-neap cycle, (3) a strong horizontal gradient caused by high concentrations in the inner estuarine area (e.g., stations P4 and P5) where ammonium and phosphate reach the highest levels, $\sim$1.1 mg l^{-1} and $\sim$0.8 mg l^{-1}, respectively. At the outer stations (P1 and P8) these values are always <0.3 mg l^{-1}. These patterns can in part be explained by the river inputs (especially Cubatão) and their seasonality mostly driven by the presence of a rainy season. The rivers account for an increase in the nutrient loads to the system, thus, enriching the inner areas where river discharges are located.

Nutrients are consumed and converted by primary producers as soon as the light conditions allow it, which is possible in inner areas of the estuary close to the nutrient sources. Since there are no outside sources of nutrients as relevant as rivers (the influence of the oceanic boundary conditions is negligible), nutrient concentrations decrease from the inner areas to the oceanic area by consumption and dilution (Figure 9). Nutrient concentrations are slightly higher in the São Vicente channel (when compared to the Santos channel), probably as a result of the many direct sewage discharges in this channel and the less energetic hydrodynamic regime which reduces the dilution, when compared to the Santos channel. The influence of the submarine outfall is noticeable in the spatial distribution of the ammonium concentration, demonstrating the contribution of this nutrient input to the nutrient levels in the bay.

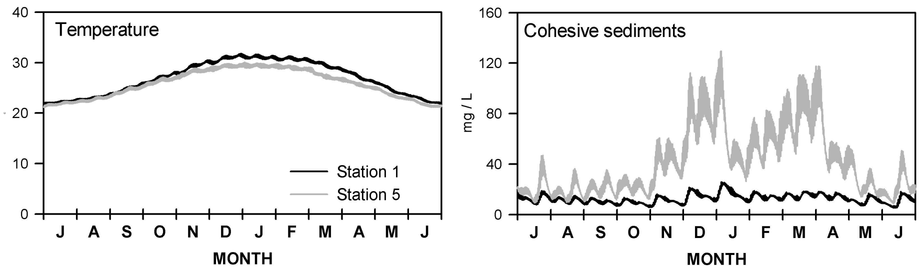

FIGURE 5: *Model results for temperature and cohesive sediment at stations P1 and P5.*

TABLE 3: *Reference flow and concentration values for a high-strength sewage wastewater (Metcalf and Eddy 2005).*

Property	Value
Flow	240 liters per capita*day
Ammonium	45 mg l^{-1}
Dissolved Organic Nitrogen	25 mg l^{-1}
Phosphate	8 mg l^{-1}
Dissolved Organic Phosphorus	4 mg l^{-1}

3.5 Phytoplankton

Being the main driver for primary production, ambient light in the water is the determinant for phytoplankton activity. Any process that affects light penetration in the water can, therefore, influence phytoplankton dynamics in the system. Other factors, such as nutrient availability,water temperature, residence time and grazing pressure, also control the phytoplankton population at any given place and time. This implies that these factors must be taken into consideration when trying to explain the dynamics of producers in any system. The model results show strong fluctuations in phytoplankton biomass (Figure 8 and Figure 9), but without a marked seasonal pattern. This is most obvious by looking at the oscillations seen in station P6 and P7 in Figure 8 where higher values (>3 mg C l^{-1}) are observed between January and May (austral summer and autumn).

The noticeable discrepancy between São Vicente and Santos channels can be partly explained by the different availability of nutrients and light in the two channels. Santos channel has systematically lower concentrations of nutrients given the lesser number of direct discharges in this area, but also because of the short residence time, when compared to São Vicente channel. The residence time controlled by the higher current velocities in this channel is not favorable to the formation of blooms. Contrary to this, the inner (and most stagnant) areas in the São Vicente channel increase retention, thus favoring bloom formation as seen in Figure 9. Also, because of the depth, light availability is usually lower in the Santos channel (Figure 6), imposing a strong limitation on phytoplankton growth.

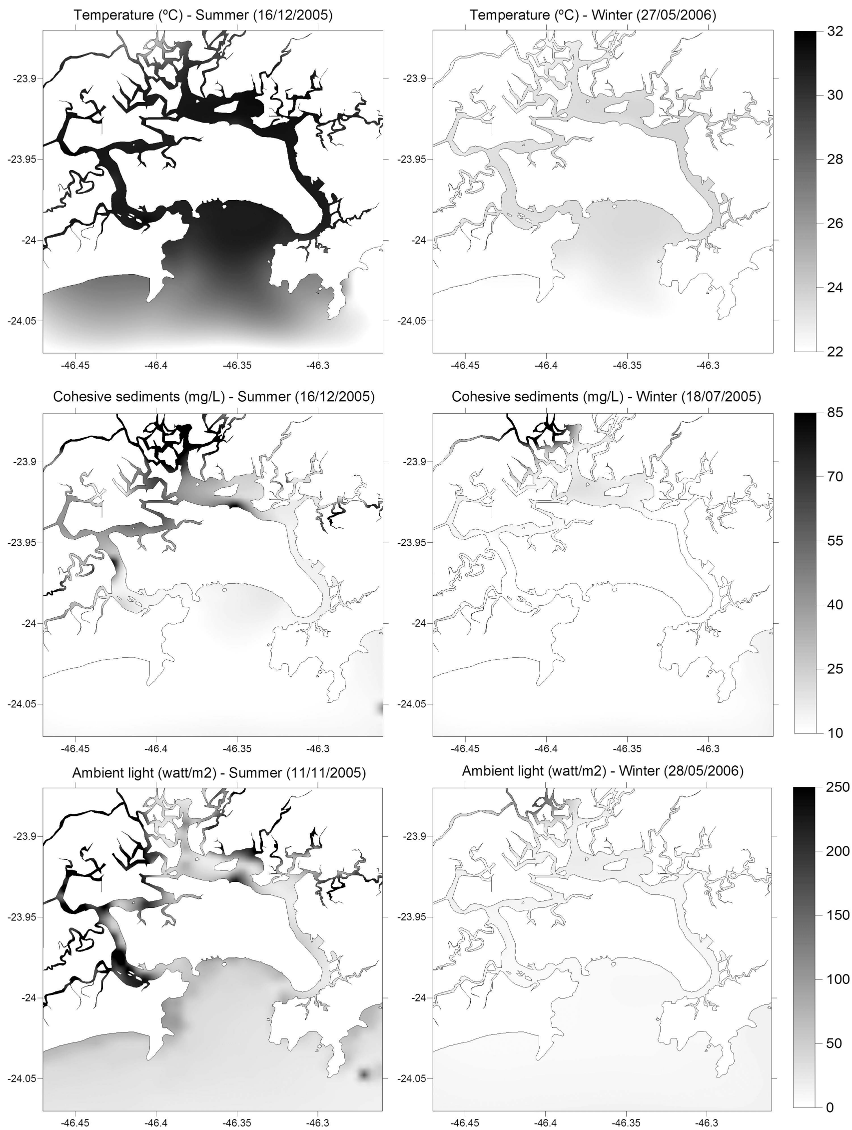

FIGURE 6: *Model results for temperature, cohesive sediments, and irradiance in the water column. Some scales have been expanded to enhance the spatial pattern.*

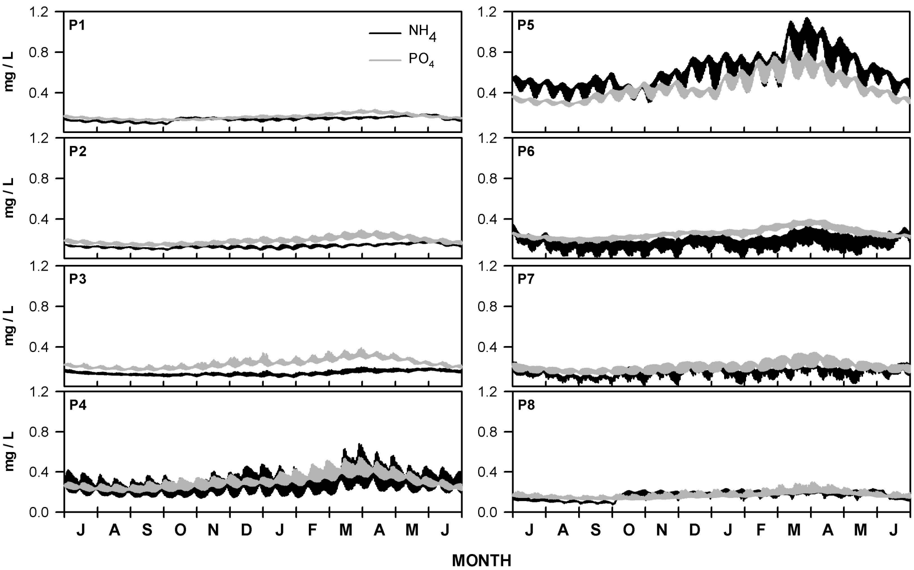

FIGURE 7: *Model results for temperature, cohesive sediments, and irradiance in the water column. Some scales have been expanded to enhance the spatial pattern.*

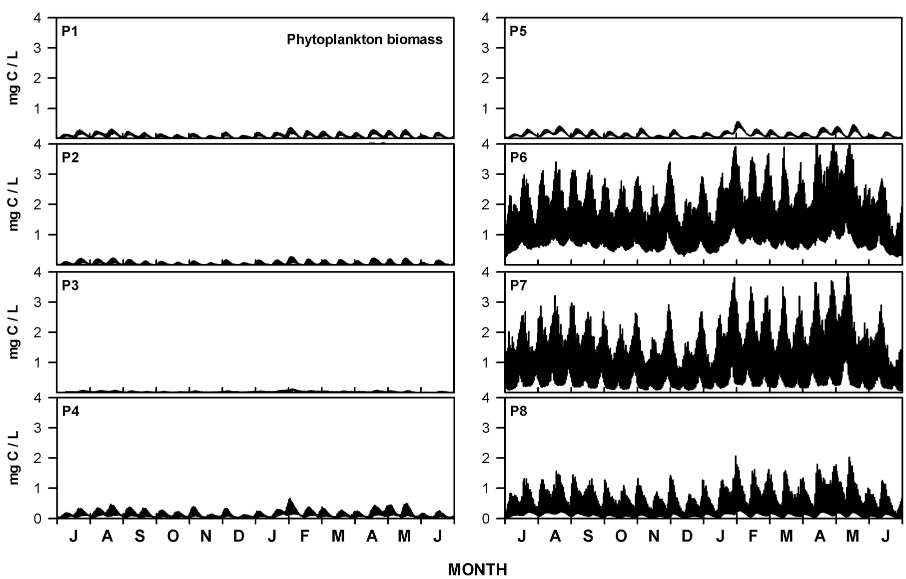

FIGURE 8: *Model results phytoplankton concentration at the monitoring points.*

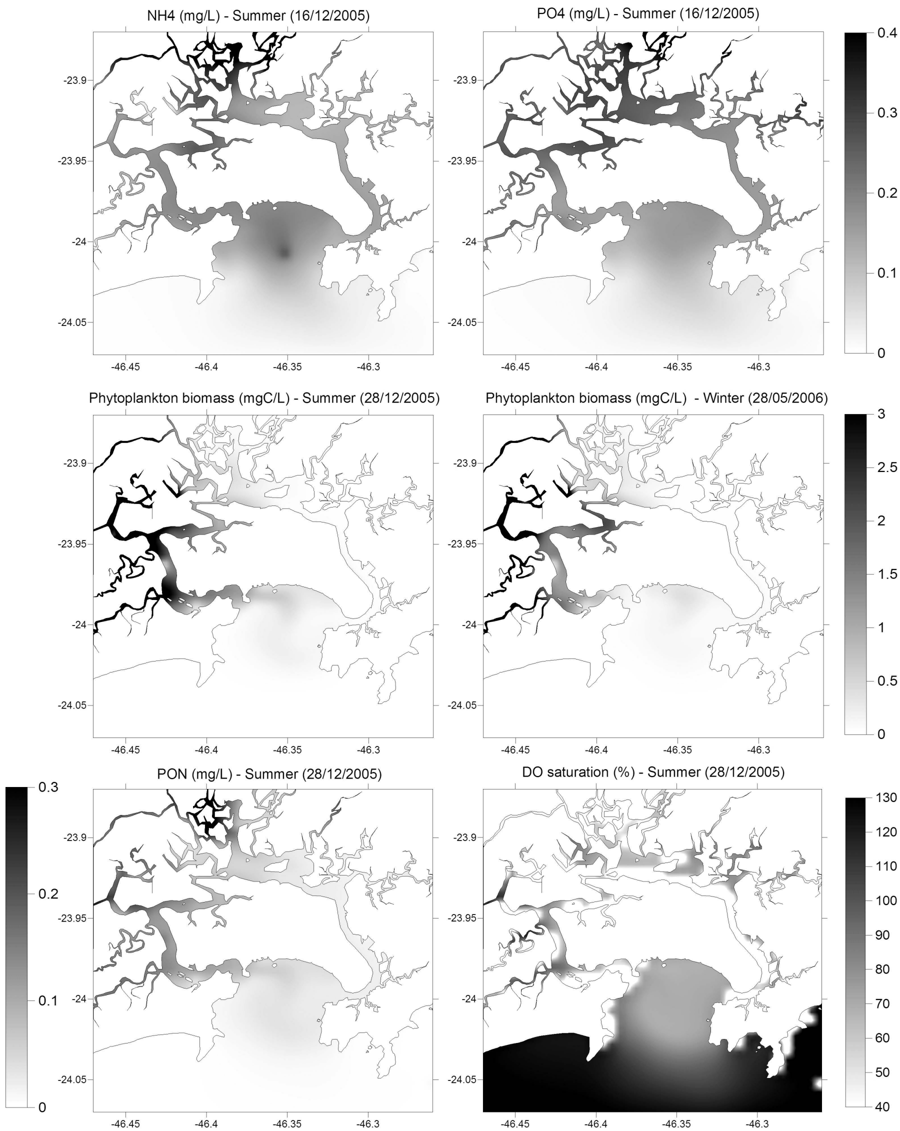

FIGURE 9: *Model results for ammonium, phosphate, phytoplankton biomass, particulate organic nitrogen (PON), and dissolved oxygen saturation.*

Model output allows a detailed assessment of the temporal variation in resource limitation. Overall, the major limitation in the estuary is imposed by light, as seen in Figure 10 for three integration boxes, followed by temperature to a much lesser degree. This limitation is mainly attributed to the high concentrations of cohesive sediments in the inner shallow areas of the estuary and to the depth of the deeper channels and outer area. Despite the high sediment concentrations in the water-column observed in some inner areas, light limitation is lower in these areas when compared with the outer areas. This is illustrated in Figure 10, where Santos channel (box 2) shows a higher limitation by light than the Cubatão area (box 4) located in the upper estuary. No limitation occurs for nutrients and the strong nutrient gradient in the estuary does not translate into an increase in nutrient limitation towards the nutrient-poor outer areas.

Due to the large freshwater contribution, the highest concentrations of dissolved inorganic nutrients are found in the inner areas of the estuary. However, these more eutrophic conditions do not necessarily result in higher phytoplankton biomass since the high sediment concentration in the water and the consequent high light attenuation lead to light limitation of the primary production. A similar mechanism has been reported for the nearby estuarine system of Cananéa-Iguape (Berrera-Alba et al. 2007).

Temperature limitation is almost horizontally homogeneous. Higher limitation is seen in winter and limitation also increases in high summer. This happens because ambient water temperature goes below or above the optimal temperature range. Finally, the model results for zooplankton biomass (Figure 11) suggest that grazing pressure may play a significant role in the control of phytoplankton populations in the more inner areas. As seen for Stations P4 and P5, zooplankton biomass is sometimes higher than phytoplankton biomass. Despite the high concentration of zooplankton at these stations (when compared to phytoplankton), the phytoplankton does not seem to be affected, suggesting high production rates.

3.6 Organic matter

In the present model setting, resuspension from the sediments, detrital material and river discharges are the sources of organic matter. So, in areas where resuspension is frequent (assuming there is PON in the sediments), where biological activity is more intense and under the influence of river discharges, OM concentrations are expected to be higher. Model results for PON (Figure 9 and Figure 12) illustrate this relation, with higher values, peaking at 0.2 $mg\ l^{-1}$, observed in inner areas of the estuary where biological activity is intense and the influence of rivers is maximal. This is also true for the São Vicente channel area when compared with Santos channel. A small seasonal variation is observed in the results. The link to resuspension of PON is not so obvious because places under strong erosion do not necessarily lead to an increase of resuspension of PON. The rationale is that in some areas, PON may be absent from the sediments.

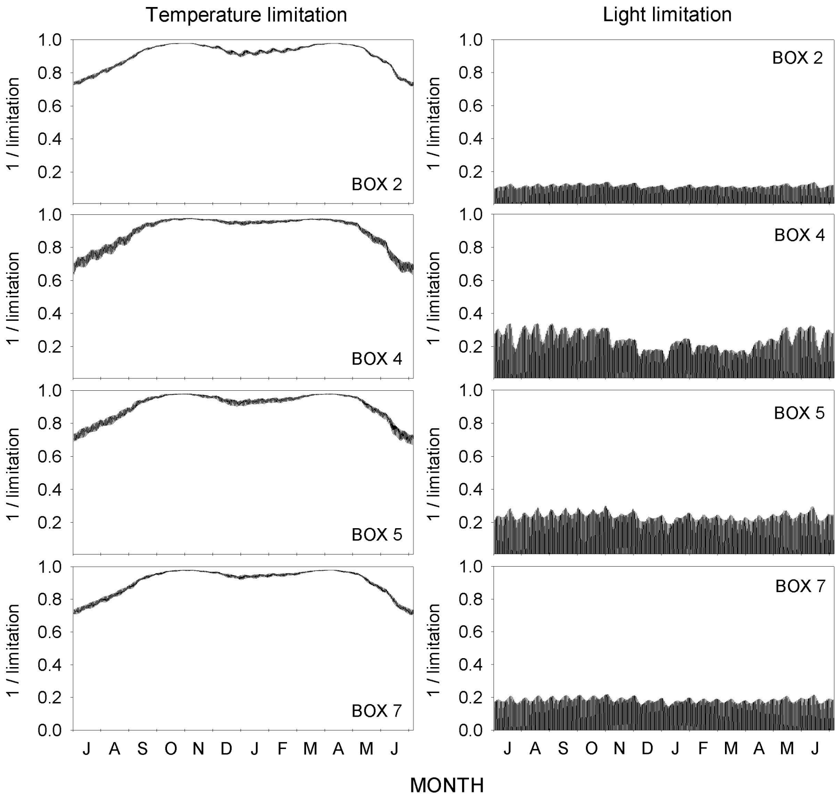

FIGURE 10: *Limitation factors calculate by the model for some integration boxes. Note that values in the Y axis stand for the inverse of limitation, i.e., the higher the value, the lower the limitation (total limitation = 0; no limitation = 1).*

3.7 Dissolved oxygen

Dissolved oxygen in the water is linked with biological activity and also dependent on water temperature. Areas with high phytoplankton biomass are expected to have higher saturations of oxygen (at least during daytime) as a result of photosynthesis. This oxygen production is balanced by the mineralization of organic matter which requires oxygen and produces carbon dioxide. So it is expected that oxygen concentrations and saturation values in the water track the dynamics of phytoplankton and organic matter closely. The interplay of these processes is seen in the horizontal fields of dissolved oxygen saturation (Figure 9). As an example, it is possible to see that saturation values vary considerably inside the estuary, especially near the influence area of the rivers Botoroca, Cubatão, Pereque, and the direct sewage discharges. In this particular case, oxygen consumption may exceed production and the high loads of organic matter at this time of year linked with decreased light availability may be the cause for under-saturation values.

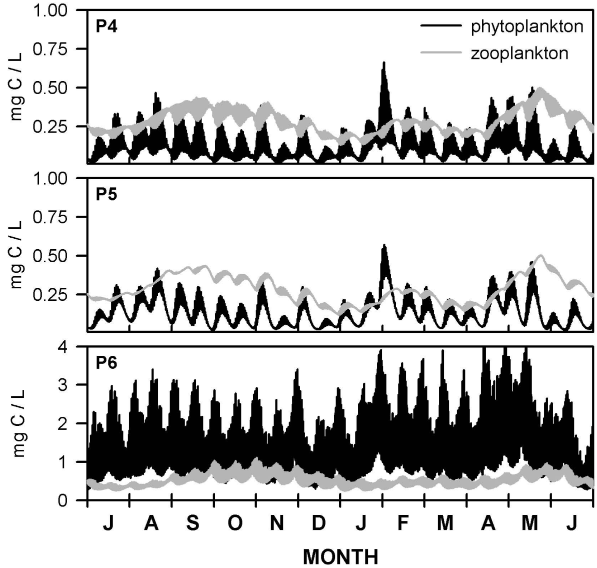

FIGURE 11: *Model results phytoplankton and zooplankton concentration at three monitoring points. Sampling points were selected to consider the inner areas (P4 and P5) and a channel (P6).*

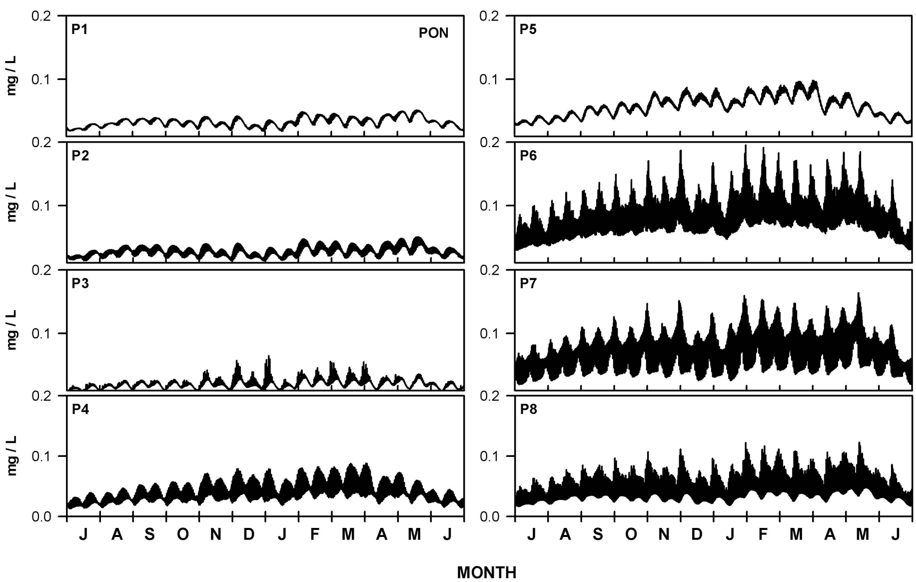

FIGURE 12: *Model results for particulate organic nitrogen (PON) at the monitoring points.*

Supersaturation values are observed in the bay and outer oceanic area, but also in some areas inside the estuary where local conditions cause production to exceed consumption. This pattern has also been observed in the nearby system of Cananéia which shares many physical and biological features with the Santos estuary (Berrera-Alba et al. 2007). This occurrence means that despite the eutrophic state of these systems, oxygen production in the water column can still compensate for the consumption caused by organic matter degradation. The results show that the system can be highly heterotrophic at some places (saturation values below 50%), due to the high loads of allochthonous organic matter, as seen in Figure 9. There is a slight decrease in oxygen concentrations in summer months observed at all stations.

4 MASS FLUXES IN THE ESTUARY

The model results for mass fluxes between different regions in the estuary (integration boxes in Figure 3) of some selected properties are listed in Table 4 and presented in Figure 13. The values were integrated for a period of one year and account for net fluxes across the boundary between adjacent boxes. Considering the boundary of the bay area (box 7) to the open ocean as a hypothetical limit of the estuary, it is possible to look at the model results as an estimate of the export-import of mass from the estuary to the coastal area. The net balance between these two areas is seen in Table 4 in the column 7 to 0. The results show that all described state-variables except oxygen are exported from the estuary to the coastal area. Under these conditions, the model shows that the estuary is exporting nutrients and organic matter (both living and non-living) and importing oxygen. These results suggest that the nutrients that enter in the system via rivers fuel local production but exceed the local needs, thus enriching the system and coastal areas. The oxygen dynamics need further attention because the estuary exports oxygen to the bay via São Vicente channel ($\sim$27 ton yr^{-1}) and much of this production enters the estuary again via Santos channel ($\sim$20 ton yr^{-1}). Because oxygen can be introduced in the system from the atmosphere, no simple inference can be made from the results about the dynamics of the system. However, it can be speculated that the production in some areas can be surpassed by respiration (mineralization of autochthonous and allochthonous organic matter that enters via rivers).

There are other interesting patterns that can be found by looking at some distinct areas of the systems. A striking feature of the system is the net balance between the bay area and both channels (box 7 to box 5 and to box 1). Apparently, the channels have distinct flux dynamics. There is a net positive balance from São Vicente channel (box 5) to the bay (box 7) for all the monitored state-variables. The opposite situation is observed for the fluxes between Santos channel and the bay, where the net fluxes are negative towards the bay, meaning that Santos channel is an entrance route for matter in the estuary. Curiously, a fraction of what is being exported by the São Vicente channel enters the estuary again by Santos channel.

According to model results, Santos channel is a sink for phytoplankton, because only 106 ton yr^{-1} transit from box 2 to 6, whereas 965 ton yr^{-1} transit from box 7 to 2. This sink of phytoplankton explains why the fluxes from Santos channel to Bagres island area (box 2 to 6) of all other variables increase (detrital matter, mineralized nutrients, oxygen consumption). In contrast, Pombeba large area (box 3) is a source of phytoplankton, where the amount that is exported to São Vicente channel (box 1) is more than double the imported quantity from the Cubatão area (box 4). Model results for the fluxes point to an eutrophication of the system.

5 CALIBRATION AND VALIDATION

The calibration and validation exercise was carried out with the aim of qualitatively assessing the model performance. The main purpose was to test the ability of the model to reproduce the overall dynamics of the system and its adequacy as a tool for the prediction of the consequences of different (reduction) scenarios for effluent flow, nutrient loads, etc. To assess the performance of the model, i.e., whether it is able to reproduce the observed spatial and temporal patterns and their variability, model results were compared with observational data. Much of the available data for calibration (Braga et al. 2000, Bosquilha 2002, Moser et al. 2002, Lima 2003) provided significant information on the patterns and variability in nutrient concentrations and organic matter components. After calibration, the model was able to reproduce some of the patterns in the dynamics of the system as presented in these studies.

Validation was made by matching simulated values with field data for the respective properties, as shown in Figure 14. Simulated dynamics correspond well with data from the eight monitoring sites in the estuary. In general, the model produced realistic estimates for temperature, oxygen and nutrients. The magnitude and timing of the phytoplankton peak in in the model reproduced field measurements satisfactorily. A match in values has been regularly achieved, but sometimes not at the same station, as seen in the Chla values for August 2005. The largest mismatches are found regularly at stations 3 and 7. Despite these divergences between model results and data, the model matched validation data with a fair degree of accuracy. The noted discrepancies between modelled and measured values may be due at least partially to our simplification of the dynamics of system components such as the mangroves and inaccurate estimates of the nutrient loads to the system.

6 MODEL PERFORMANCE EVALUATION

The model was able to reproduce the major features of a typical tropical estuarine ecosystem such as Santos Estuary: large temperature variation along the salinity gradient, high mean water temperatures, low light penetration, and variability in the flushing times and sediment and nutrient discharges caused by marked temporal variability in fresh water discharges (Eyre and Balls 1999). The model results reveal a complex interaction of these factors. The seasonal cycle is evident in the results, mostly governed by river discharge associated with the rainy season, and by the light regime (with both temporal and spatial variation). Also, the results show a marked (fortnightly) spring-neap cycle, evident in the time series plots for all properties.

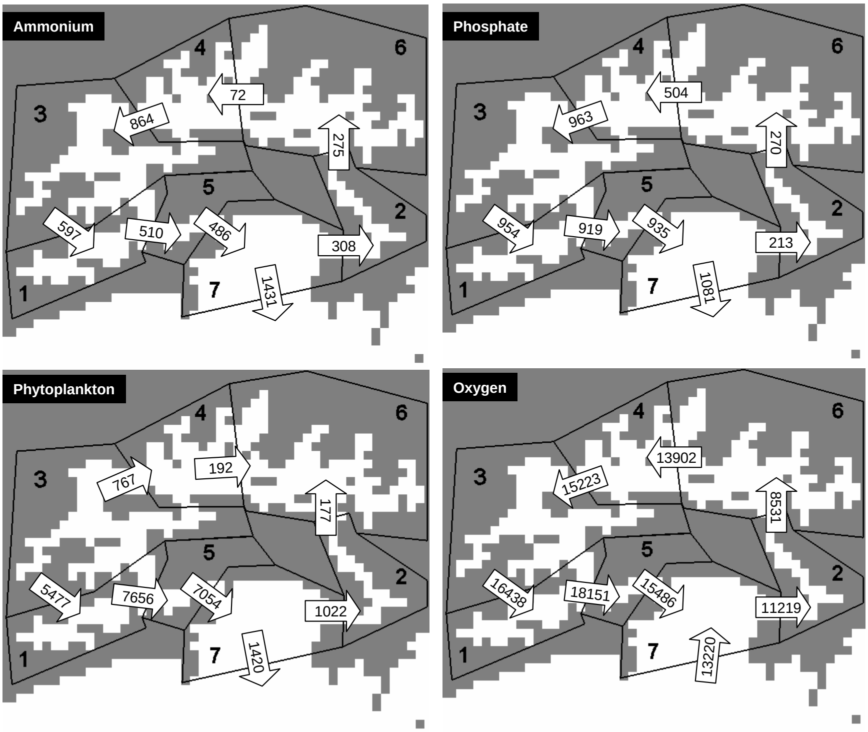

FIGURE 13: *Model estimates for the fluxes of nutrients, phytoplankton biomass and oxygen between different areas of the estuary. All values in metric tones per year.*

TABLE 4: *Mass fluxes between adjacent boxes integrated for one year at the boundary of each box. Positive values mean a net positive balance from one box to the other, while negative values express a negative balance. Column 7 to 0 is relative to fluxes at the boundary of box 7 with the coastal area.*

	mass fluxes between boxes (ton yr-1)							
	1 to 5	**2 to 6**	**3 to 1**	**3 to 4**	**4 to 6**	**5 to 7**	**7 to 0**	**7 to 2**
Ammonium	510	275	597	-864	-72	486	1431	308
Nitrate	2067	742	2332	-2907	-1673	2115	2482	464
Phosphate	919	270	954	-963	-504	935	1081	213
Oxygen	18151	8531	16438	-15223	-13902	15486	-13220	11219
Phytoplankton	7656	177	5477	767	192	7054	1427	1022
PON	407	81	324	-88	16	467	287	92

The organic matter and nutrient concentrations calculated in this study generally fall in the same range as the field data. However, the modelled phosphorus values appear to be higher than the corresponding field data. This suggests that P loads can be overestimated or that the boundary conditions may be too high. From this, we can conclude that a correct characterization of inputs to the system is a prerequisite for a successful calibration effort. The model reproduces the spatial pattern for the properties, where higher values occur in inner estuarine areas, with a limited circulation, and lower values in the main channels with shorter residence times.

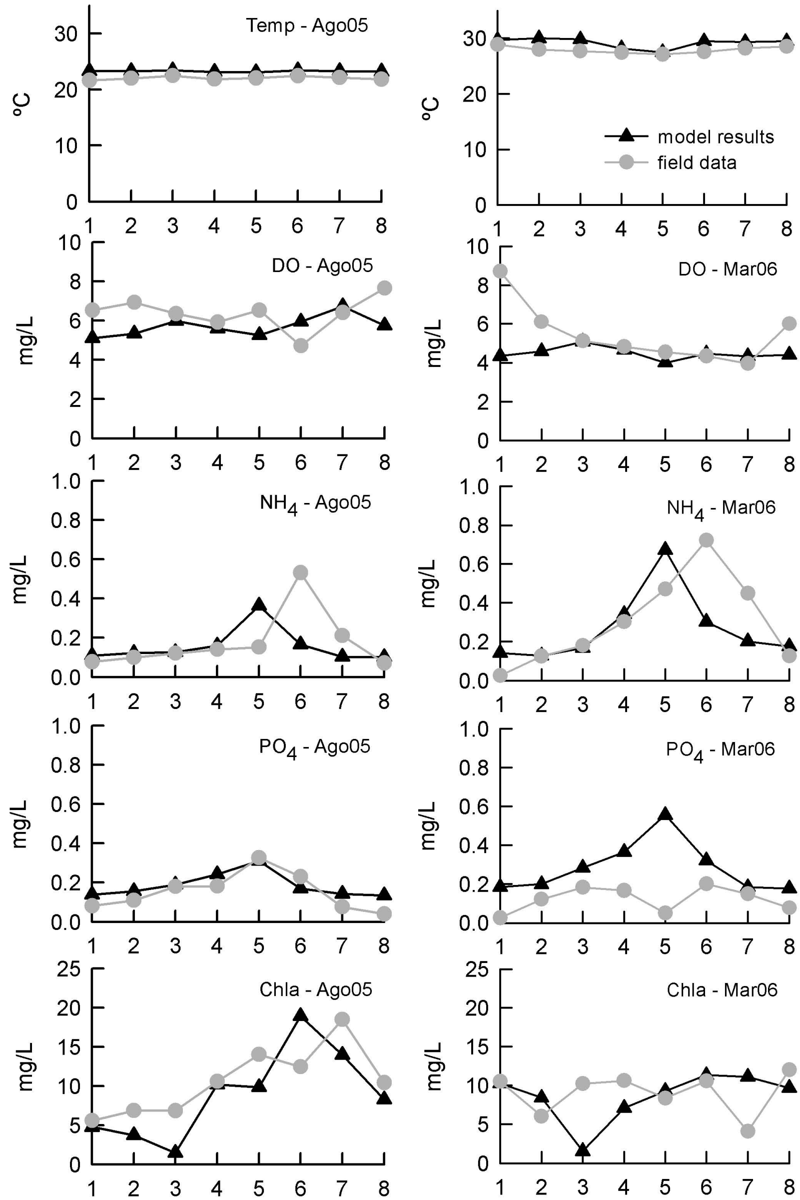

FIGURE 14: *Model output at the monitored stations for several simulated properties (▲) compared with validation data (●). Validation data from the ECOMANAGE monitoring campaigns.*

The model results suggest that São Vicente channel is the main exit route of materials from the estuary, a part of which enters the estuary again via the Santos channel. The model shows a distinct dynamic between these two channels, controlled mainly by the residence time (controlled by physical factors) and the presence of nutrient sources. The residence time in São Vicente channel is higher than in Santos channel, a difference that controls much of the dynamics of material between the channel and the bay. The fast renewal time of water in Santos estuary means that much of the water comes from the bay, which explains the import of material exported from the São Vicente channel.

Although the recycling of nutrients in the system appears to be important, there is a clear control by allochthonous nutrients. Much of the behavior of the system is determined by the large amounts of nutrients (mineral and organic) that are discharged via sewage and rivers. A striking pattern to notice is the oxygen demand in the system, most obvious in the importation of oxygen from the bay (Figure 13). This behavior of the system (as a large bioreactor) poses a challenging demand its management, considering the large anthropogenic pressure expressed in the marked eutrophication of the estuary. In contrast to oxygen, the estuary exports nutrients and organic matter to the coastal area. These results strongly suggest that the Santos estuary is in a highly eutrophic condition, compromising the water quality in the adjacent coastal areas.

Overall, the model is able to reproduce much of the features from the conceptual model derived from field data. The most significant features where the model agrees with data can be summarized in:

- A general pattern in dissolved nutrient concentrations with higher values in the direction of the estuary's head waters, especially in the areas of industrial effluents (e.g., Piaçaguera channel and Largo do Canéu), and with relatively low values in the outer areas close to the open sea. This pattern is shaped mostly by physical mixing processes. Some increases are found near densely populated urban areas where sewage is directly discharged into the estuary. A clear gradient of dilution from the estuary's interior to its mouth is seen along the natural channels (Braga et al. 2000).
- The relation between the spring-neap cycle and the nutrient and phytoplankton concentrations. Phytoplankton has marked variations during neap-tide when nutrient, light availability and residence time all reach maximal values (Moser et al. 2005). Light is usually the limiting factor for phytoplankton growth in estuaries (Cloern et al. 1995, Cloern 1999). The oscillations in light and nutrient conditions between spring and neap tides shape phytoplankton growth. Neap tide conditions enhance the stability in the water column, decreasing the suspended sediments that block the light, and create bloom conditions by increasing the residence time. This pattern has been observed in Bertioga channel, a channel connected to the upper Santos channel (Gianesella et al. 2000), suggesting that the Santos Estuary also shares this feature. Changes in light conditions and flushing ultimately control phytoplankton distribution in the estuary.

- Export of organic matter, inorganic nutrients and phytoplankton to the bay highlighting the contribution of the estuary to the eutrophication of Santos bay, especially during the rainy season when river flow is higher (Moser et al. 2005).
- The increase in particulate matter is mostly from allochthonous sources, not from local phytoplankton production.
- Nutrient limitation does not play a significant role in phytoplankton dynamics. While in the inner parts of the estuary this pattern is explained by the nutrient inputs from rivers and direct sewage discharges, in the bay area this can be explained by the inorganic nutrients contributed by the submarine outfall (Moser et al. 2002, Moser et al. 2004).

7 THE MODEL AS A MANAGEMENT TOOL

For a model to be useful as a management tool it must reproduce the main features of the system under study. This implies that the model must capture the significant processes and interactions between compartments. In short, the model must be able to reproduce the basic elements of the conceptual model of the system.

Under a DPSIR framework, the model establishes a clear relation between anthropogenic nutrient sources (Pressures), the cycling of carbon biomass in the system and oxygen-related problems (State). The outfall emission, river discharges, storm drains and sewage discharges all contribute to its enrichment with nutrients. At the same time, the emissions have a clear impact on oxygen dynamics by increasing the organic matter concentrations, enhancing heterotrophic activity, which in turn contributes still more nutrients to the system. The more complex the system is the more difficult is the task to model it. This is particularly true in cases where the anthropogenic influences are many-faceted, as they are in the Santos estuary. There is always a degree of uncertainty in the load estimates used in the model because of the impossibility to quantify the loads accurately, and this uncertainty negatively affects the modelling exercise. Despite all the uncertainty, the model has the capability to reproduce major features of the ecological conceptual model of Santos and the links between environmental State and human Drivers.

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A MODELLING APPROACH TO THE STUDY OF FAECAL POLLUTION IN THE SANTOS ESTUARY

A.F.P. SAMPAIO, M. MATEUS, R.B. RIBEIRO AND G. BERZIN

1 INTRODUCTION

The need for adequate disposal techniques and sites for urban wastewater has long been recognized. The most convenient place to discharge wastewater, whether treated or untreated, is usually into any nearby body of water. Communities located in estuarine and coastal areas have several alternatives for disposal their wastewater: direct discharge into the sea through submarine outfalls, in the watercourses and tributaries feeding the estuary, and directly in the estuarine waters. The availability of a nearby water body leads many communities to discharge untreated or partially treated wastewater into estuarine and coastal waters. A basic assumption is made that dilution can lessen pollution-related problems. However, mixing in coastal waters is far from complete and in estuaries even less complete, thus leading sometimes to the presence of plumes with high concentration of polluting agents. In more stagnant areas the wastewaters can promote the formation of ”hotspots” in the immediate vicinity of the discharge point, where concentrations rise to significant levels.

Domestic wastewater contains a large number of pathogenic organisms originating from humans who are infected with disease or who are carriers of a particular disease. The most common pathogens found in sewage are those that cause typhoid fever, dysentery, gastroenteritis, diarrhea and cholera. The faecal coliform group of bacteria is usually used as a proxy for pathogenic agents in wastewater. On average, each person discharges from 100 to 400 billion coliform organisms per day, along with many other potential harmful bacteria and virus. The number of viable coliforms in fresh domestic sewage ranges from 10^8 to 10^9 MPN (Most Probable Number) per 100ml (Bishop 1983).

2 FAECAL POLLUTION IN THE SANTOS ESTUARINE SYSTEM

Until recently in Brazil less than 13% of Municipal Wastewater was treated before disposal in a river, lake or ocean (UNEP 2003). As such, the faecal pollution in the Santos area is a common challenge shared with many other Brazilian coastal systems. The main health problems observed in coastal populations in Brazil include the increase and re-emergency of diseases like yellow fever, dengue, malaria, water borne disease (diarrhea, hepatitis, typhoid fever, cholera) and virus diseases (Garreta-Harkot 2003).

This chapter focuses on a model application to assess the health of the marine water bodies along the Santos - São Vicente estuary, with special attention to a specific indicator of faecal contamination, *Escherichia coli*. The need of this analysis was based on the fact that the region - as other coastal metropolitan areas - has densely populated urban areas without a sewage drainage network or sewage treatment and a high number of irregular dwellings. Because of their location close to the mangrove and river banks, a significant part of these housing nuclei drain considerable loads of domestic effluents directly to the estuary. This type of pollution has not been as intensively studied as other research lines developed in the region (which were more frequently associated to the history of chemical pollution of the sediments and the water by industrial activities). However, some works have pointed out the need for these analyses due to the high and increasing rate of faecal coliforms found in the waters of the estuary (Cetesb 2005, 2006). The presence of many slum quarters and quarters without sewage network or treatment, as well as the three sewage treatment plants and the submarine outfall, create a considerable anthropogenic pressure on the aquatic environment, affecting the biota as well as human health in the Santos - São Vicente estuarine system (Braga et al. 2000, Abessa et al. 2005).

The *in-natura* domestic sewage dumping in estuarine channels and rivers has been classified as a potential source of pollution in Santos - São Vicente estuarine system (Braga et al. 2000, Cetesb 2001, Lima 2003, Gianesella 2006). The extent and the degree of contamination as well as the resilience of this environment to the increase of microbiological contamination is still unknown. Numerical models are very useful to estimate contaminant dispersion, particularly faecal coliforms, because they can combine hydrodynamic and water quality processes (Frick et al. 2001). This work presents such an application, based on the MOHID model system (Leitão et al., Mateus and Fernandes, this volume), aiming to validate a faecal decay model for the Santos Estuary so it can be used as a management and predictive tool.

3 MODEL IMPLEMENTATION

The water quality model is coupled with the hydrodynamic model previously described (Leitão et al., this volume) and so the same assumptions for the physical features of the system are valid here. These are: (1) the water-column is not stratified (2D horizontal); (2) the hydrodynamics in the bay are not affected by shelf water circulation. The external conditions include river discharges, forcing functions, like solar radiation and air temperature, and boundary conditions. The simulations were performed with variable T90 decay model for *E. coli* (see Mateus and Fernandes, this volume).

3.1 Atmospheric forcing

Climatological radiation levels were calculated by the model for the domain geographical coordinates. Air temperature, relative humidity and cloud cover (Figure 1) were also used to force the model, with monthly values taken from field observations made at CODESP meteorological station at Alemoa during 1997. For cloud cover the only complete historical series found is from 1999, obtained by observation at the Brazilian Navy's meteorological station located on Moela Island, at a zone adjacent to Santos bay, about 17 km away from the CODESP meteorological station at Alemoa.

3.2 Initial and boundary conditions

The boundary conditions (temperature, salinity, cohesive sediments and *E. coli* concentrations) considered for open Atlantic boundaries and the initial conditions for the model are given in table 1. The model considers six river inputs inside the domain. In two of these rivers (Cubatão + Henry Borden and Moji + Piaçaguera) a monthly mean faecal coliforms concentration was considered (Figure 2) according to a bimonthly value average between 2000 and 2005 obtained from CETESB's monitored points. The pollution point sources used in the model were determined based on current information on the sanitary conditions of the basin, so that the development of a numerical reference scenario for the current situation could therefore be established. The model considers 31 sewage discharge points, three sewage treatment plants (STP) and the submarine outfall in the bay (Figure 3 and table 2). All discharges are characterized by a cohesive sediment concentration of 120 mg l^{-1}, a salinity of 0.5, a temperature of 24 °C.

TABLE 1: *Initial conditions and boundary conditions defined for each property in model simulations.*

Properties	Units	Initial conditions	Boundary conditions
Temperature	°C	20	20
Salinity	psu	20	36
Cohesive sediments	mg l^{-1}	100	25
E. coli	MPN/100ml	0	0

TABLE 2: *Sewage discharge areas, E. coli concentration and the respective effluent flow (m^3 s^{-1}). Discharges include the outfall, sewage treatment plants (STP), slum quarters and quarters out of sewage drainage (Not treated).*

Sewage discharge points	MPN/100ml	Flow m^3 s^{-1}
Not treated	1.00×10^8	0.436
STP Cubatão	3.72×10^5	0.200
STP Humaitá	5.30×10^5	0.040
STP Samaritá	2.30×10^5	0.040
Santos submarine outfall	7.48×10^6	2.500
Total		**3.216**

3.3 Model validation

The model results were validated with *E. coli* field data from two ECOMANAGE campaigns carried out in August 2006 and in April 2007 (Figure 4). These analyses allowed the identification of the degree of microbiological contamination in the water, especially in the interior of the Santos Estuarine System where data of this nature are scarce. The sampling campaigns were carried out by means of a cooperation regime between UNISANTA through ECOMANAGE Project and SABESP - São Paulo State Basic Sanitation Company: the field sampling was carried out by the UNISANTA team and the analyses by the SABESP team.

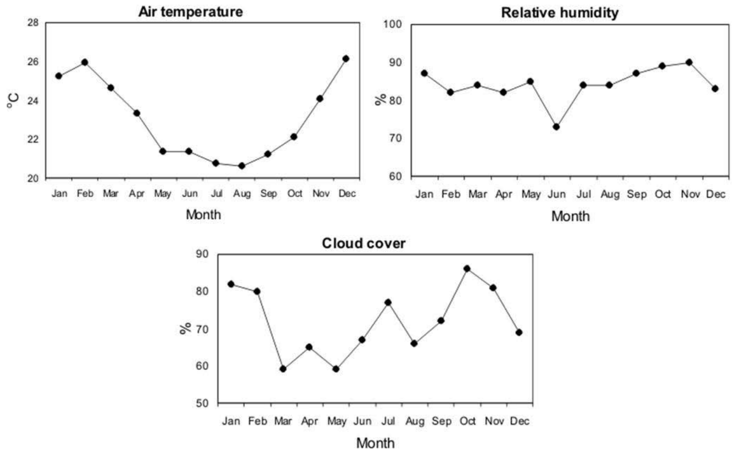

FIGURE 1: *Monthly mean values for air temperature, cloud cover and relative humidity used to force the model.*

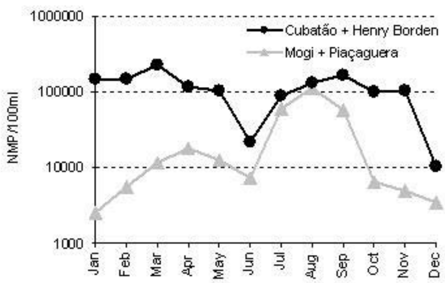

FIGURE 2: *Monthly mean values of E. coli assumed for the major river discharges: the joint discharges of Moji + Piaçaguera and Cubatão + Henry Borden.*

4 MODEL RESULTS

Considering the objective of this work, an analysis of faecal contamination in the Santos Estuarine system, only the model results for faecal coliforms are discussed here. Eight points inside the Santos Estuary (Figure 5) were chosen for the output of model results in order to simplify the analyses. Field data from these stations was used to perform the validation.

The faecal coliform concentration in the water column shows a greater spatial variation (Figure 6 and 7). In the Largo da Pompeba (P6) and Barreiros channel (P7) values above 10^3 MPN/100ml are observed (Figure 6), being the point 6 the most critical in terms of concentration. This higher concentration can be associated with the amount of slum quarters and quarters out without sewage network. Although the Cubatão, Mogi and Piaçaguera rivers discharges contributed with higher concentration of E.coli, the Piaçaguera channel (P5) and Largo do Canéu (P4) areas show concentrations below 100 MPN/100ml.

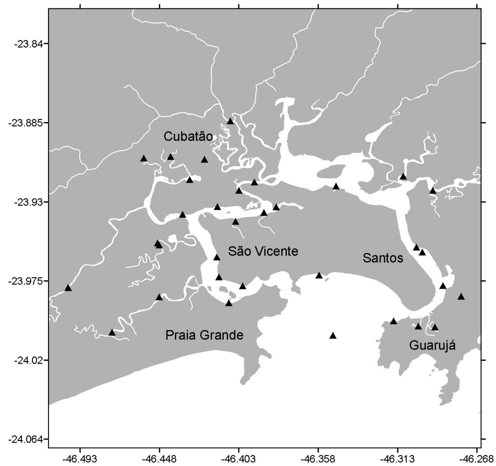

FIGURE 3: *Sewage discharge points defined for the model simulation.*

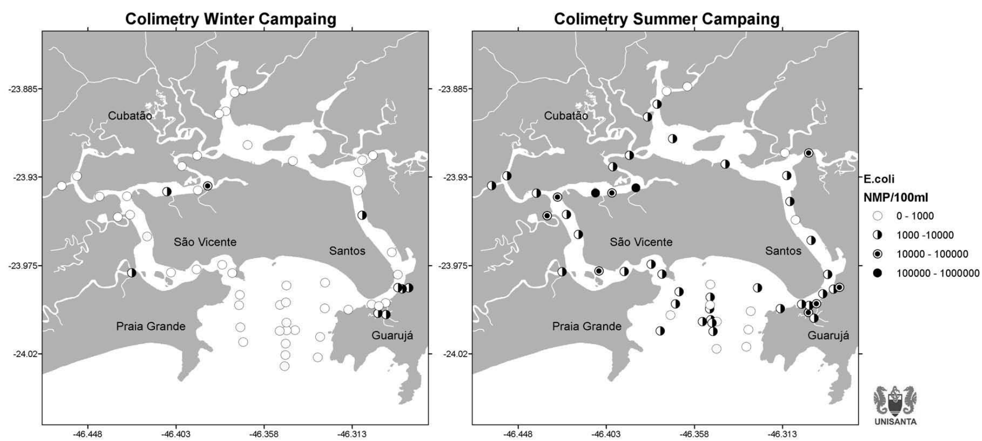

FIGURE 4: *Results from the E. coli data sampling campaign (MPN/100ml) in the summer and winter.*

A reason for this result can be related to the fact that these areas, being shallower, have higher solar radiation levels in the water column, the main agent that controls the survival of enteric bacteria in the water (Sarikaya and Saatci 1995, Serrano et al. 1998). Besides, these points are far from the sources of dumping, so dilution also contributes to the low concentration. At Santos channel (P2) the model shows concentrations below 10^3 MPN/100ml, slightly lower than the measured data. This discrepancy can be related to the discharges of untreated sewage of Vicente de Carvalho besides the slum quarters. Spring and neap tide cycles cause great variation on *E. coli* concentrations, as seen in the model results (Figure 7). A greater variability is seen in the monitored points located in the Santos (P1) and São Vicente (P8) channels, São Vicente having higher concentrations. The concentrations in these areas are regulated by hydrodynamic conditions. During a spring ebb tide the model shows that these channels contribute to increase the faecal contamination at Santos bay (Figure 7), mainly close to the São Vicente channel.

The influence of the submarine outfall at the Santos Bay is clearly noted, but the high concentration of *E. coli* from this source does not seem to reach Santos and São Vicente beaches, as seen in Figure 4. However, it is worth pointing out that the simulated conditions refer to a 2D model integrated in vertical, representing patterns of circulation in Santos forced only by astronomical tide and without the influence of wind.

5 MODEL VALIDATION

Validation was performed by matching simulated values with field data, as illustrated in Figure 8. The model performed very well for the winter, producing realistic estimates for *E. coli* concentrations in most of eight points. Only the results for points 3 and 5 were not so well reproduced. The model estimates for summer did not match results as well as for winter. In the summer only point 2 matched the campaign result. This difference between model results for summer and winter conditions can be attributed to the fact that an elevation in the discharges by urban drainage was not considered for the summer in the model, except for the river flow. Historical data on total and faecal coliform samplings carried out at Santos bay shows a significant elevation in microbiological contamination rates in the rain season coincident with the summer in the region (Sartor 2000, Lima 2003). Besides, one has to consider an increase in sewage generation incurred by the population growth in this season too. The measure campaign performed within the project corroborates this scenario, considering that the number of samples that exceeded the 10^3 MPN/100ml in summer was 5 times higher than winter samples (from a total 57 samples). This variation was observed all over the estuary. Another point that must be considered is that the data used to validate the model has been from surface samples and the model presents the water column average results.

6 FINAL CONSIDERATION AND FUTURE WORK

E. coli concentrations were used in this study as indicators of aquatic pollution and anthropic interference in Santos - São Vicente estuarine system originated from urban residual water dumping on the superficial water bodies. Two microbiological data sampling campaigns performed during the project showed a marked faecal coliform fluctuation between the summer and the winter when compared to each other - where the summer analyses showed a number of points (45) over 10^3 MPN/100ml. The results clearly reveal a seasonal variability within the estuary. The results corroborate the existing literature related to the contribution of urban pluvial drainage to microbiological contamination (Sartor 2000, Lima 2003). Still, in this estuary these simulations tend to become even more complex owing to the need of considering an increase in tourist population, as well as a greater contribution of diffuse loads from pluviometric precipitations, both being a typical summer phenomena in the region.

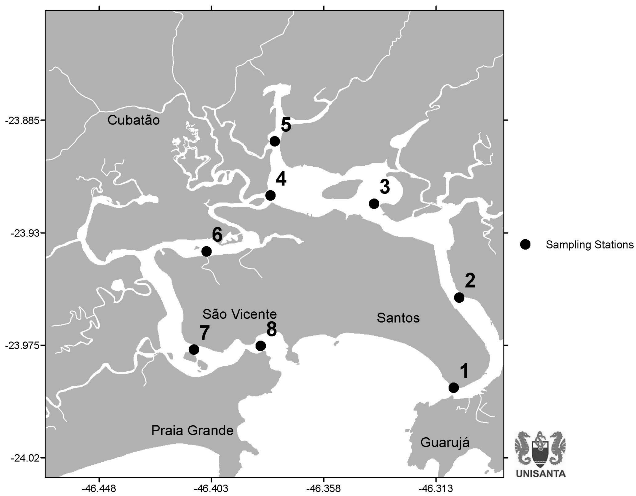

FIGURE 5: *Sampling stations for the data survey campaigns.*

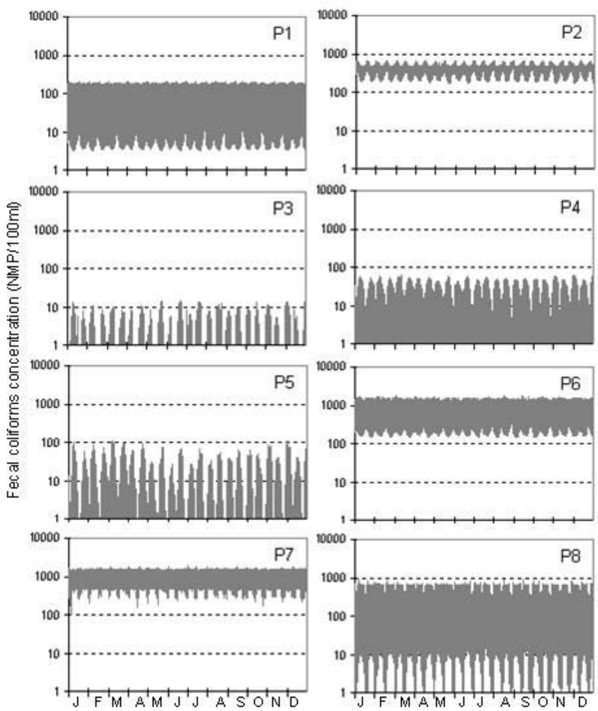

FIGURE 6: *Model output results for E. coli concentration (MPN/100ml) at the eight monitored points.*

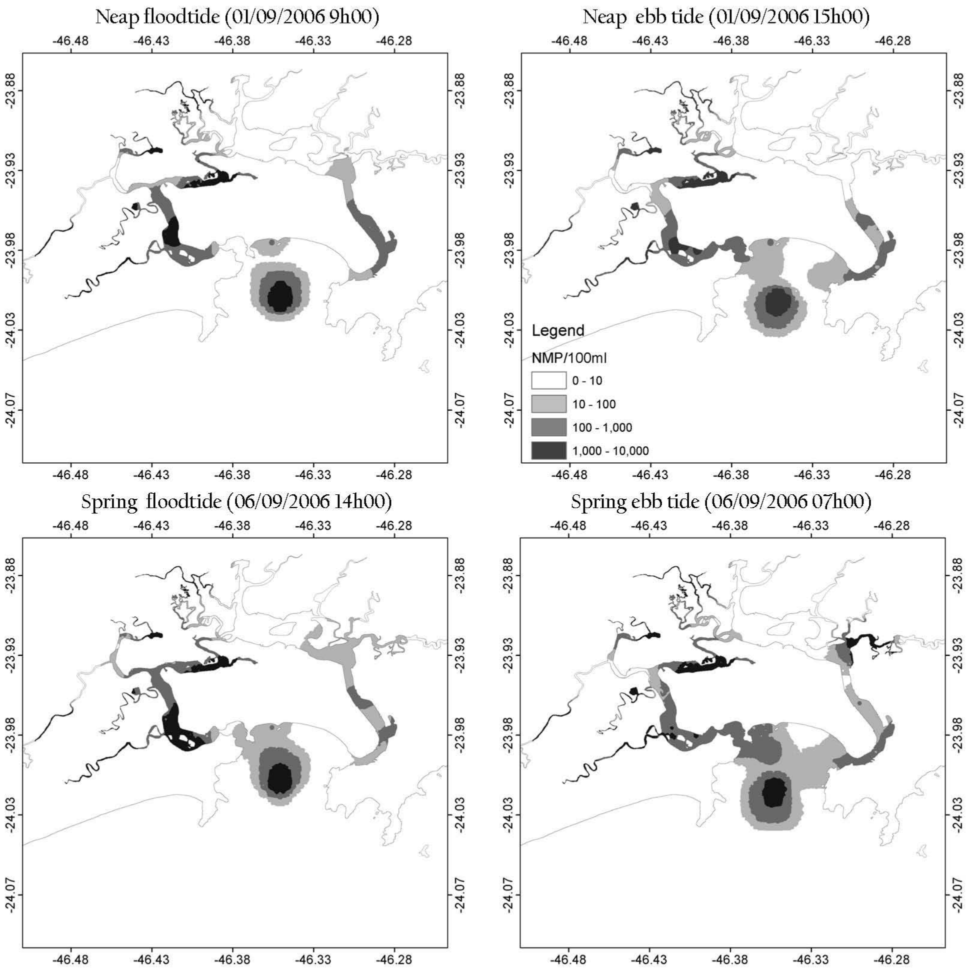

FIGURE 7: *Model output results for E. coli concentration (MPN/100ml) in different tide conditions.*

The model captures the spatial-temporal patterns of *E. coli* dynamics in the system. However, the summer results showed the importance of considering the affluent discharges of pluvial origin typical of this season in future works for a better reproduction of the coliform behavior in this period. Therefore, it is recommended that more research be done to achieve an effective assessment of the role of this contribution in modelling simulations. Even though the solution of the model is integrated for the water column, the comparisons between measures and model results were satisfactory, especially in Santos and São Vicente channels, both characterized by a stronger natural hydrodynamic regime when compared to other areas.

The validation outcome for the dispersion of microbiological plumes has been positive, but the need for a more advanced validation process, based on a broader and systematic field monitoring is evident. Although this sort of ongoing monitoring is an expensive effort, the results are fundamental for a proper understanding of the system's functioning, therefore fundamental for a correct calibration and validation of the numerical models. Together, more field data and modelling simulations will increase the knowledge of the seasonal microbiological behavior in the interior of the estuary. This is of paramount importance to determine spatial-temporal variation in faecal concentrations with more precision and, consequently, perfecting the forecast capability of the modelling system. Such a system is required for a proper management of the Santos bay water quality, given that the source of much of its contamination is located inside the estuary, as suggested by field data and supported by model results.

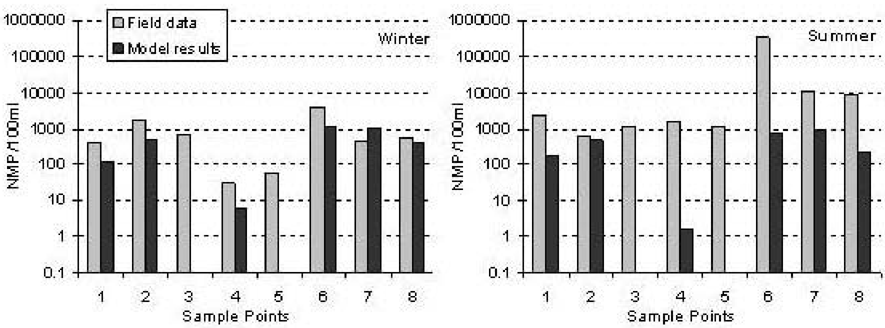

FIGURE 8: *Comparison between model output (black bar) and data field (gray bar) at the monitored sample points.*

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ASSESSING THE IMPACT OF SEVERAL DEVELOPMENT SCENARIOS ON THE WATER QUALITY IN SANTOS ESTUARY

A.F.P. SAMPAIO, M. MATEUS AND R.B. RIBEIRO

1 INTRODUCTION

Growing human populations and associated infrastructures in coastal areas have been gradually destroying the balance between terrestrial and marine environments. Among these human pressures, the impact of sewage inputs and urban runoff has been highlighted as a major global problem requiring urgent action. Faecal pollution is a consequence with serious implications for human health. Eutrophication is another symptom with direct consequences for the ecosystem. Eutrophication is usually recognized as an undesirable effect, a fact that explains the numerous recommendations regarding reduction of nutrient inputs from wastewater treatment issued over the past decades (e.g., Helsinki Comission - HELCOM).

The Santos area faces the same basic problems experienced by many developing countries, such as shortage of adequate housing and lack of sewerage system (Garreta-Harkot 2003). There is growing anthropogenic pressure on the coastal zone resulting from the main socio-economic Drivers that include the rapid increase in the coastal population, industrial development, etc. Input of domestic sewage and estuarine discharge has been pointed out as the main source of eutrophication in this system (Moser et al. 2004), and studies have established a link between the contribution of Santos and São Vicente estuaries as the major cause of eutrophication of Santos bay (Moser et al. 2005).

The development of management policies for the Santos estuary has to deal with faecal pollution and eutrophication-related problems. An informed management for a sustainable use of estuarine services requires the capacity to assess their state, and to predict future states in the face of different development perspectives and choices. The definition of development strategies must consider that the existing Pressures can be mitigated or additional ones can appear. In addition, the overall Impact on the environment must be forecast. Part of this effort can be achieved by testing the result of different development scenarios with numerical model simulations. Models are increasingly becoming indispensable tools in environmental studies and management decisions. In the DPSIR framework models are commonly used to elucidate each component and the relation between different the different components (e.g., the pressures with the state). Combining the DPSIR with numerical models allows the generation of predictions on the potential levels of selected impacts, allowing policy makers to promote responsive remedial or mitigating actions before the predicted impacts manifest themselves in the environment.

This chapter describes such a modelling study, made to assess and forecast the result of four different management scenarios for wastewater disposal in the Santos Estuary. The study aimed to characterize the extension and degree of microbiological contamination and changes in the eutrophic state in the estuarine waters of Santos under different input scenarios of urban discharges and temporal and spatial evolution. This study does not address the feedback implications that each development scenario has on the socioeconomic activities in the Santos area.

2 DEVELOPMENT SCENARIOS

The simulated scenarios include a number of possible scenarios within the field of probable. The conditions that range from the ”business as usual” scenario, i.e., no changes to the actual conditions, meaning that no actions are taken or management policies implemented to improve the sewage treatment and disposal, to an ”optimum case” scenario (hypothetical). In this last scenario the population growth and its impact is minimized by adequate management procedures that significantly reduce untreated domestic sewage discharges into the estuary. The ”business as usual” scenario is used as the reference condition.

Two intermediate scenarios have been projected having in mind the changes that are expected to occur in the estuary as a result of projected improvements in the sewage drainage system, such as the implementation of drainage systems in some slum quarters already urbanized, the construction and extension of the actual system in other areas, and the treatment of sewage effluent prior to disposal inside the estuary. These scenarios result from a prospective analysis that was made considering the main Drivers in the system. Finally, the fourth scenario, the ”optimum case”, is based on hypothesized desirable conditions.

The sewage input for all scenarios was estimated according to current sanitary conditions, future projects in basic sanitation foreseen for the region and one hypothetic scenario. A qualitative and quantitative secondary data survey on urban discharges was carried out on the model’s land boundary, that is, all over the estuary’s drainage basin, in order to characterize and estimate the pressures resulting from current urban occupation. This estimate was based on the data survey of a group of socioeconomic components related to the homes’ sanitary characteristics described by Sampaio and Ferreira (this volume). Their data were obtained from the last demographic census (IBGE 2000) and from internal data from the Companhia de Saneamento Básico do Estado de São Paulo (São Paulo State Basic Sanitation Company - SABESP). By cross-checking these sources of information, it was possible to estimate quantitatively and more accurately the volumes drained to the estuary without treatment from the homes without a connection to the sewage drainage network and the loads of domestic effluents from the sewage treatment stations and the submarine outfall.

The discharge points defined in the simulations (Figure 1) were set based on current information on the sanitary conditions of the basin, so that a numerical reference scenario for the current situation could be established. Once this stage - consisting of the implementation, calibration and validation of a water quality model - had been accomplished, it was possible to prospect future development conditions. Thus, the aspects that involve the lack of coverage of urban sanitary services were also analyzed in a projection towards 2010 and 2015, to assess the impact of population growth and the possible advancements of public policies foreseen in this temporal window. Besides, the simulation of a hypothetical scenario - where all the loads that were not addressed by sanitary projects until 2015 were directed to treatment stations or to the existing outfall - was included in order to verify what the results achieved were.

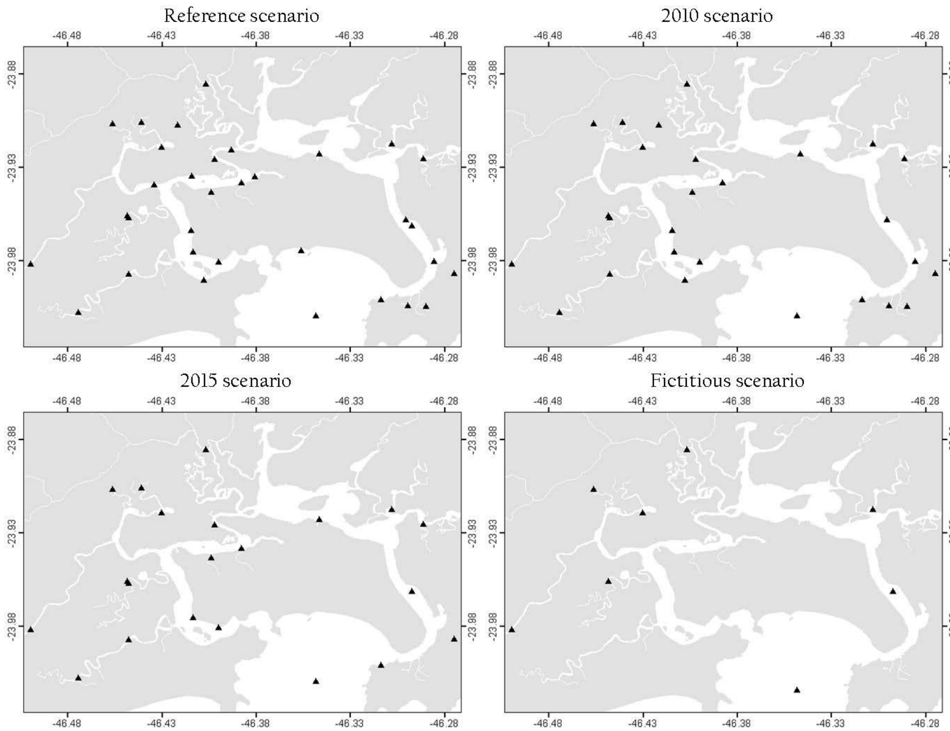

FIGURE 1: *Sewage discharge points for each scenario.*

2.1 Scenario 1: "business as usual"

This scenario can be seen as the "worst case scenario" since it implies that the actual situation is not improved by remedial actions. This scenario considers 31 sewage discharge points, three sewage treatment plants (STP) and the submarine outfall. This is considered as the reference scenario and has been addresses by Sampaio et al. (this volume) for the faecal pollution and by Mateus et al. (this volume) for the ecology.

2.2 Scenario 2

This scenario (2010 scenario) considers 27 sewage discharge points that include the improvements in the sewage drainage systems provided by the works announced for the region by the federal government and also an increase in sewage generation incurred by the estimated populational growth. Among the major changes from the reference scenario (Scenario 1) is the diversion of some sewage discharges from Piaçabuçu River to the Praia Grande outfall, part of the sewage diverted to the Santos outfall in the bay, the increase in the loads at Site 8 as a result of urban development in the surrounding area, and an increase in the discharges at Pompeba area.

2.3 Scenario 3

This scenario is based on predictions for the year 2015 for population growth and the sewage system improvement work that will be developed from 2010 to 2015. It considers 21 sewage discharge points. Compared to the reference situation, in this scenario the water treatment plan (WTP) flow at Cubatão is over three times higher, there is a new WTP at Santos channel, the diversion of some sewage discharges from Santos channel to the Guarujá outfall. Also, the sewage outfall flow in the bay was increased by 0.5 $m^3\ s^{-1}$ and its point of disposal moved 400 meters seawards. Improvements from "Onda Limpa", a great sanitary project (Diário do Senado Federal 2004) were included in addition to the previous works already foreseen in the 2010 scenario. Scenarios 2 and 3 were designed according to data provided by stakeholders, and existing chronograms being that analysis based upon the dates predicted for the end of the works.

2.4 Scenario 4: "optimum case"

This is a fictitious scenario where all but three of the remaining slum quarters are connected to the sewage drainage network and directed to the submarine outfall or to sewage treatment plants. The loads have been calculated for the expected population in 2015.

3 MODEL SIMULATIONS

The water quality model used here to simulate the ecological dynamics and faecal pollution have been previously calibrated and validated (see Mateus et al., Sampaio et al., this volume) and the results presented in these previous modelling studies used to characterize the reference state. As such, the modelling settings used in the present study (forcing functions, initial conditions, assumptions, etc.) are the same as defined in these chapters. The only differences between scenarios are the discharge points inside the estuary and the loads.

Unlike the majority of water quality indicators, faecal contamination is a bioindicator. This means that it's persistence time in aquatic systems is short, usually ranging from less than an hour to a day, and occasionally up to a few days days. Thus, even in intense hydrodynamic regimes, the range of their spreading is limited. The short life span of these organisms means that modelling fecal decay in aquatic systems can be done with small-scale applications and short-time runs. As such, the simulations to assess faecal pollution were made for a period of one month to comprise a full spring-neap cycle. The period of simulation for the ecological model, on the other hand, was set to 10 years to allow the system to stabilize under the new forcing conditions imposed in each scenario. The nutrient loads for each scenario were calculated based on the population growth estimate and according to the methodology previously described by Mateus et al. (this volume). In Table 1 are the sewage inputs of faecal coliforms concentration for the Submarine outfall, sewage treatment plant (STP), slum quarters and quarters out of sewage drainage (not treated) and the respective effluent flow for the sewage discharge, both used for all scenarios.

TABLE 1: *Sewage discharge points, the fecal coliform concentration and the respective effluent flow (m^3 s^{-1}) for all scenarios. Discharges include the outfall, sewage treatment plants (STP), slum quarters and quarters beyond sewage drainage (Not treated). (*) Estimated value for new STP.*

Sewage discharge points	MPN/100ml	Reference	2010	2015	Fictitious
Not treated	1.00×10^8	0.436	0.361	0.134	0.012
STP Cubatão	3.72×10^5	0.200	0.210	0.322	0.347
STP Humaitá	5.30×10^5	0.040	0.042	0.049	0.061
STP Samaritá	2.30×10^5	0.040	0.046	0.049	0.094
Santos submarine outfall	7.48×10^6	2.500	2.674	3.003	3.271
STP Vicente de Carvalho	5.00×10^5 *	---	---	0.267	0.273
	Total	**3.216**	**3.333**	**3.824**	**4.058**

4 MODEL RESULTS FOR THE DEVELOPMENT SCENARIOS

4.1 Faecal pollution

Regarding scenario 2 (2010 scenario), although some changes were made compared to the reference scenario, such as reducing the *in-natura* sewer contribution in the estuary, the model showed that there was an increase in faecal contamination in many places (Figure 2), mainly in the areas previously described as critical, like point 6. This was due to population growth, resulting in an increase of the discharges (flows and loads). Therefore, in general, the alterations attributed to this scenario showed that there was no significant improvement in the quality of the water of the estuary, as a consequence of population growth.

Results for scenario 3 (2015 scenario) show a significant reduction on *E. coli* concentrations when compared to the previous scenarios (Figure 1 and 2). The main reduction was observed in the Santos channel (P1 and P2), with the sewage treatment from Vicente de Carvalho quarters through the implementation of a new STP and the removal of three discharge sources associated to irregular dwellings in Guarujá to the outside of the basin. In São Vicente, the Jockey Club quarter link in the sewer line only caused a small improvement of water quality at Largo da Pompeba area (P6), probably due to the significant number of slum quarters in this area. *E. coli* concentrations are still close to 10^3 MPN/100ml. Concentrations at São Vicente channel are still high, more likely due to Mexico 70 slum quarter contribution (a densely populated slum quarter). This sewage input has a great impact on the water quality of the place.

However, the influence of São Vicente channel at Santos - São Vicente bay has been greatly reduced (Figure 3), compared to the reference scenario.

Finally, for scenario 4 (fictional) it is obvious that the connection of untreated sewage areas with the sewer line (thus assuming that the whole population has basic sanitation) causes a great improvement in water quality at the Santos estuarine system, as the model results for this scenario clearly corroborate. (Figure 2 and 3).

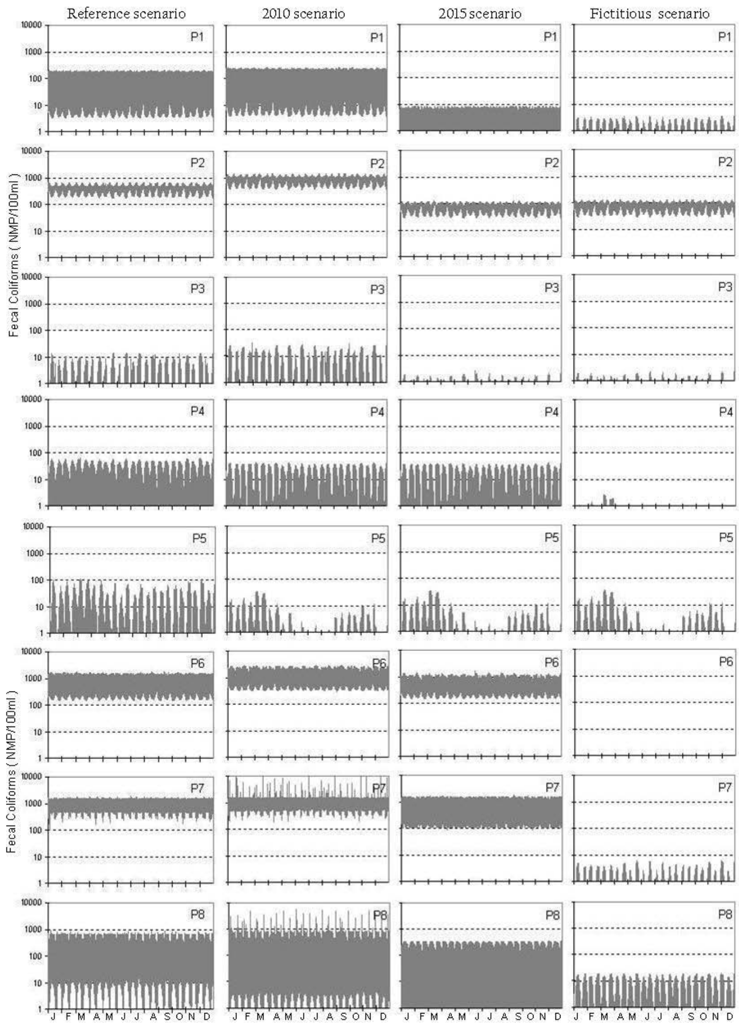

FIGURE 2: *Model results for E. coli concentration (MPN/100ml) at the monitored points for each scenario.*

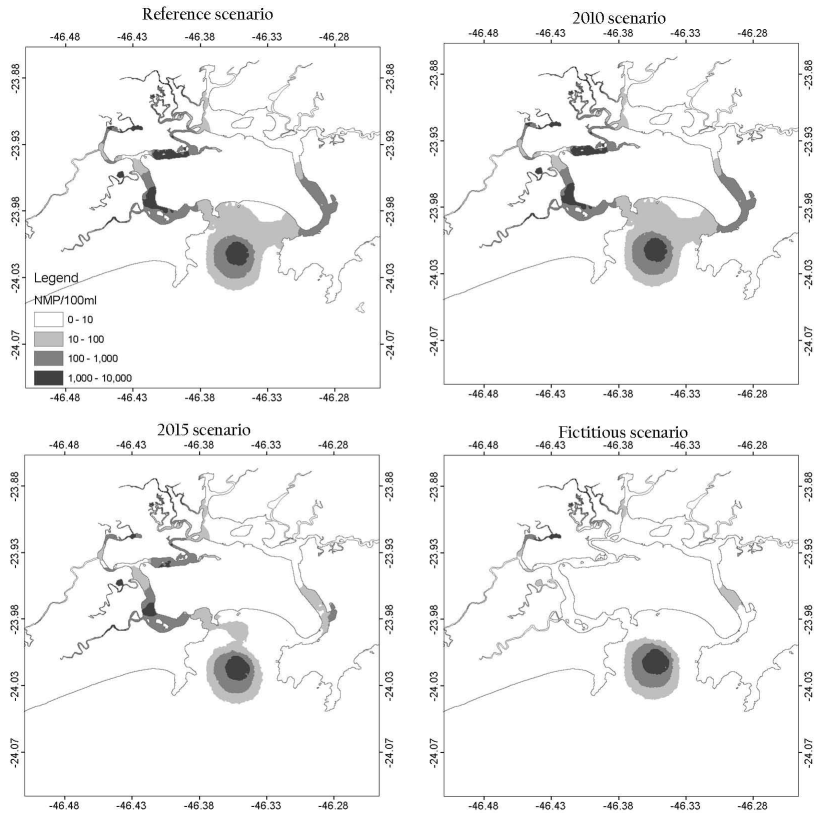

FIGURE 3: *Model results for fecal E. coli (MPN/100ml) for each sceanrio (same tidal conditions).*

4.2 Ecological status

The results obtained by the modification of the forcing conditions in the model (in terms of nutrient loads and input locations in the system) to generate a range of ”development scenarios”, generally conformed to expectations. Because nutrient inputs did not decrease from scenario 1 to scenario 2, and only slightly decreased from scenario 1 to scenario 3, the concentration of nutrients in the system is not significantly affected, as seen in Figure 4 for ammonia at various locations in the estuary. The most significant change is found when comparing the reference scenario with scenario 4. This is an expected occurrence since a part of the sewage has been diverted to the submarine outfall.

The decrease in nutrient concentrations in the system is not reflected in a general decrease in phytoplankton biomass (Figure 5). This outcome is not surprising considering that the estuary is a light limited system. As such, the magnitude of the reduction in nutrients achieved by the changes in each scenario does not seem to affect phytoplankton. The most expressive change is seen in scenario 4 at Station 8, suggesting that in this part of the estuary the conditions of this scenario are more significant.

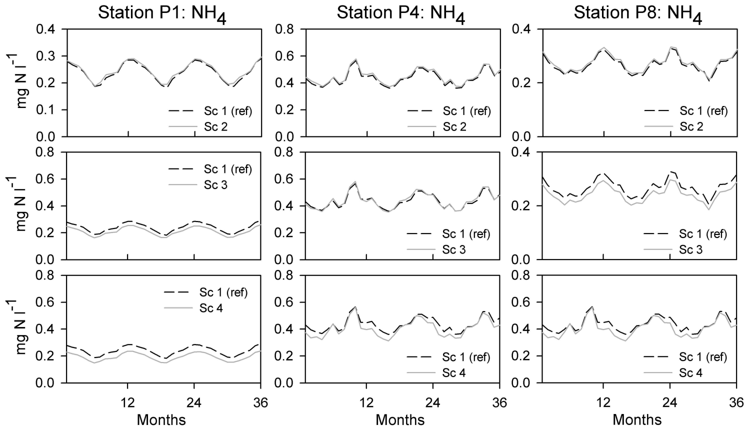

FIGURE 4: *Model results for ammonia at Stations P1, P4 and P8. Last three years of the 10-year simulation run.*

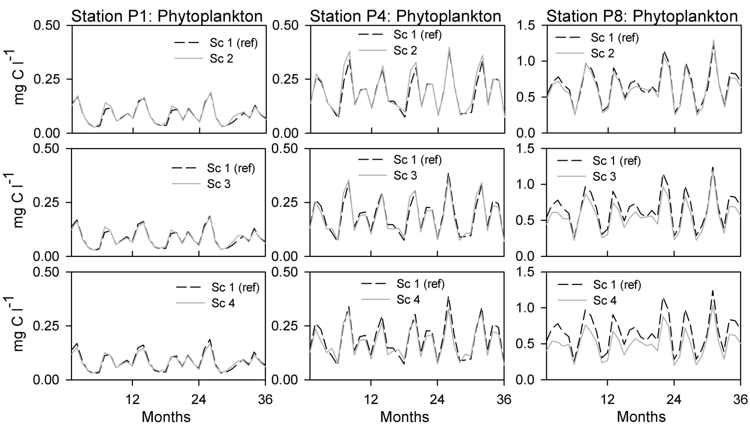

FIGURE 5: *Model results for phytoplankton at Stations P1, P4 and P8. Last three years of the 10-year simulation run.*

Dissolved oxygen concentration is one of the most important water quality parameters given the role of oxygen in the dynamics of aquatic systems. So, the health of a system can be assessed by looking at oxygen saturation in the water. The scenarios 3 and 4 tested in this study imply a reduction of nutrient loads to the system. Since a fraction of that nutrient load corresponds to particulate material, reductions in these inputs mean less bacterial activity, thus lower oxygen demand. The result of reducing organic matter inputs to the system is clearly seen in the results for dissolved oxygen concentrations (Figure 6). There are no apparent changes between scenario 1 and 2 but, as the reduction becomes more relevant in the other scenarios, the increase in oxygen levels becomes more significant. Hence, the largest changes are found when comparing the reference scenario with the fictitious scenario.

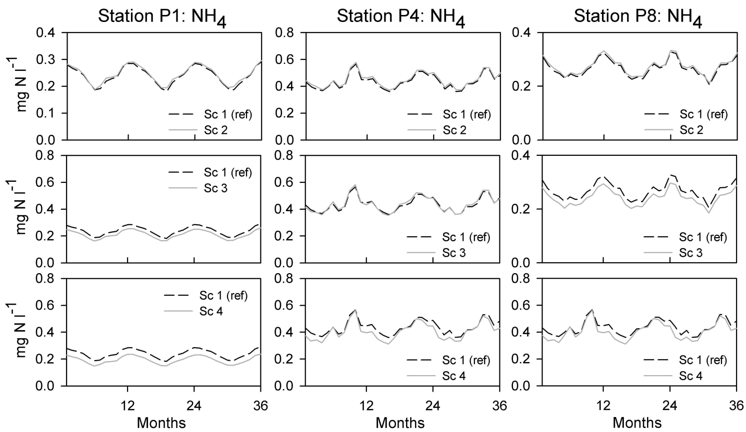

FIGURE 6: *Model results for dissolved oxygen at Stations P1, P4 and P8. Last three years of the 10-year simulation run.*

5 DEVELOPMENT SCENARIOS AND THEIR IMPLICATIONS

Results of future scenario analysis have suggested, as expected, that the treatment of domestic sewage effluents in the basin for the 2015 and fictitious scenarios reflected significant improvements in water quality, except for some points where critical zones remain due to the permanence of effluent discharges from the housing nuclei not included in these projects. The current and 2010 scenario analysis showed how the disorderly occupation of territorial space along with insufficient proper sanitary infrastructure can pose great threats for the riverside communities and other populations that use these waters for recreation, leisure and food. *E. coli* concentrations were some orders of magnitude above the level allowed for water bodies of Class 1, according to Resolution 357/05 from CONAMA (2005) which sets a mandatory maximum limit of 10^3 MPN/100 ml for brackish water. These values were estimated by the model under different tidal conditions and also obtained in laboratory analysis. Higher values imply human health risks in use of these waters by the population.

The estuary is subjected to intense anthropogenic impact from urban activities, manifested mainly in high loads of organic matter and nutrients discharged in the system. These pressures not only affect the (eu)trophic state of some parts of the estuary, but may also contribute to alter the ecologic dynamics of the system as a whole. The increase of both organic matter and nutrients in the system above background levels, as a result of eutrophication, poses serious threats. Oxygen depletion is one of the most serious threats that coastal systems such as estuaries can face (NRC 2000). If the enrichment occurs via the addition of organic material, then an increase in bacterial activity may be expected, leading to potential oxygen depletion, both in the water column and in the sediment. So, while not significantly changing the nutrient concentrations in the estuary, the decrease in nutrient loads from human occupation contributes to the increase in the oxygen concentration, as seen in the scenarios that have been tested. This is particularly relevant in a system like Santos estuary where the physical and biological characteristics can lead to anoxic conditions in the inner areas and in places where the water column may become stratified, or to less extreme conditions such as oxygen sags in places where residence time is long.

6 FINAL CONSIDERATIONS AND FUTURE WORK

Models are increasingly becoming indispensable tools in environmental studies and management decisions. This work shows the relevance of water quality modelling in the important and difficult task of integrated analysis of components responsible for the alteration of state in a dynamic environment under strong anthropogenic influence, such as densely populated estuaries. This study shows that even mild improvement of sanitary conditions (e.g. scenario 2) can already lead to situations where people, especially children, have free access to the water and may use it safely as a playground. From an ecological perspective, the results suggest that inappropriate management practices which result in an increase in nutrient inputs to the system, especially in the form of organic matter, inevitably contribute to its degradation.

This work represents the first step in the development of a water quality forecast system for the region, which can be used to improve the quality of future decision-making. Also, this study points to the benefit for the human and system's health if proper management practices are implemented with regard to the nutrient loads that reach the system in sewage. The results will provide relevant information to stakeholders for an effective management of issues related to the region's environmental sanitation, and to public health problems and further ecosystem degradation that might occur if appropriate management practices are not adopted.

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POTENTIAL USE OF ECOLOGICAL TOOLS TO DIRECT PUBLIC POLICIES: AN INTEGRATIVE APPROACH IN THE SANTOS ESTUARINE SYSTEM

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1 WHY AN INTEGRATIVE APPROACH?

The integration of environmental data (physical, chemical, biological, ecotoxicological and ecological descriptors) can be performed through multivariate analyses, resulting in a wider and more robust interpretation of the data set. Principal Components Analysis (PCA) is one of the most common methods to integrate data from different nature and it has been successfully used in environmental quality assessments (Cesar et al. 2007, Riba et al. 2004, Del Valls et al. 2002, Del Valls and Chapman 1998).

Physical, chemical, biological and ecotoxicological data obtained in surveys conducted during August 2005 and March 2006 in the Santos estuarine system were used to assess the water and sediment quality (see chapter on Santos Estuary, Part B). Such approaches, when applied alone, may result in a lack of realism and/or large uncertainties; but when they are used in an integrative manner, more reliable information about the environmental condition is provided. The complex nature of the water and sediment data matrices makes it difficult to identify the components that cause biological effects and their correlations. In this context, the PCA (Factor analysis) is an interesting tool due to the fact that: (1) it provides an ecological interpretation of the environmental properties; (2) it integrates different lines of evidence; (3) it supports conclusions based on a weight-of-evidence approach.

The objective of this chapter is to establish water and sediment quality in different areas affected by different sources of contamination by using an integrative method, providing a better understanding of the real impact of human activities on this estuarine environment.

2 THE MULTIVARIATE ANALYSIS

Physical, chemical, biological and ecotoxicological data were integrated by factor analysis using a principal components analysis as the extraction procedure. This is a multivariate technique to explore variable distributions. The original data set used in the analysis was divided in two main matrices: one referring to water and another to sediment data. After that, each main matrix was divided in two, based on the sampling period. They were named: Water Column (winter), Water Column (summer), Sediment (winter) and Sediment (summer).

Water matrices included the results of toxicological analysis evaluating sea-urchin embryolarval development (SUED), microbiological parameters (total coliforms (TC) and *Escherichia coli* (EC)), local depth (LD), euphotic zone depth (Zeu), chlorophyll-a concentration integrated over the water column (Cl int), salinity (SAL), dissolved oxygen saturation (DOS), concentration of ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-) total inorganic nitrogen (N total), inorganic phosphate (PO_4^{-3}), silicate ($Si(OH)_4$), Chlorophyll-a concentration (Chl-a), Chlorophyll-c concentration (Chl-c), Carotenes (Carot.), percentage of active Chlorophyll -a in relation to the sum of Chlorophyll-a plus phaeopigments (% Chl-a act), and total suspended matter or seston (Se).

Sediment matrices included the results of sea-urchin embryolarval development in interstitial water (SUEDI), sea-urchin embryolarval development in elutriate (SUEDE), sea-urchin embryolarval development in sediment-water interface (SUEDSW), *Nitocra* sp. offspring rates in interstitial water (ORIN), *Nitocra* sp. offspring rates in sediment (ORSN), *Tiburonella viscana* survival in sediment (SSA), analyses of sediment composition as percentage of fine particles (%F), organic matter (OM), salinity (SAL), and several ecological indexes such as number of species (S), number of individuals (N), Margalef´s species richness (d), Pielou´s evenness (J'), Shannon's diversity (H'), and Simpson's dominance (D).

Factor analysis was performed on the correlation matrix, which means that the variables were auto scaled (standardized) so as to be treated with equal importance. All the analyses were performed using the PCA option for the multivariate exploratory techniques procedure, followed by the basic set-up for factor analysis procedure from the STATISTICA software tool (Stat Soft Inc 2001 version 6). The eigenvalues obtained that were higher than 1 (Kaiser's criteria) and the factor loadings higher than ± 0.40 were considered as significant, according to Tabachinic and Fidell (1996). Besides the analysis of the variables aggregated by PCA, a representation of factor scores from each sampling point to the centroid of all cases for the original data was made; the factor scores quantify the prevalence of every component for the sampling points and are used to confirm the factor description.

3 RESULTS AND DISCUSSION

3.1 Water column (winter)

Results of the factor analysis applied to the Water Column (winter) matrix are summarized in Table 1. The three principal factors explained 82.04% of the original data set bulk variance. The loadings following varimax rotation for the three factors are presented. Each factor is described according to the dominant group of variables. The first principal factor (F1) accounts for the majority of the variance (44.43%), and it was positively associated to high concentrations of total coliforms and *E. coli*, as well as to nutrients as ammonium, nitrite, total inorganic nitrogen, phosphate and silicate. Negatively related to F1 are the embryolarval development of sea-urchins, salinity and oxygen saturation. Such association of variables indicates that F1 represents nutrient rich estuarine water undergoing nitrification process and high bacterial activity decreasing the oxygen levels in the water column, due to the predominance of heterotrophic process over the autotrophic ones. Factor 2 (24.03%) is related to phytoplankton biomass (integrated chlorophyll-a in water column and average values of chlorophyll-a, c and carotenes) and to availability of light in the water column, indicated by positive correlation to euphotic zone and negatively related to total seston, which means low water turbidity. Factor 2 can be ascribed as favorable conditions to phytoplankton development, once light is the primary limiting factor for phytoplankton growth in estuarine waters (Ancona 2007). Factor 3 explained 13.58% of the variance related to nitrate, total nitrogen availability and high proportions of active chlorophyll-a, indicating a new production process or a less intense grazing pressure.

Besides the analysis of the variables aggregated by PCA, a representation of estimated factor scores from each station to the centroid of all cases of the original data was made in order to confirm the factor descriptions and to characterize the water quality at each sampling station (Figure 1). Factor 1 was very important only at point 6, at Largo da Pompeba in São Vicente Channel, which means this site presented unique characteristics related to the other sampling points, namely, the worst environmental conditions. Foremost, the low oxygen availability, and the bad development of the sea-urchins embryos in the ecotoxicological tests attest the heavy contamination of the inner estuary. The winter represents the less rainy season in the area, which results in small loads of sediment transported by runoff, as indicated by the negative association to seston. The significant contribution of domestic sewage at this site is pointed out by the high concentrations of total coliforms and E. coli. The sampling points located in Santos Channel (P1 to P3) do not correlate to any factor (Figure 1). This feature is due to the lower nutrients and phytoplankton biomass (i.e. chlorophylls) concentrations found in this region compared to the ones observed at the inner estuary (P4 and P5) and in São Vicente Channel (P6 to P8). Indeed, P4 and P5 presented positive factor scores to F3 suggesting new biomass production (high percentage of active Chl-a) supported by nitrate.

Factor 2, representative in points 5, 7 and 8, indicates eutrophic conditions in terms of phytoplankton biomass. According to the data presented in Ecological status of the Santos Estuary water column chapter (Gianesella et al., this volume) nutrient availability is also high at these sites. The present analysis indicates point 6 as the place under heaviest sewage contamination. Nevertheless, this effect is diminished towards inner estuary and towards the São Vicente Channel downstream. Moreover, São Vicente Channel presents more eutrophic characteristics in terms of phytoplankton biomass and nutrients than Santos Channel.

3.2 Water column (summer)

The PCA analysis applied to the Water Column (summer) matrix distinguished four principal factors. The loadings following varimax rotation for the four factors are presented in Table 2. The four factors explained 87.36% of the variance in the original data set. The first principal factor (F1) accounted for 41.04 % of the variance. This factor was positively related to the high levels of dissolved CO2 and inorganic nutrient concentrations (ammonium, nitrate, nitrite, phosphate and silicate). Negatively related to the factor thereof are salinity, the percentage of oxygen saturation and seston concentration. Similar to the Water Column (winter) data, the F1 represents the estuarine waters rich in nutrients under higher heterotrophic activity. The

F2 (20.01% of the variance) was associated to photosynthetic pigments (chlorophylls a, c and carotenes) and inversely related to phosphate and seston. Again, as in the winter matrix, F2 represented phytoplankton biomass abundance. The third factor explained 16.26% of the variance and associated local depth, euphotic zone thickness (Zeu) and high percentage of active chlorophyll-a. Inversely correlated to F3 were chlorophyll-a by area unit (Int. Chl.) and seston. Therefore, F3 represents the light availability at the sampling points. Factor 4 (10.05%) was related to positive sea-urchin embryolarval development, high values of total coliforms and E. coli, besides reduced nitrogen forms (ammonium and nitrite). This factor indicates the relation between the presence of coliforms and high ammonium levels in the water due to the nitrification processes.

Considering the factor scores at each sampling point along the Santos Estuarine System (Figure 2) the relation of factor 1 with the inner estuarine sites is evidenced. F1 became more representative from P3 to P5 and decreased from this point to P7. This factor separated the regions under more intense salt water influence from the ones dominated by brackish water. The F2 was representative in the sampling points P1, P3 to P5 and P8. This factor basically represented the sites with high phytoplankton biomass: the channels inlets (P1 and P8) and the inner channel stations with low seston concentrations allowing the phytoplankton growth. Factor three (F3), representative in Santos Channel stations (P1 to P3), P4 and P6, was related to the availability of light in the deepest layers of the water column. In these points light could reach the bottom layer, in opposition to the points 7 and 8, for instance, where the euphotic zone occupied less than a half of the total depth. This is an important factor for phytoplankton growth, once cells can be light limited during a great part of the day through this course along the mixing layer, and this is what was probably happening in São Vicente downstream.

Finally, factor 4, related to high coliforms and ammonium levels, indicative of domestic sewage contamination, was more representative in point 6, as observed in winter data. However, the weight of such variables in summer were smaller than those observed in winter, probably due to the more intense rainfall in summer, which enhances the dilution and dispersion of contaminants, decreasing the impact observed in winter scenario. In summer, the highest rates of bacterial secondary productivity and primary productivity were detected at this place, conferring to P6 a unique characteristic in relation to the other portions of the estuary. Factor 4 was also representative in point 1, in the Santos Channel inlet. This result can be a consequence of clandestine domestic sewage disposals from Vicente de Carvalho, a neighboring area which is growing faster and disorderly.

3.3 Sediment (winter)

Physical, chemical, ecological and ecotoxicological data of sediment samples collected in winter were associated by PCA resulting in three principal factors (Table 3). The loadings following varimax rotation for the three factors are found in Figure 3. Such factors explained

76.5% of the variance in the original data set. The first principal factor (F1) was predominant and accounted for 43.67 % of the variance. This factor associated only the ecological data S, d, J', H' and D. This factor exhibited the correlation of ecological descriptors without association with biological adverse effects assessed by the toxicity assays. Regarding the Scores, this factor was significant to the sampling points 2 and 6, where the highest ecological descriptors were revealed in winter.

The second factor (F2) accounted for 18.71% of the variance and associated positively amphipod survival in sediment, percentage of fines and organic matter. This factor showed the correlation between toxicity to embryolarval sea-urchin development and *Nitocra* offspring rate with a reduced number of individuals, indicating that inner areas of the estuary where the percentage of fines and organic matter is higher may present some kind of contaminants, which could explain the toxicity and reduced number of individuals found in the benthonic macrofauna. The Factor Scores related with Factor 2 were significant to the sampling points 3, 4, 5 and 6. Those areas are influenced by river drainage and effluents from the industrial complex (Points 3, 4 and 5), besides receiving untreated domestic sewage from irregular housing (Points 4, 5 and 6).

The third factor (F3) accounted for the lowest variance (14.12%) and associated *Nitocra* sp offspring rates in interstitial water and Salinity inversely to percentage of fines and organic matter. This factor presented the influence of environmental characteristics on biological responses obtained from toxicity tests, corroborating Factor 2 regarding *Nitocra* offspring rates and sedimentologic parameters. Related to this Factor were the sampling points 1, 2, 3, 4 and 8, where environmental characteristics as grain size, organic matter and salinity are related to *Nitocra* offspring.

3.4 Sediment (summer)

Physical, chemical, biological, ecological and ecotoxicological data of sediment samples collected in summer were associated by PCA resulting in three principal factors (Table 4). The loadings following varimax rotation for the three factors are found in Figure 4. Such factors explained 82.95% of the variance in the original data set. The first principal factor (F1) was predominant and accounted for 39.43% of the variance. This factor associates sea-urchin embryolarval development in sediment-water interface, sea-urchin embryolarval development in elutriate, *Nitocra* sp. offspring rates in sediment, organic matter and Pielou´s Evenness (J') inversely to amphipod survival in sediment, number total of individuals and Simpson's dominance. This factor exhibited the correlation of ecological descriptors (Pielou´s Evenness and Simpson's Dominance) with biological responses assessed by the sea urchin embryolarval development and the *Nitocra* sp offspring rate. The survival of amphipods could be affected by the increase in organic matter and its association with contaminants, which were available only in sediment matrix. Regarding the Scores, this factor was significant to the sampling points 2, 3 and 5.

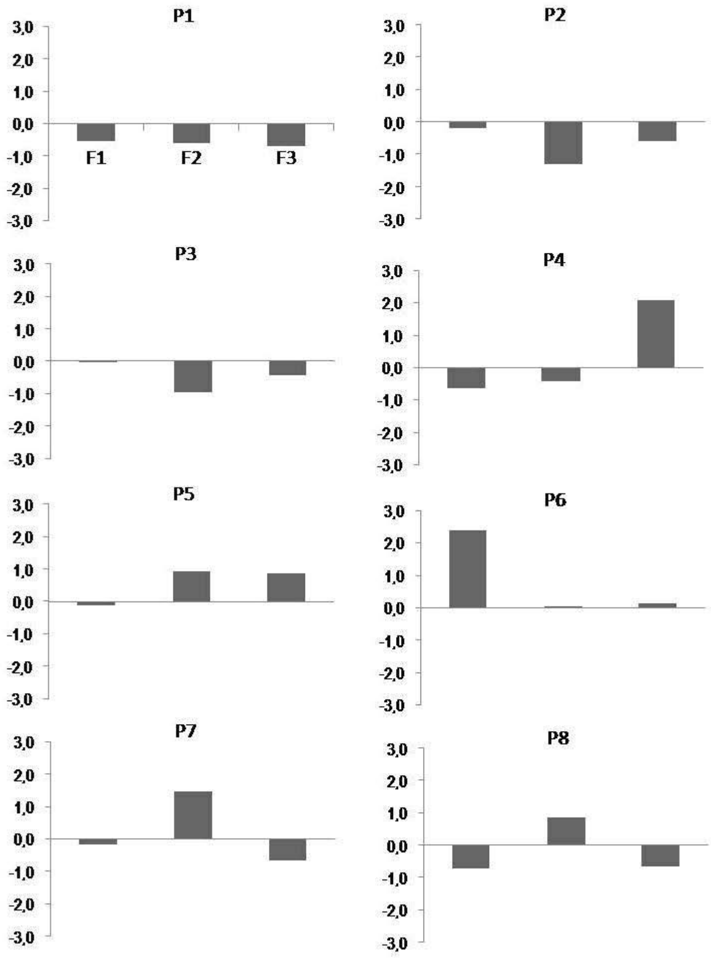

FIGURE 1: *Estimated factor scores (F1 to F3) to the eight sampling points along the Santos Estuarine System considering data of the winter water column matrix. The factor scores quantify the prevalence of every component for each station and are used to confirm the factor description.*

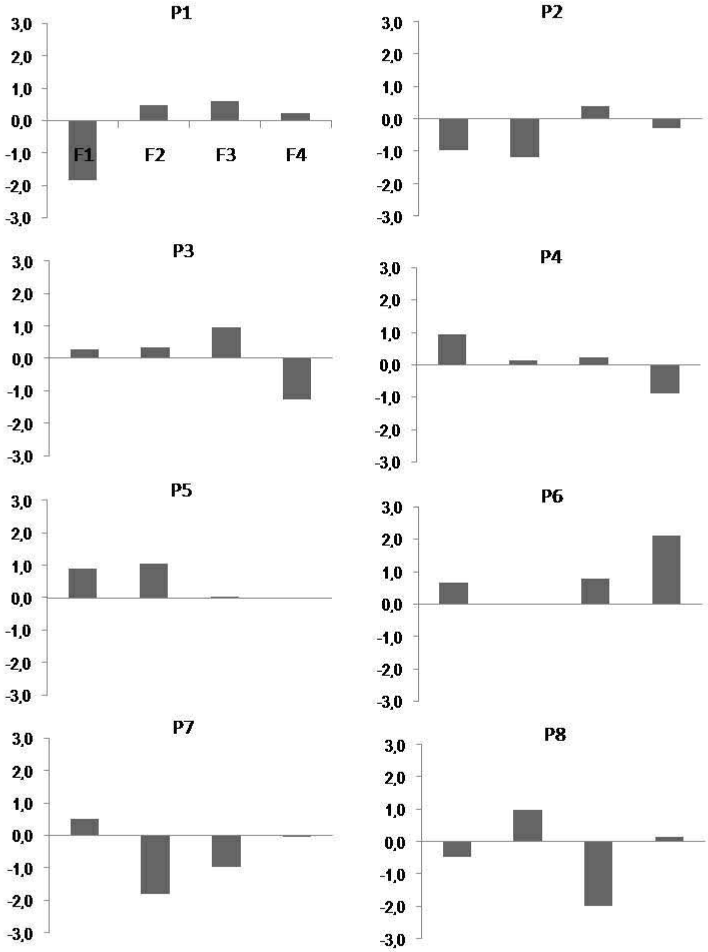

FIGURE 2: *Estimated factor scores (F1 to F4) for the eight sampling points along the Santos Estuarine System considering data of the summer water column matrix. The factor scores quantify the prevalence of every component for each station and are used to confirm the factor description.*

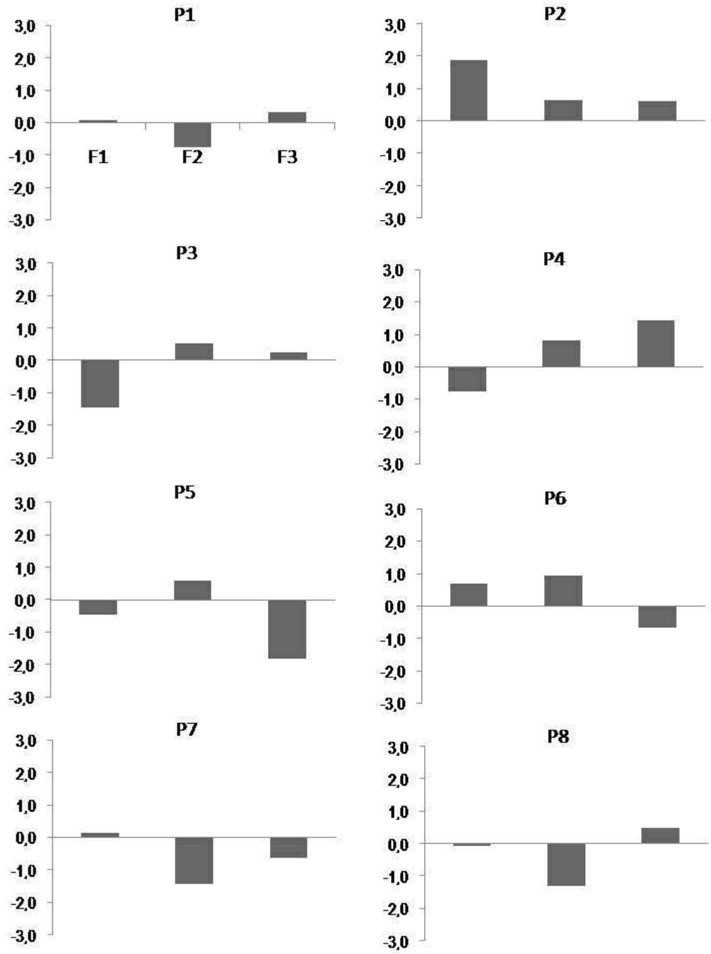

FIGURE 3: *Factor scores (F1 to F3) from eight sample points (P1 to P8 referred to sites 1 to 8 from Santos ECOMANAGE data field campaigns) considered for the winter sediment.*

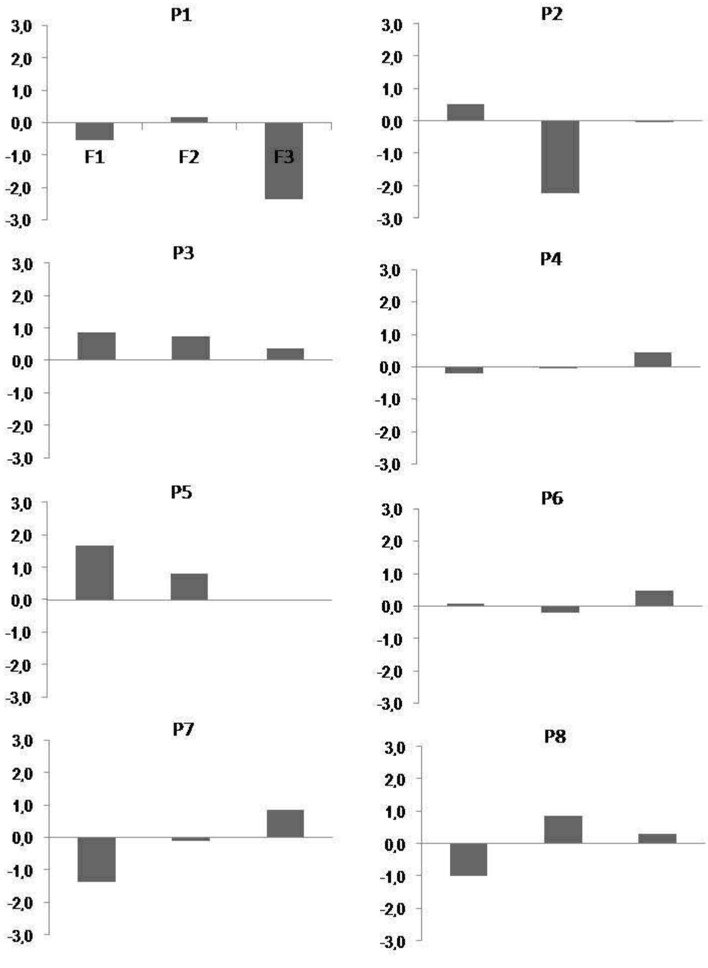

FIGURE 4: *Factor scores (F1 to F3) from eight sample points (P1 to P8 referred to sites 1 to 8 from Santos ECOMANAGE data field campaigns) considered for the summer sediment.*

TABLE 1: *Sorted rotated factor loadings of the original 19 variables on the three principal factors of winter water column data from Santos Estuarine System.*

Parameter	Factor 1	Factor 2	Factor 3
% Variance	44.43	24.03	13.58
SUED	-0.94	---	---
TC	0.89	---	---
EC	0.92	---	---
Depth	---	-0.79	---
Zeu	---	0.53	---
Int. Chl	---	0.76	---
Salinity	-0.94	---	---
% DOS	-0.82	---	---
NH_4^+	0.96	---	---
NO_3^-	---	---	0.92
NO_2^-	0.76	0.42	---
N Total	0.84	---	0.49
PO_4^{-3}	0.43	---	0.62
$Si(OH)_4$	0.77	0.41	---
Chl-*a*	---	0.88	---
Chl-*c*	---	0.95	---
Carotene	---	0.90	---
% Active Chl	---	---	0.89
Total seston	---	-0.57	-0.60

TABLE 2: *Sorted rotated factor loadings of the original 19 variables on the three principal factors of summer water column data from Santos Estuarine System.*

Parameter	Factor 1	Factor 2	Factor 3	Factor 4
% Variance	41.04	20.01	16.26	10.05
SUED	---	---	---	0.61
Total Colimetry	---	---	---	0.85
E. coli	---	---	---	0.86
Depth	---	---	0.96	---
Zeu	---	---	0.96	---
Int. Chl	---	0.48	-0.77	---
Salinity	-0.93	---	---	---
% DOS	-0.94	---	---	---
% CO_2	0.74	---	---	---
NH_4^+	0.76	---	---	0.62
NO_3^-	0.77	---	---	---
NO_2^-	0.57	---	---	0.65
N Total	0.82	---	---	0.54
PO_4^{-3}	0.57	-0.41	---	---
$Si(OH)_4$	0.81	---	---	---
Chl-*a*	---	0.95	---	---
Chl-*c*	---	0.92	---	---
Carotene	---	0.97	---	---
% Active Chl	---	---	0.96	---
Total seston	-0.60	-0.51	-0.57	---

TABLE 3: *Sorted rotated factor loadings of the original 15 variables on the three principal factors of winter sediment data from Santos Estuarine System.*

Parameter	Factor 1	Factor 2	Factor 3
% Variance	43.67	18.71	14.12
SUEDI	---	---	---
SUEDE	---	-0.57	-0.58
SUEDSW	---	0.74	---
ORIN	---	---	0.94
ORSN	---	-0.95	---
SSA	---	---	---
% F	---	0.62	-0.58
OM	---	0.57	-0.78
Salinity	---	---	0.59
S	0.91	---	---
N	---	-0.86	---
d	0.93	---	---
J'	0.87	---	---
H'	0.97	---	---
D	-0.95	---	---

TABLE 4: *Sorted rotated factor loadings of the original 15 variables on the three principal factors of summer sediment data from Santos Estuarine System.*

Parameter	Factor 1	Factor 2	Factor 3
% Variance	39.43	29.37	14.15
SUEDSW	0.74	---	---
SUEDE	0.68	---	0.45
SUEDI	---	---	-0.96
ORIN	---	---	-0.93
ORSN	0.69	---	---
SSA	-0.57	-0.52	---
OM	0.93	---	---
S	---	-0.91	---
N	-0.76	---	---
d	---	-0.99	---
J'	0.92	---	---
H'	---	-0.96	---
D	-0.54	0.72	---

The second factor (F2) accounted for 29.37% of the variance and associated Simpson's Dominance inversely to amphipod survival in sediment, number of species, Margalef's species Richness and Shannon's Diversity. This factor exhibited the direct association of ecological descriptors with effects on amphipod survival, probably related with contamination and/or sedimentological parameters. Significantly related to this Factor were the sampling points 1, 3, 5 and 8. Finally, the third factor (F3) accounted for the lowest variance (14.15%) and associated sea-urchin embryolarval development in interstitial water, sea-urchin embryolarval development in elutriate and *Nitocra* sp. offspring rates in interstitial water. This factor exhibited association of toxicity responses in interstitial water. Significantly related to this Factor were the sampling points 3, 4, 6, 7 and 8.

4 CONCLUSIONS

This integrative analysis applied to the whole estuarine area clearly indicates that the inner portion is more intensively affected by the anthropogenic impacts at the present time. The worst water conditions were found in Largo da Pompeba (P6), especially during the winter, when rainfall is less intense and the dilution or renewal rates of the estuarine water are smaller. Another important aspect to be taken into account is the small water volume in São Vicente Channel due to its shallowness, preventing a quick and efficient dispersion of the dissolved and particulate material which enters in the water column, opposed to the observed at Santos Channel. Despite the high ammonium levels which were detected in most parts of the estuarine system, the integrative analysis of nutrients, toxicity tests and colimetry data was able to point out the more critical pollution sites, discriminating them from those where phytoplankton can develop in a healthy condition.

Regarding sediment conditions, it was possible to identify worse quality in Piaçaguera and Santos Channels, which exhibited more significant alterations probably related to the previous intensive industrial discharges that occurred in this area, suggesting that the persistence of pollutants in this environmental compartment are still producing toxicity and consequently negative effects to the benthic biota. Apparently, the input of contaminants in the Santos Estuarine System water bodies from industrial discharges is relatively under a better control than the untreated domestic sewage discharges, which affect mostly the São Vicente Channel. These results demonstrate how good the decision made by the researchers was to focus on the urban occupation and sewage disposal, among so many others, as the pressures for the DSS study model. The present results strongly indicate the collection and treatment of domestic sewage, especially in the São Vicente Channel, as priorities for the improvement of the estuarine environmental condition.

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BUILDING OF THE DECISION SUPPORT SYSTEM IN THE SANTOS ESTUARINE SYSTEM

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1 INTRODUCTION

This chapter is intended to present the work done by the ECOMANAGE project in applying the information gathered during the project to the pressing environmental problems which are threatening the Santos Estuarine System and are therefore urgently in need of effective management responses. As described before, this region is marked by several conflicts whose associated complexity made it inevitable to make a choice for a problem that could be tackled within the project's time frame. Thus, after carefully reviewing the magnitude of the impacts that these existing conflicts were having on both human health and water quality, as well as the associated political will to discuss each problem, two issues where selected: the housing and domestic sewage scenarios. In the next sections both problems will be briefly illustrated, followed by a description of the work developed to implement the Decision Support System, which was built together with the local stakeholders in order to draw up possible responses for these problems.

2 SCENARIOS ADDRESSED IN THE DSS DEVELOPED FOR THE SANTOS ESTUARINE SYSTEM

The choice of the scenarios that were going to be analyzed for application in the DSS was based on the difficulties exposed by stakeholders both from the local and state public administration, which were mainly related to issues regarding the urban development that has occurred in this region, as well as on the state analysis of the estuarine system made by the ECOMANAGE project. From these, three main issues were identified: (1) the lack of available spaces near urban centers where basic infrastructure was already established; (2) the high costs of projects related to housing demand and basic sanitation services; and (3) the amount and resulting impacts of diffuse pollution originated from both urban drainages of the neighborhoods without a sewerage drainage network and sub-normal housing nuclei. Consequently, two main problems were selected: the housing demand for the resettlement of populations in under-developed housing and the respective supply of basic sanitation services.

Most of the populations that live in under-developed housing are occupying ecologically sensitive or risky areas such as riverbanks, mangrove areas and hill slopes, which are classified as permanent preservation areas by the Brazilian legislation and therefore, cannot be occupied. This situation is a result of the disorderly urban expansion that occurred in the past, and whose consequences last up to these days and has led to a high demand for appropriate space for resettlement. Since this pitfall of finding urban spaces adequate to this demand has actually been aggravated over the last decades, the issues associated with the urbanization of sub-normal human settlements were chosen as the first scenario, called Housing scenario for the application of the multiple tools developed within the project, namely the DSS.

The second scenario evaluated in the project equally derives from the form of urban development that has established itself in the region and is related to the supply of basic sanitation services and its features. Because this urban expansion was not adequately planned or guided, neither was the supply of these services, areas with serious insufficiencies of basic sanitation were originated, where the sewage ends up being dumped directly into the estuarine waters, without any kind of treatment. Thus, the second scenario selected by the ECOMANAGE project was related to the aspects involved around the dumping of residual waters into the estuary and was called Sewage scenario.

2.1 Main Aspects of the Assessed Scenarios

According to PRIMAHD (2005), IBGE (2000) and more recent municipal surveys, about 200,000 people live in irregular housing in the Santos - Sao Vicente Estuary drainage basin that have insufficient or no basic urban infrastructure such as water supply, electricity or garbage and sewage disposal. Municipal surveys further show that most of these irregular settlements correspond to population agglomerations that surpass 1,000 inhabitants. This is actually the case for 60% of the sub-normal occupations located in Santos and of all occupations in Sao Vicente (Young 2006).

The lack of adequate areas for housing is one of the main problems in the region and results from three factors. The first is a direct consequence of the natural characteristics of the coastal plain where the study area is located, since the urban center of the region is situated in the insular portion of the estuary where there is limited space for any kind of construction. This part of the estuary is in fact already undergoing under a major process of population densification. The second aspect derives from the fact that the peripheral urban areas, though less dense, also have their expansion limited for they are surrounded by Atlantic rainforest and mangrove vegetation whose ecosystem services, described in previous chapters, are fundamental to the overall maintenance of the region and its quality of life and therefore should not be disturbed. Also, the further destruction of remaining vegetation areas that are in intermediate and late succession stages was recently prohibited by means of a state resolution (SMA 2007). This is all the more valid since the areas that have had their vegetation altered but have no longer been disturbed ever since can be found nowadays in an advanced state of regeneration. And last, the third aspect is related to the existence of a great number of areas with contaminated soils in the region (CETESB 2001) that compromise their use by society and are still waiting for an appropriate solution.

One of the attempted solutions to this problem has been the acquisition by the government of areas that are prone to receive populations from sub-normal housing. In this sense, the use of the available spaces in already urbanized areas or those adjacent to them for such purposes would be ideal, since they already have the necessary urban basic infrastructure and this would allow for easier integration with the urban centers. Nevertheless, the lack of space in the region has resulted in an increase in land value and has made this solution too costly and many times not feasible. This phenomenon of rise of urban land values, already detailed in the chapter related to socio-economy of the Santos Estuarine Region has been intensified over the last decade by the increasing interest of private companies for these areas. This process has been mainly powered by the resumption of the real estate sector and growth of the port sector, whose expansion in activities has elevated the demand for useful areas, valuing not only the scarce urban lots found in upmarket areas but also peripheral urban areas, especially those in good environmental state. This is particularly the case for retroportuary areas because the elevation of container cargo trading that has been taking place in the Port of Santos since 2000 has dramatically increased the demand for low-cost areas usually located in the outskirts of the urban area and consequently speculated their price.

This already complex scenario made up by the increasing demand for retroportuary areas, the strong activity in the real estate market and the high degree of urban densification will probably be worsened by industrial expansion, as revealed in some municipal urban occupation plans from 2007 where incentives to this end can be found. This setting has therefore led to a situation of fierce competition among the economic sectors to the detriment of the underprivileged, whose basic necessities have been left aside. This is all the more true to those communities situated in fragile and environmentally risky areas that have no conditions of hygiene and safety and should therefore be transferred to other areas that have adequate urban infrastructure.

To cope with this situation, another solution that has been tried in the region to solve the problem of sub-housing has been to maximize the re-urbanization of the areas already degraded by these settlements, only providing for the relocation of these populations when absolutely necessary and to the smallest possible percentage of people. However this solution also has serious hindrances, as pointed out by a World Bank report (IBAM 2004), which together with Brazilian institutions, analyzed several Brazilian experiences in land urbanization and regularization of human irregular settlements. These include financial constraints, related to securing, managing and applying the necessary financial resources; technical constraints, related to the geomorphologic and environmental characteristics of the area, as well as those related to the kind of occupation that is in place; and constraints related to the lack of judicial and institutional capacity and political willingness. This work has also pointed out that the absence of effective control over further growth and expansion of settlements after re-urbanization is completed is another factor that threatens the continuity and sustainability of programs of this nature in the long term.

Some slum re-urbanization experiences were actually followed during the ECOMANAGE project and it was possible to witness the difficulty that exists in effectively carrying out projects of this nature. Among them, is the case of irregular settlements that existed in the insular portion of Sao Vicente and where re-urbanization of slum quarters that extended into mangrove area was in fact accomplished but due to the lack of adequate inspection and control mechanisms, allowed for the establishment of new sub-housing settlements, forcing the public administration to take extreme measures in order to limit its proliferation. This was done by building a road between the estuary channel and the area that had been re-urbanized and transferring the new sub-houses situated over the water, only leaving mangrove after the road in order to effectively avoid new occupations. Finally, vertical construction has also been attempted in order to maximize space but the pedological characteristics of the region make this kind of construction expensive, which again complicates its implementation for a popular housing end.

Chosen as the second scenario, the urban discharges were acknowledged by the stakeholders as also one of the main problems of the region, since a solution to the issue of sub-normal housing regularization has still not been found making the estuary the main direct receiver of this organic load. This situation is aggravated by the fact that these sub-normal houses are illegal, therefore preventing the public administration from providing the necessary infrastructure that would mitigate the impacts originated by this situation. This, of course, will only happen once these areas have been properly regularized by the State. There is actually a major sanitary project currently under way in the region, whose aim is to raise the actual sewage collection levels up to 95% of the total Baixada Santista population (Diário do Senado Federal 2004). However, this project does not consider the inhabitants that are occupying the illegal areas, which represent about 20% of the total population of the region. So even though this project will improve the quality of life and to a certain extent the environmental state of the estuary, it will still leave the most needed uncared for.

The lack of appropriate responses and the worrisome status of the Housing and Sewage situation in the Santos Estuarine System are consequently issues that were considered to need urgent attention and were therefore prioritized in the ECOMANAGE project.

3 METHODOLOGY APPLIED IN THE DSS AND MAIN RESULTS FOR THE HOUSING AND SEWAGE SCENARIOS

3.1 Assembling a stakeholder group

The Decision Support System (DSS) was the main tool developed by the ECOMANAGE project to assess the stakeholders view on the Housing and Sewage issues and jointly find the best answer to these problems. The first step then was to build a stakeholder group amongst the several institutions (governmental and non-governmental) and interested parties that were directly involved with these problems and willing to collaborate. Actually, a constituency was quickly built soon after the beginning of the project since there were over a hundred entities showing a strong interest to participate in the project, which allowed for the establishment of a consultative council. The stakeholders that belonged to this council represented the parties that wanted to collaborate with the ECOMANAGE project and so, they were also the ones that were invited to form a more technical group, whose objective was to participate in the development of the DSS. A quite diverse stakeholder group was therefore assembled with more then thirty representatives of state and local government agencies, academia and local NGOs, whose work dealt directly or indirectly with the Housing and Sewage drivers.

3.2 Filling in the decision matrixes

The next step was to set up a series of meetings with this group in order to build a decision-matrix for the DSS, which consists of an alternative/effects matrix, as explained in the methodology part of this book. In this way, the first meeting was intended to explain what a DSS consisted of and how it could help the stakeholders to evaluate different management alternatives for the selected issues, as well as to outline what was expected from their participation. Also, a brief consideration was made regarding the housing and sewage state of the region, mainly to present the findings that the ECOMANAGE project had assembled and to bring everyone to the same level of information. The Housing state was therefore considered in view of the actual distribution of slums and urbanized areas, as well as the projections of population growth for the next 10-15 years for the five municipalities. As for the sewage state, a diagnostic of critical areas, the sewage treatment level and the existing management measures were laid out.

In the second meeting, a matrix that had already some alternatives and effects filled in by the team was handled to the stakeholders as means to guide them, since none of them had ever participated in such work and debate was open to find more alternatives and effects for the Housing and Sewage problems. The alternatives and effects filled out by the team were done by considering all the information gathered by the project to evaluate the state of the region, as well as from experts' knowledge. This was done over several internal meetings that took place at UNISANTA and IO-USP and where some of the pressures and impacts of housing and urban discharges were defined, particularly regarding the biological aspects of such issues.

After noticing that participation had dramatically fallen from the first to the second meeting, fact that is explored later on in this chapter when the difficulties and opportunities of the implementation of the ECOMANAGE are assessed, a strategy change had to be made. Thus, after carefully considering time constraints and the difficulties that were occurring to gather and sustain such a diverse and numerous group to work together, the initial plan, which consisted of filling in the matrixes during the meetings so that consensus could be achieved by discussing issues together, was dropped and it was determined that the matrixes were to be completed by each stakeholder in their working environment and then sent over by email.

In the end, six alternatives for the Sewage scenario and four alternatives for the Housing scenario were jointly put forward by the team and the stakeholders, as well as several biological and socio-economical effects. The stakeholders also gave a score to each alternative/effect option, which was sent by email and the result was processed with the Mathedit® software in order to find the best alternative for each scenario. A final third meeting was then made to present the findings to the stakeholders and discuss the results. Again, this meeting had very little attendance when compared to the initial meeting, which corroborated the change of strategy that was made in the beginning of this process.

3.3 Finding the best alternative for each scenario

The results obtained for Housing showed that none of the proposed alternatives could be considered appropriate to deal with the actual situation since even the ones that got higher scores still remained quite far from the best (b) option, as it can be seen in Figure 1. Nevertheless, out of the four alternatives considered, the removal of slums into newly horizontally built areas was the best-placed alternative, closely followed by removal of slums into newly vertically built areas. This is actually consistent with the data gathered by the ECOMANAGE project, which shows that there are several unoccupied areas that are suited for relocation projects, once these are better planned.

As for Sewage, the best alternative was to build concrete cesspits followed by treated submarine outfall, as shown in Figure 2. Stakeholders therefore considered that concrete cesspits could be a good solution for the current sewage situation since the majority of the people that are not attended by municipal sewage services live in slum quarters. This general conception that concrete cesspits could attenuate the problem of having a deficient sewerage system exists amongst most stakeholders but is also in fact the general public opinion, who believes that this could be a solution in many other different areas in Brazil that are missing appropriate sanitary conditions. This is due to the fact that cesspits are a low technology and low cost option, which would make them ideal solutions when obtaining the necessary funding to install a fully operating sanitation system is a problem.

However, from a technical point of view, concrete cesspits are not considered a good solution in areas where the groundwater level is close to the surface, such as in the Santos Estuarine System, because this prohibits its construction. Furthermore, concrete cesspits only retain the solid part of the sewage. As a result of this, the contaminant elements (i.e. bacteria) are not removed from the effluent, meaning it will eventually reach the estuary and contaminate it, unless it is treated by chloride (ABNT-NBR7229/1993). Having put this, the best alternative should then be considered the expansion in number and capacity of the wastewater treatment plants and the construction of new submarine outfalls.

In conclusion, the DSS developed within the ECOMANAGE project showed great potentiality to back up environmental decision-making in the Santos Estuarine System. It must be acknowledged that the management alternatives considered for the Housing and Sewage scenarios were not very specific and so the results obtained were not surprising and were even expected. This was in part due to the lack of some crucial information that was necessary to further detail the alternatives and that could not be retrieved in time. Nevertheless, this was the first time that most of these stakeholders and even local project team members used a DSS to analyze environmental problems and consequently evaluated management alternatives depending on their cumulative social, economical and environmental effects.

Also, the DSS brought together stakeholders that have an important role in the management of the Santos region so that these alternatives could be jointly discussed and thought over. This was later recognized by these stakeholders to be quite gratifying and that it had helped them to consider other standpoints, showing them aspects that they were not aware of or that they did not realize that had any association with the tackled scenarios. In fact, one of the participating stakeholders, who represented the state environmental control and regulation agency, even showed its interest to apply the DSS on a particular issue related to the environmental and social impacts of a relocation project that they were handling at the time, recognizing in this way the potentiality of this tool. Consequently, the development of the DSS in the Santos Estuarine System, although it did not serve to help choosing a real management response to the problems of Housing and Sewage, it aroused awareness among the stakeholders involved that integrated decision-making by promoting an analysis of both natural and socioeconomic aspects of an issue, can help to build sound, sustainable environmental decisions and above all, can be a powerful tool to reach consensus in this highly conflicted area.

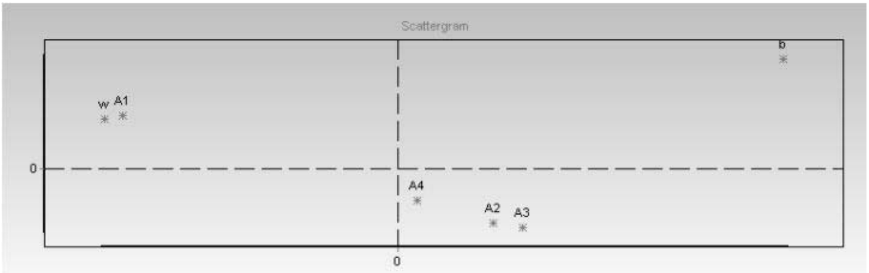

FIGURE 1: *Ranking results for Housing: w - worst alternative; b - best alternative; A1 - Slum expansion ("Business As Usual"); A2 - Removal of slums for vertical constructions; A3 - Removal of slums for horizontal constructions; A4 - Urbanization of former slum area (X and Y axes refer to relative spatial distribution of the alternatives).*

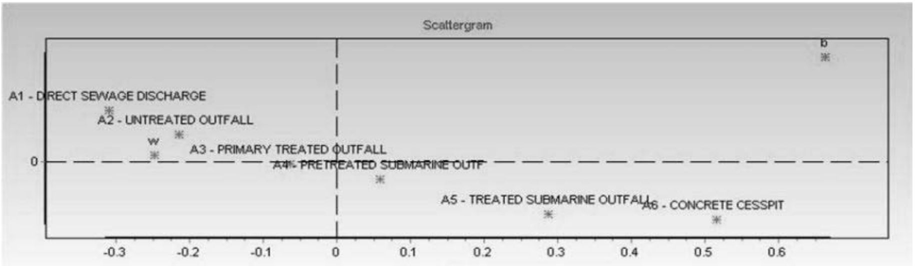

FIGURE 2: *Ranking results for Sewage alternatives: w - worst alternative; b - best alternative; A1 - Direct Sewage Discharge ("Business As Usual"); A2 - Untreated Outfall; A3 - Primary Treated Outfall; A4 - Pretreated Submarine Outfall; A5 - Treated Submarine Outfall; A6 - Concrete Cesspit (X and Y axes refer to relative spatial distribution of the alternatives).*

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DIFFICULTIES AND OPPORTUNITIES FOUND DURING THE IMPLEMENTATION OF THE ECOMANAGE PROJECT IN THE SANTOS ESTUARINE SYSTEM

C. BELCHIOR, S.M.F. GIANESELLA AND A.F.P. SAMPAIO

The purpose of this chapter is to present the main obstacles encountered to develop and implement a project such as ECOMANAGE in the Santos Estuarine System, but also to put forward some of the main opportunities that were revealed to the team during the project implementation. This outcome evaluation was preferred to a performance evaluation since this kind of evaluation is unusual in Brazil and by actually reporting on the existing difficulties and opportunities to implement an ICM project rather then just doing a simple output evaluation, it is believed that one can better contribute and improve the process of coastal management in the region or elsewhere, as discussed before. But foremost, it is necessary to briefly sketch the context of coastal management in Brazil with regard to what has been achieved up to now and what the existing constraints are to the implementation of the national programme.

The National Programme of Coastal Management (GERCO) in Brazil was first established in 1985 by the Interministerial Commission for Marine Resources (CIRM), with the first national coastal management law being published in 1988 (law n^{o} 7.661/88). This law established a National Coastal Management Plan (PNGC) aimed at proposing rational uses of coastal resources for the benefit of the local population and conservation of coastal ecosystems. It took a while to start implementing the PNGC´s instruments and working towards its specific objectives on promoting the sustainable development of Brazilian coastal zones (Asmus and Kitzmann 2004), although it represented a hallmark for ICM in the country.

Progress was made over the years, especially powered by a growing awareness of the national population of environmental issues, which added to the pressure imposed by international convention commitments and led to the development of legal and institutional conditions that allowed for ICM evolution and strengthening in the country. Thus, currently Brazil has a well developed coastal management programme, with specific coastal management plans developed in all its 17 coastal states, several technical and regulating tools set in motion and a central unit to coordinate all these efforts. Nevertheless, even though important advances have been made at the legal and administrative levels, there are still serious obstacles in the way of ICM in Brazil.

Several political, institutional, technical and sociocultural constraints have been identified (Diegues 1999, Asmus and Kitzmann 2004, Polette and Vieira 2005), with the main ones as follows: lack of political will to truly implement the National Coastal Management Programme, especially by local governments; decisions strongly influenced by economic interests, with existence of powerful lobbies from sectors such as the very state, tourism or the industry; lack of institutional integration, resulting in uncoordinated public policies that are related to coastal environmental issues; lack of a truly decentralized governance, leaving the crucial decisions in the hands of the federal government; significant differences between the states in the institutional and technical capacity to implement the National Programme, resulting in strong asymmetries in the country concerning achieved results; poorly capacitated and too few technicians in governmental coastal management teams, as well as inadequate funding; lack of involvement of specialists such as members from academia or consultants that have extensive experience in environmental, coastal and marine sciences; continuous changes in government institutions and positions, which ngatively affect the consistency and outcomes of coastal management actions; insufficient public participation in decision-making and last, but not least, the lack of effective social mobilization of local communities in support of coastal management measures.

Due to these constraints and although coastal management in Brazil is supported by a strong legal framework as well as other pertinent institutional arrangements, it has been considered that it remains a technocratic exercise, without so far having had a significant impact on the promotion of sustainable development of coastal zones (Diegues 1999, Polette and Vieira 2005).

Many of the above obstacles were also felt during the implementation of the ECOMANAGE project in the Santos Estuarine System, which was divided in two periods in order to successfully complete the different steps of the DPSIR model: a first one, which corresponded to the first two years of the project, where the focus was on collecting data and information to feed and validate the numerical models to be applied in the estuarine region but also to build a social and environmental diagnosis of the area, as a result of which the problem of Housing and Sewage in the region was identified as the core problem. In this phase, Drivers, Pressures and State were determined. The second and last period was marked by the development of the Decision Support System, which was carried out in order to build and evaluate management alternatives for both the Housing and Sewage problems, as previously described, determining in this way the possible Impacts and Responses for these selected problems that would result from the implementation of different public policy alternatives.

These two phases also corresponded to different degrees of stakeholder participation. Participatory management was a central tenet of the ECOMANAGE project so one of the main concerns was to constitute right from the beginning of the project a diverse group of stakeholders that would adequately represent all the interested parties in coastal management in the region. This was done by sending invitations to government agencies (local, regional and state) and representatives of local governments, private sector (industry, port, etc), as well as local NGOs to form a consultative council of the project.

During the first phase, these stakeholders were only asked to participate in consultative meetings where they were informed of the main goals, ongoing work and the next steps of the project. In these meetings, they were encouraged to express their opinions and make suggestions, without further commitment. In the second phase, a more active participation was requested and their views were incorporated into the Decision Support System that was built together with the team experts.

Since the work developed by the ECOMANAGE project was so diverse, several technical, institutional, political and sociocultural obstacles were also perceived. As stated before, these were similar to those reported by other authors along the whole country. In this way, the main technical obstacles were not so much in the lack of crucial information, but rather its unavailability or inconsistency. There exist many universities and other research institutions in the area, as well as in nearby Sao Paulo that have been studying various environmental and socioeconomic aspects of the region for quite a long time. Also, there already is a significant amount of official data, available to be used. The problem remained in the difficulty of actually obtaining that information, which is highly dispersed through several institutions and poorly disseminated or even of a classified nature. Due to their distinct institutional origin, the available data were also heterogeneous in respect to the methodology of obtainment and time-series frequently lacked continuity.

Some political and institutional obstacles were felt during the project development and since they pose a serious difficulty for the implementation of projects related to coastal and environmental management, they deserve to be outlined. These consist basically in the lack of political motivation to tackle environmental problems and the existence of important pressures on local government from lobby groups, mainly from industry, port and even some state sectors. Some municipalities took concrete action trying to solve some of the most serious and urgent socio-environmental conflicts in their jurisdiction with measures such as reallocating people that live in slum quarters or environmental rehabilitation but generally there is a lack of real commitment and interest in solving these conflicts, with economic development still being the main priority for local decision-makers.

Another serious institutional constraint found was the weak cooperation between institutions responsible for backing up decision-making or for its enforcement, compromising an effective dialog between the actors involved in local coastal management. It is interesting to note that there already exist some institutional arrangements in the region that should be promoting an integrated view of the area and a holistic course of action. These include an official sectorial group of coastal management, made up by governmental and societal stakeholders that was formed as a council to discuss and assist coastal management in the region; a hydrographical basin committee, also made up by governmental and societal parties, where issues related to the management of the watershed are discussed; a metropolitan agency, which was constituted to integrate the planning and organize the execution of governmental actions in the region; and a council for development of the metropolitan region, made up by representatives of the different municipalities and of the state government, where common issues to the region are discussed. Yet, these institutions or councils remain too often isolated from each other, when their work could and should be intertwined in order to combine efforts to build consensual measures and thus, improving decision-making.

The environmental offices of the local municipalities are also working separately, which results in actions that are only aimed at solving issues inside their own jurisdictional boundaries. Although some issues might originate only in one municipality, their direct or indirect impacts are almost never limited to that municipality, which makes these local measures ineffective to improve the state of the whole estuarine region. The municipalities also lack clear public environmental policies. Thus, local governments are not viewing problems from an ecosystem's perspective, their actions remaining strongly sectorial and the intergovernmental institutions or decision-aid councils that should be promoting or assisting this integrative vision are not fulfilling this role.

In respect to socio-cultural constraints, these were associated with the lack of willingness and true commitment from local stakeholders to participate in environmental decision-making. Although a formal participatory structure already exists in this region, with official councils and committees such as those mentioned above, where society is given a place to actively participate in the decision-making process, the main outcome of this participation has been quite disappointing so far, particularly to the social participants. Since their views and wishes are most of the time not taken into consideration, due to the tripartite (1/3 state government, 1/3 local government, 1/3 civil society) system of participation or due to the prevailing economics-driven course of action, this has led to a disbelief in this kind of management and therefore a lack of motivation to embrace participatory actions.

This was particularly felt during the second phase of the project, when a truly participatory approach was undertaken. In the first phase, the project had hoped to build the necessary constituency by gradually showing the potentialities of the developed tools for the environmental management of the Santos Estuarine System and also by demonstrating that this was a really democratic exercise, where all parties would be equally important and equally heard. However, when those stakeholders were invited to participate in the more technical meetings of the second phase, this time to get actively involved in the project, the attendance dramatically decreased over those meetings. Moreover, the social stakeholders were also those that participated least, when compared to governmental actors, which reveals their lack of enthusiasm for this kind of process. In fact, the determination to participate demonstrated by governmental actors, especially from agencies that weren't directly involved in decision-making (such as the agency responsible for the environmental regulation and the one responsible for the water and sewage planning) was somewhat surprising since, as mentioned before, governmental interests in the region are still strongly associated to economical issues and in this case, we were dealing with social and environmental problems. This fact may be attributed to a higher degree of environmental conscientiousness by the governmental technicians.

The ineffective participatory management currently taking place in the Santos estuarine region can also be explained by the fact that although there are several NGOs, whose activities are concerned with social welfare and environmental protection and awareness, as well as several projects of the same nature led by local universities, these actions remain disjointed. Civil society has still not succeeded in organizing itself to have a strong and unified voice, which would most certainly improve the effectiveness of their actions and their role in environmental management.

Although there were several obstacles that caused difficulties for the work developed by the ECOMANAGE team, many opportunities and potentialities were also revealed during the project period, which indicate that important steps have been taken to show the importance of integrated coastal management in finding sound and democratic responses to the local environmental problems and, therefore, suggesting that the work initiated by the ECOMANAGE project will be carried on independently, after its completion. First, although stakeholder participation was not as expected, important cooperation relationships could still be established with those that did participate, which may indicate that there are good opportunities to establish further relations of this nature. Furthermore, successful participation experiences, as small as they may seem, are an achievement that must be appropriately acknowledged in order to promote future actions in this course. This is particularly important in regions such as the Santos Estuarine System, where there is a background of poor social participation in environmental decision-making, which has created an image that participatory management is pointless in face of the power of economic interests.

In fact, the reasons behind the decrease in participation during the project development were not fully ascertained. It is a reality that there is a true difficulty in maintaining discussion with committed actors to debate the socio-environmental conflicts in the region and that the power of participation has fallen into disbelief but external factors might also have contributed to the lack of commitment. One suggested explanation is lack of agenda, that is, participants were already overworked and didn't have any more time or will to engage in volunteer activities. Another possibility could be a misunderstanding of what was being asked of the stakeholders, in this case representing a project team failure, since several authors emphasize the necessity of specialists in communication to make the interface between science and society (Beckers et al. 2007) and the project team had little experience as a whole on this subject. Therefore, the fact that they did succeed in forming a small but dedicated group of stakeholders that committed in such a way to the project and interacted actively with the team incorporating a proposed tool, must be considered the most favorable outcome of the ECOMANAGE project.

It was also clear to the stakeholders that this kind of forum where everyone could express their views, as well as listen to each other and work with integrated tools that allowed to structure problems from both their envronmental and socio-economical sides as well as from a watershed perspective (and not from the jurisdictional boundaries) was something innovative in the region and something that helped them to fully grasp the complexity of these issues and of the necessary responses. This positive feedback has in fact led to the continuation of partnerships between the Brazilian Institutions that comprised the local ECOMANAGE team and participating governmental institutions that have seized the potential and adequacy of the offered tools. In addition, another partnership has been formed at the end of the project with a metropolitan agency that did not participate during the project due to legal constraints, but after realizing the potentialities and applicability of the project's tools, has decided to overcome those constraints and show its willingness to work together. Therefore, these concrete partnerships that were generated indicate that the work initiated by the ECOMANAGE project has great possibilities to be continued, fulfilling the knowledge transfer objective of the project.

Since these relationships were formed between governmental institutions and universities, they might also indicate the possibility for other future cooperation partnerships between these entities in such a way that the scientific and technical capacity that exists in the many universities of the Santos region will be used to support and improve decision-making. This is in fact another achievement since the potential of academia to assist in coastal management in Brazil has been systematically underexploited. Finally, a data bank has been built over the Internet with free access that centralizes all the data and information gathered during the ECOMANAGE project, making these easily accessible to all interested parties. This will undoubtedly facilitate future projects and assist local governments and thus contribute to the development of further coastal management actions.

In conclusion, in spite of all difficulties encountered by the research team of the ECOMANAGE project in attaining its objectives, the efforts allowed major breakthroughs in the Santos estuarine region. It represented a relevant opportunity to consider complex issues such as Housing or Sewage that directly affect the well-being of local populations and the environmental integrity of the estuary, from a holistic point of view, gathering different and often conflicting stakeholders to discuss these matters in a democratic way and using integrated management tools to back-up decision-making. This pilot project resulted in the establishment of important partnerships between governmental institutions and universities, which have expressed their trust in the developed tools and their wish to carry on the work initiated, showing that in spite of the complexity of the existing social-environmental problems of the region, progress towards more sustainable forms of coastal management can be achieved if the appropriate course of action is taken.

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THE EFFECT OF FLOWRATE VARIATIONS OF THE SAUCE CHICO AND NAPOSTÁ GRANDE RIVERS ON THE INNER PART OF THE BAHÍA BLANCA ESTUARY

M.E. CARBONE, F. LIMBOZZI, E. ALBERDI AND P. ABALO

1 INTRODUCTION

Transition zones between fluvial and marine environments represent complex regions of high importance to man. The interface between fresh and sea water has a significant role in determining the dynamic of the estuaries and the sediment transport, as well as the boundary conditions for coastal river plumes and their chemical and biological properties. Hence the need to study the linkage between these two dynamical systems connecting the driving forces of the watershed activities to the coastal ecosystems and resources. To this purpose, in the conceptual index framework of DPSIR, the use of the flowrate discharge of rivers into the estuary (Leitão, this volume) was proposed as quantity state indicator for surface water. The river flowrate is the most important parameter to determine the possibilities of hydric resource management; it depends basically on three factors: the climate, the vegetation and the soil - substrate complex. In addition to these factors, some associated processes that directly influence the flowrate are: precipitation, evaporation, interception, transpiration, infiltration and storage (Pedraza Gilsanz 1996). Moreover, this indicator also reflects the extra demands (pressures) due to urban and agriculture activities.

This chapter presents an analysis of the hydrological behavior of Sauce Chico and Napostá Grande rivers in the lower part of their watersheds, particularly at their mouths, and the relationship between their flowrates and rainfall events. In addition, both their incidence on and their interaction with the hydrodynamics of the marine environment in the inner area of Bahía Blanca estuary are evaluated. It is shown how a significant flowrate increase produced by one storm event of a single exceptional rainfall affects this section of the estuary. To this purpose a preliminary hydrodynamic model based on the MOHID Water Modelling System has been developed.

2 STUDY AREA

The Bahía Blanca estuary is a mesotidal system conformed by an important network of channels of diverse dimensions oriented in a NW-SE direction. These channels have sinusoidal courses separated by islands and extensive tidal flats usually composed of silty clay sediments. The tidal regime is semidiurnal, the average tidal amplitude at the ports of Ingeniero White and Galván (inner zone) being 3.8 and 2.7 m in spring tide and neap tide respectively (Perillo and Piccolo 1999). Sauce Chico and Napostá Grande are part of the Atlantic basins and are the only permanent tributaries of the Bahía Blanca estuary (García and García 1964) which form a hierarchized drainage network of third order (Strahler 1952). The Sauce Chico

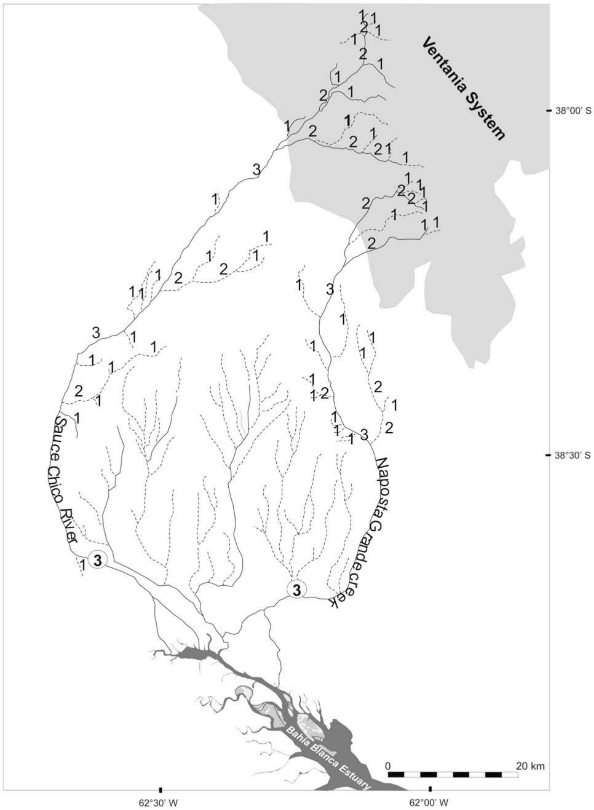

FIGURE 1: *Drainage network hierarchy.*

watershed has an area of 1,600 km^2 and the length of the river is 110 km. This course drains from the middle sector of the pedemont of the Sierra de la Ventana ranges, crosses the plain and then bends east-westwardly. Seven kilometers before arriving at the estuary, at the head of the Principal Channel, the river turns to the east and branches off demarcating its alluvial fan. According to its regional morphology, this tributary is the main collector of all watercourses which have drained and nowadays drain into the estuary (González 1997). Its hydrographic parameters were measured at the upper part of the watershed by the Dirección General de Agua y Energía which determined a module of 1.504 $m^3\ s^{-1}$ (1952-1978) and maximum and minimum flowrates of 570 and 0.310 $m^3\ s^{-1}$ respectively (Fig. 1).

With a length of 105 km and a watershed of over 1,237 km^2, the Napostá Grande river, drains into the central-western part of the Sierra de la Ventana ridge (Carrica 1998). This watercourse flows east to west and, after a short distance, converges southwardly. Its middle section traces a pronounced semicircular path, similar to that of the Sauce Chico River, until it turns abruptly in the direction of the estuary. In the urban area of Bahía Blanca, the

Napostá Grande has been canalized and runs through an underground pipe during part of its course. Ten kilometers before its mouth the stream bifurcates into the main course and the Maldonado channel. The morphology of the basin is characteristic of a plain environment with a smooth slope towards the south (Carrica 1998). Its drainage network is made up of permanent and temporary watercourses with low to moderate density. Its hydrographic parameters were measured by the Dirección General de Agua y Energía, which determined maximum and minimum flowrates of 220 $m^3\ s^{-1}$ and 0.061 $m^3\ s^{-1}$ respectively, with an annual average of 0.425 $m^3\ s^{-1}$ (1952-1970) at the upper basin and 0.904 $m^3\ s^{-1}$ (1963-1968) in the end of the middle basin (Carica 1998).

Besides the urban and industrial activities, the main socioeconomic activity in the region is a mixed livestock-agriculture production system. One of the zones with greater activity is located at the upper part of Napostá Grande watershed. The upper part of Sauce Chico watershed is also occupied by a large livestock development; in the middle basin there are extensive horticultural lands while in the lower basin, close to the river mouth, there are numerous farms of smaller size. In Napostá Grande watershed, the horticultural land use area is small and there are only two horticultural producers located in the lower basin. The water of both rivers is used for irrigation throughout their watersheds and particularly in the lower basins, on horticulture crops.

3 METHODOLOGY

In order to calculate the flowrates of the Sauce Chico and the Napostá Grande rivers a technique of indirect gauging in natural channels was used (Chow 1982).The depth of the water was continuously measured in both watercourses using a limnimeter-phreatimeter Mod. LF-325 provided by Génica Ingeniería (Bahía Blanca, Argentina). This data enabled the calculation of the flowrates using the rating curve obtained by a working team of the Instituto Argentino de Oceanografía (IADO) thanks to the qualified support of the staff of the Hydraulics Laboratory of Universidad Nacional del Sur (UNS), both institutions of Bahía Blanca. The measure sections were selected considering their accessibility conditions and making sure that their flow regimes were continuous in the lower part of the watersheds near both mouths (Figure 2). To get a comprehensive view of a heavy rainfall event, a dense network of precipitation and stream gauges is necessary. However the rainfall data available for this study area is scarce and is provided by two rainfall gauge stations located in the middle and the lower part of Napostá Grande watershed. The first one, a Davis GroWeather Station is placed in a town named Tres Picos (38°17´26.2" S and 62°10´16.6" W) and the other one is situated in the Comandante Espora Aerometeorologic Station (38°43´47.8" S and 62°09´5.9" W).

The analysis of the precipitations registered between November of 2005 and September of 2007 in the middle and lower basins of the Napostá Grande river helped establish the relation between the rainfall and the flowrates obtained for that period. Although the data series corresponds only to three years, it is enough to make a preliminary characterization of the region studied. In addition, a study was carried out based on the hydrograms showing the precipitation events of a single heavy rain and their relation with the flowrate increase at the mouths of both courses. In order to analyse the interaction between fluvial and marine environments in the inner part of Bahía Blanca estuary, a preliminary hydrodynamic model using MOHID Water Modelling System was developed. This application allowed the simulation of the effects of exceptional flowrates upon the estuary according to the rainfall events above referred. The grid used was 200x200 with 5 layers and the geometry was of the sigma type.The parameters incorporated for the MOHID execution were: bathymetry of the internal zone of the estuary, salinity and temperature data of its waters and flowrates higher than 90 $m^3\ s^{-1}$ registered for their main tributary watercourses. Other information considered included level curves and speed and direction of local winds. The estuary bathymetry was provided by Doctors Susana Ginsberg, Eduardo Gomez and Gerardo Perillo from Instituto Argentino de Oceanografía (IADO). Wind and estuary water parameters were obtained from gauge stations by personnel of the same institute.

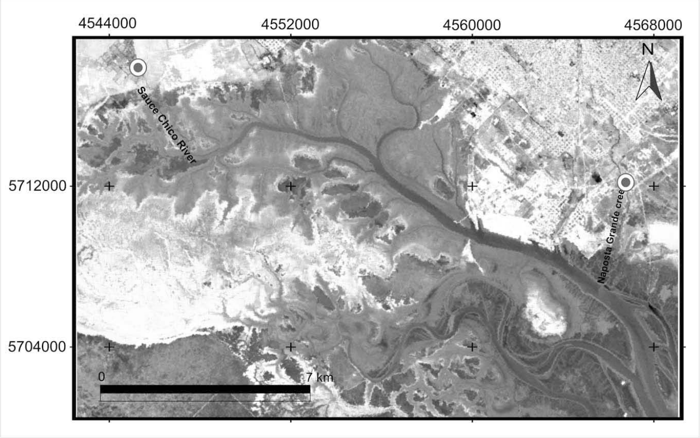

FIGURE 2: *Location of the limnimeters at the head of Bahía Blanca estuary.*

4 FLOWRATE VARIATIONS OF SAUCE CHICO AND NAPOSTÁ GRANDE RIVERS

For the period of November 3, 2005 to October 31, 2007, the average flowrate of the Sauce Chico was 1.72 $m^3\ s^{-1}$. Due to a damage suffered by the limnimeter sensor, no data was gathered from February 21 to March 31, 2006. The highest value was recorded in spring time with a peak of 18.32 $m^3\ s^{-1}$ in October 2006 while the minimum flowrate was registered in summer time, with a value 0.03 $m^3\ s^{-1}$ in January 2007. The seasonal average flowrates for this period are shown in figure 3. The average monthly flowrates of the analysed series presented their highest levels in the months of October 2006 and March, September and October 2007. The periods of low water for the Sauce Chico occurred during the months of December when precipitations were not at their lowest levels but there was elevated evapotranspiration, high sunstroke and soil moisture insaturation in the basin. As regards the year 2006, with the exception of the values corresponding to October, the average monthly flowrates in the lower part of Sauce Chico were always lower than its annual average flowrate (1.47 $m^3\ s^{-1}$). The minimum absolute value registered was 0.11 $m^3\ s^{-1}$ and the higher annual flowrate was 18.32 $m^3\ s^{-1}$. During 2007 the maximum flowrate registered for this series was 17.36 $m^3\ s^{-1}$ and its minimum value was 0.03 $m^3\ s^{-1}$. The annual average flowrate calculated was 2.12 $m^3\ s^{-1}$. The average flowrate of the Napostá Grande was 1.05 $m^3\ s^{-1}$ (between 9 March 2006 and 12 November 2007). The highest value was recorded in autumn time with a peak of 167.10 $m^3\ s^{-1}$ in April 2007. The minimum flowrate, 0.30 $m^3\ s^{-1}$, occurred in August 2006.

Although the limnimeter in the Napostá Grande river was installed in November 2005, works of cleaning and rectification of the river bed were carried out during February 2006. For this reason no flowrate data was registered prior to March 9th of that year. During 2006, the average monthly flowrates of the Napostá Grande presented their maximum values in the spring months of October and November with averages of 1.12 and 1.10 $m^3\ s^{-1}$ respectively (Figure 5). The lowest values were registered during the month of March with average volumes of up to 0.38 $m^3\ s^{-1}$ Concerning the year 2007, the maximum average flowrate for the period analyzed was observed during the autumn, with an average value of 3.91 $m^3\ s^{-1}$ in April. The minimum rate, 0.73 $m^3\ s^{-1}$, was registered during January (Figure 6).

5 AVERAGE MONTHLY FLOWRATES AND RAINFALL RELATION

The monthly precipitations in the Sauce Chico watershed do not indicate an important seasonal variability throughout year 2006. The larger amounts were registered during October, July, June and January (127, 72, 61 and 34 mm respectively). For this year, a relationship between the flowrates and intense rainfall periods, such as the one reported in October, is observed. However correlations between the average monthly flowrates and precipitations are not usually observed (Figure 7). During the autumn a considerable divergence between both variables can be noticed since the flowrates increase at the same time as the rainfalls diminish significantly. In 2007 there is again no coincidence between the variables. February is the month when the marked increase in the flowrate (a maximum of around 3 $m^3\ s^{-1}$) presents values which are the most out of step following a rainy month with a maximum of 139 mm. In addition, April is the only month in which a concurrence between the analysed parameters is observed.

The relationship between rainfalls and average monthly flowrates displays a different behaviour for the Napostá Grande with respect to the one observed in the Sauce Chico, presenting a better correlation between both parameters through all analysed series (Figure 8). In the year 2006 maximum average flowrates coincide with the highest rainfall averages registered during October (1.12 $m^3\ s^{-1}$ and 99.9 mm respectively). Furthermore soil humidity conditions at the watershed support the high average flowrate of November (1.10 $m^3\ s^{-1}$). Moreover, a coincidence between the rainfall increase and flowrates for the month of July is observed, with values of (0.54) $m^3\ s^{-1}$ and 51 mm respectively. During 2007 rainfalls increased monthly, reaching a maximum of 132 mm in April which resulted in a significant increase of the average monthly flowrate (3.91 $m^3\ s^{-1}$). In order to understand the behaviour of these watersheds in relation to the rainfalls the time of concentration, also denominated balance or response time, was calculated. Llamas (1993) defines it as the time required to achieve the stationary state during a uniform heavy rain; that is, the time necessary for all the system (for the whole river basin) to contribute effectively to the generation of flow in the water-drainage. The time that takes a water particle to travel from the point of the watershed furthest removed from the water-drainage where it has fallen to the actual water-drainage site is the one usually considered as the time of concentration. This was obtained following Kirpich' equation (Wainelista, 1997):

Time of concentration (minutes): $t_c = 3.97\left(\frac{L^{0.77}}{S^{0.385}}\right)$

where L is the length of channel (km) and S is the average slope (mm^{-1}). The time of concentration is determined by the physical characteristics of the river basin (surface, average slope, length of the channel). The time of concentration for the Sauce Chico river is of 17 hours 20 minutes and for the Napostá Grande the time of balance is of 7 hours 17 minutes.

6 SURFACE FLOW HYDROGRAPH SEPARATION METHOD

Surface or total flow (F) of a water stream is mainly composed of (1) the direct runoff or overland flow (Fd), produced in the watershed above the place where it is measured, and resulting from precipitation that does not infiltrate into the soil surface and that is not retained (for example in the plant canopy, buildings, dams, etc.), and (2) the base flow (Fb), resulting from water that infiltrates into the soil, goes through the subsurface and eventually comes to the surface, the discharge of groundwater to the watershed being the sum of both flows, that is F = Fd + Fb. In order to analyze the behavior of the flowrates for both the Sauce Chico and the Napostá Grande a hydrograph was done (see Oliveira et al., this volume), reflecting events where the watercourses swelling was the result of a single heavy rain which was not preceded by immediate rainfalls (for the considered series). Regarding the Sauce Chico, from the analysis of its rising limb it was determined that, while the hydrograph peak takes five days on average to be perceived, the direct runoff lasts an average time of eight days. The duration of the discharge of groundwater (depletion curve) is proportional to the maximum flowrate reached (Chow 1982). Therefore for gauges up to 4,5 $m^3\ s^{-1}$ the depletion curve can last more than twelve days (Figure 9). An analysis of the hydrographs for the storms occurred in the area of the Napostá Grande was also carried out. It was determined that, whereas the average time it takes the stream to rise (rising limb) is about three days, the period required for its draining (recession curve) is eight days on average. Meanwhile, groundwater runoff (depletion curve) for flows of up to 3 $m^3\ s^{-1}$ extends over a period of seven days (Fig. 10).

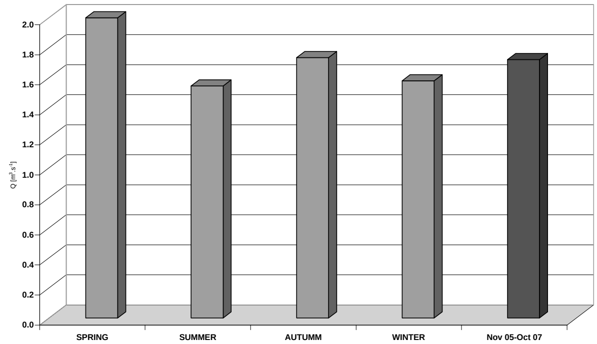

FIGURE 3: *Seasonal average flowrates and general average for the evaluated series (November 2005-October 2007) in the lower part of Sauce Chico River.*

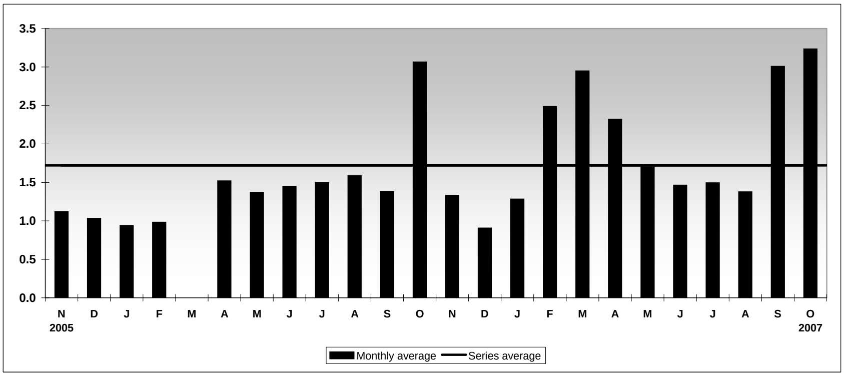

FIGURE 4: *Average monthly and series average flowrates at Sauce Chico lower basin.*

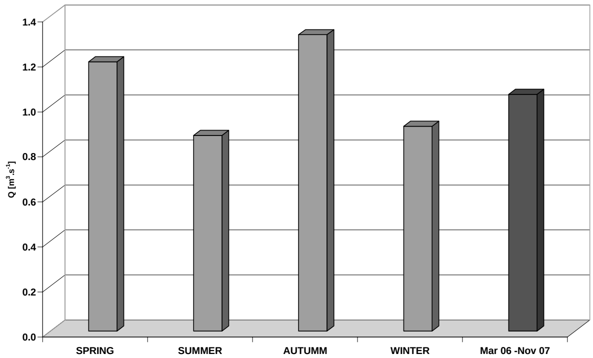

FIGURE 5: *Seasonal average flowrates and general average for the evaluated series (March 2006-November 2007) in the lower part of the Napostá Grande watershed.*

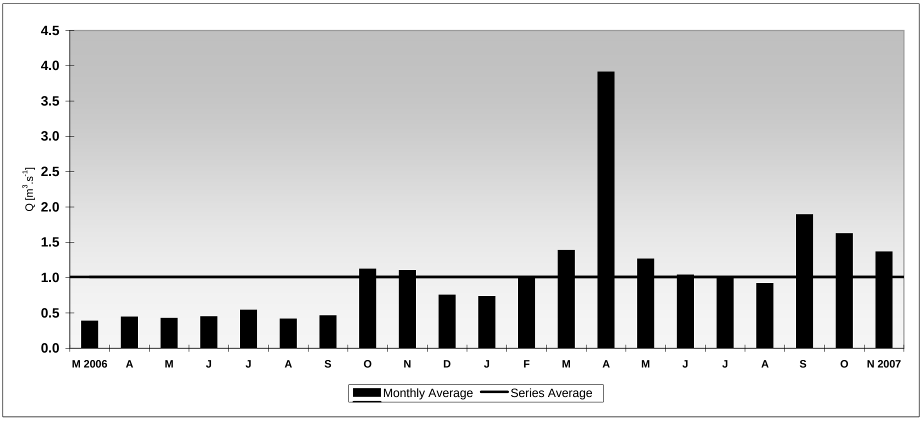

FIGURE 6: *Average monthly flowrates and series average in the Napostá Grande lower basin.*

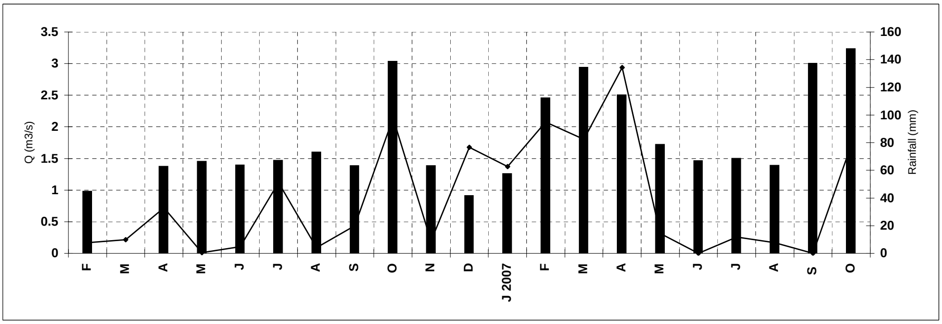

FIGURE 7: *Curve of average monthly volumes in relation to monthly precipitation in the Sauce Chico river.*

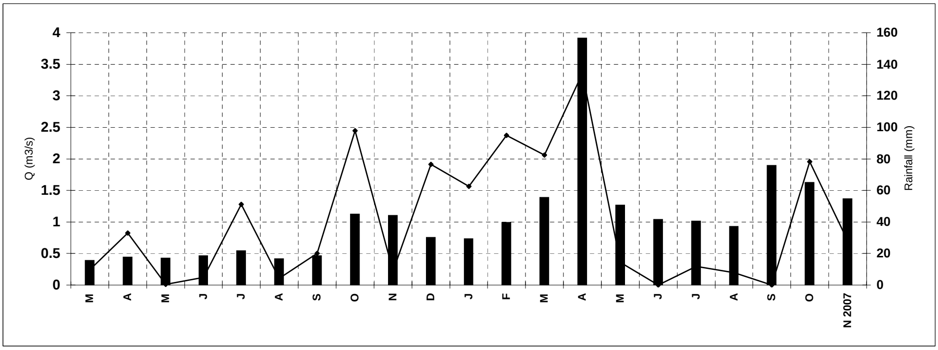

FIGURE 8: *Relationship between average monthly flowrates and monthly rainfalls in the Napostá Grande river.*

7 APPLICATION OF THE MOHID WATER MODELLING SYSTEM

The simulation of the discharge of both the Sauce Chico and the Napostá Grande waters into the estuary enabled the observation of a significant difference in the hydrodynamical behavior at its head. The mouth of the Sauce Chico is located in a mesotidal zone with a gentle slope and an important network of medium-depth channels. Therefore, its flow is constantly conditioned by the characteristics of the sector. The morphology of the islands, which are no higher than 2.10 m, and the complex network of tidal channels result in a significant run-off at low tide in exceptional rainfall events. The maximum depths range from 6 m to 1 m in the innermost part of the head and sediments are predominantly sandy (Gómez et al. 1996).

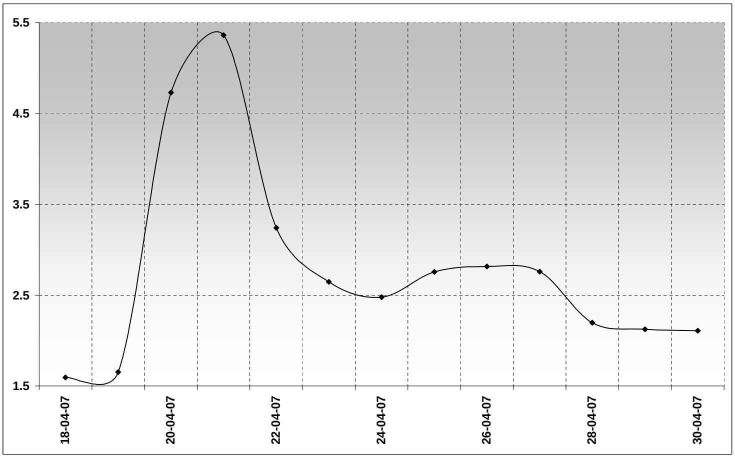

FIGURE 9: *Hydrograph of the Sauce Chico river.*

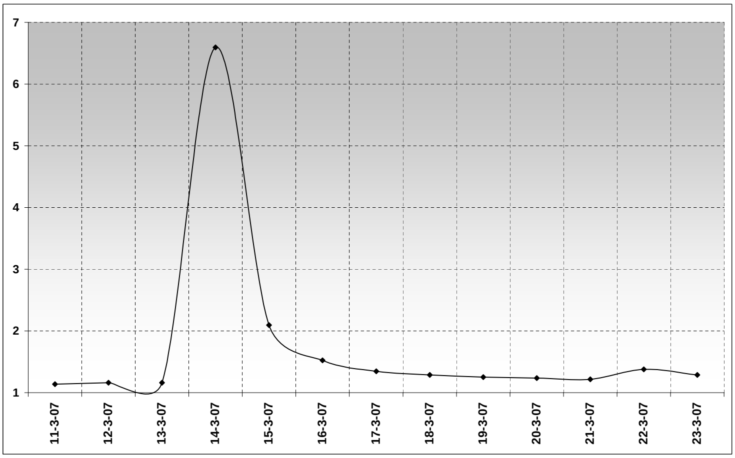

FIGURE 10: *Hydrograph of the Napostá Grande river.*

Based on measurements taken on the site, the simulation showed a maximum discharge speed of 0.5 $\mathrm{m\,s^{-1}}$ and an exceptional flowrate of 67 $\mathrm{m^3\,s^{-1}}$ resulting from an extreme precipitation event of a single heavy rain (Figure 11). The Napostá Grande presents a slight discharge at its mouth, which immediately interacts with the morphology of the area- in other words, its waters are channelled, reaching a speed no higher than 1 $\mathrm{m\,s^{-1}}$. The maximum flowrate in this simulation is 90 $\mathrm{m^3\,s^{-1}}$. The depths in the inner sector of the estuary range from 5 to 15 m, being about 20 m at high tide and the wind blows predominantly from the north-west (Figure 12).

FIGURE 11: *Simulation of the discharge of the Sauce Chico river.*

8 CONCLUSIONS

This preliminary study on the behaviour of the Sauce Chico and the Napostá Grande rivers shows that, although they are both part of Atlantic basins, they present singular characteristics. From a hydromorphological point of view, for instance, both watercourses flow across lands with different gradients and determinants. The characteristics of both the use of the land and the productive activities carried out in the area could modify the flowrates registered by the stations located in the lower basins; nevertheless, the absence of data upstream and downstream of sectors with great productivity precludes verifying this supposition. Despite the lack of a more extensive precipitation database over the November 2005 - September 2007 period the series analysed has helped establish a preliminary characterisation of the flowrates over the last two years and their direct correspondence with rainfall events. Once the flowrates of both watercourses have been characterised, the data obtained can be used in the construction of a hydrodynamical model. The simulation of the discharge of the Sauce

Chico and the Napostá Grande not only aided in determining the differences presented by the hydrodynamical behaviour in the inner sector of Bahía Blanca estuary, it also showed how these features influence the runoff in both watercourses. The speeds registered were 0.5 and 1 $m\,s^{-1}$ and the flowrates ranged between 67 and 90 $m^3\,s^{-1}$ for the Sauce Chico and the Napostá Grande, respectively.

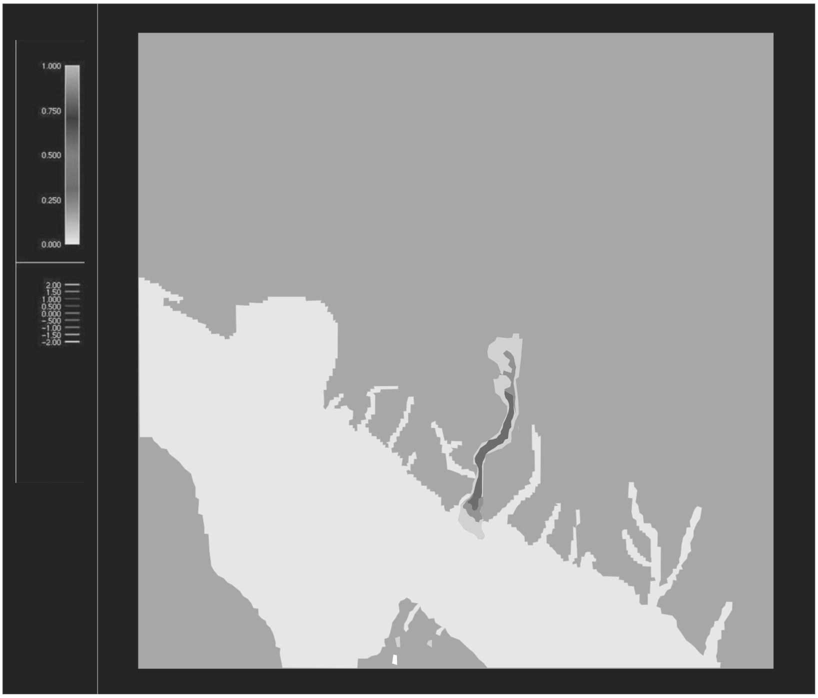

FIGURE 12: *Simulation of the discharge of the Napostá Grande river.*

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HYDRODYNAMICS AND SEDIMENTS IN BAHÍA BLANCA ESTUARY: DATA ANALYSIS AND MODELLING

F.J. CAMPUZANO, J.O. PIERINI AND P.C. LEITÃO

1 INTRODUCTION

The Bahía Blanca estuary, as indicated by its name, is a bay or rather collection of bays located on the Southern part of Buenos Aires Province (Argentina) on the uppermost terrace of the Argentinean Continental Shelf (ACS), according to the bathymetric division proposed by Parker et al. (1997). The channels and bays with North-West to South-East orientation run through extensive tidal areas, namely from North to South: Principal Channel, where the main human settlements are located, Falsa Bay and Verde Bay. On the Principal Channel one of the most important deep water ports of Argentina is located, and though the harbours and the navigation channel are dredged to a nominal depth of 13 m, a tidal prediction model "is urgently required" for navigational security (Perillo and Piccolo 1991).

The Principal channel has an elongated shape with a total length of 68 km, being 200 m and 3-4 km wide near its head and mouth, respectively (Aliotta and Perillo, 1987). The Principal Channel ends inland in a salt flat known as Salitral de la Vidriera. Falsa and Verde bays are also funnel shaped with total lengths of around 30 km and around 4 and 6 km wide respectively. In the study area (Figure 1), average depth is around 10 m with a maximum value of around 22 m. Intertidal areas account for about 40% of the domain represented in Figure 1 and have a concomitantly large influence both on water and sediment dynamics. In the nearby coastal area the isobaths have a NE-SW orientation (Pierini 2007).

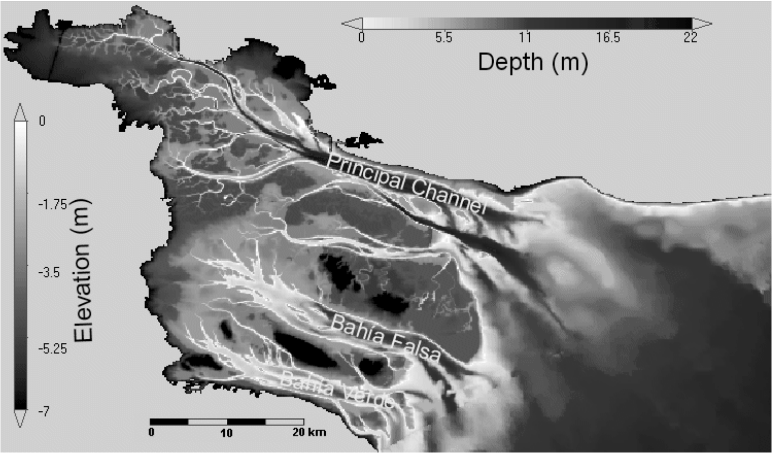

FIGURE 1: *Bahía Blanca model domain bathymetry and elevation of intertidal areas.*

Water circulation in Bahía Blanca estuary is mainly driven by tides, though it complex geometry and the prevailing winds produce deviations from the astronomical tides (Perillo and Piccolo 1991, Palma 1995, Etala 2000). Though it traditionally has been considered as an estuary (Piccolo and Perillo 1990), nowadays the only natural fresh water sources discharge into the Bahía Blanca bay innermost area (Heffner et al. 2003) at low flowrates. As such, only this inner area might be considered a true estuary.

2 GEOMORPHOLOGY

The Bahía Blanca estuary is a dynamically active system in which the main driving force that contributed to its formation is no longer present. Melo et al. (2003) described the evolution of the Bahía Blanca area from 20000 years BP when the study area was a coastal plain that the Colorado River traversed on its way to the open ocean. The freshwater inputs began to grow a deltaic front that, together with the increase in river flow and sea level rise, by 6000 years BP created the system of channels. From 5000 years BP on, the partial decrease of the mean sea level, along with the progressive migration to the South of the Colorado River and the migration North of other creeks decreased fresh water inputs into the Bahía Blanca system. Those losses were compensated by the entry of oceanic water that led the system to the present situation. The Colorado River discharges nowadays South to the Bahía Blanca estuary with an average flow of 138 $m^3\ s^{-1}$. However, it appears to have almost no influence on hydrodynamics or in sediments input into the system anymore (Cuadrado et al. 2002). Currently, the only natural fresh water sources in Bahía Blanca estuary are Sauce Chico River, Napostá Grande creek and Maldonado creek, which is formed by the junction of Saladillo de García and Dulce creeks (Heffner et al. 2003).

At present, the Bahía Blanca estuary comprises three large interconnected parallel tidal channels with a NW-SE orientation, namely from South to North: Verde Bay, Falsa Bay and Principal Channel. To these bays arrive channels of all types and dimensions, from large straight tidal channels to meandering creeks and gullies (Ginsberg and Perillo 2004). The Principal Channel is the only bay that receives freshwater discharges and near its mouth shifts from the funnel shape to a meandering channel. This main channel is connected mainly at its southern margin to other minor tidal channels. On the other hand, Verde and Falsa bays´ head act like a basin catchment where the different tidal creeks join. Nevertheless, the system is in permanent evolution as was observed by Ginsberg and Perillo (2004) who detected a lateral migration on some tidal channels at a rate of 25 $m\ yr^{-1}$.

3 HYDRODYNAMIC DATA DESCRIPTION AND ANALYSIS

3.1 Tides

In the Bahía Blanca estuary, tidal data are collected in three different stations (Table 1, Figure 2) from the open ocean to the inner area of the Principal Channel: Torre Mareográfica (hereafter TM), Puerto Belgrano Harbour (hereafter PB) and Ingeniero White Harbour (hereafter IW). Ten-minute water level timeseries for the three permanent tidal gauges were provided by the Bahía Blanca Port Consortium (CGPBB) for more than three years. The Bahía Blanca estuary is a mesotidal estuary (Piccolo and Perillo 1990, Cuadrado et al. 2002, Ginsberg and Perillo 2004), with a tidal range of 2-4 m. Averages performed on each tidal gauge water level data for the year 2000 show that the tidal range increases along the Principal channel from mesotidal to nearly macrotidal.

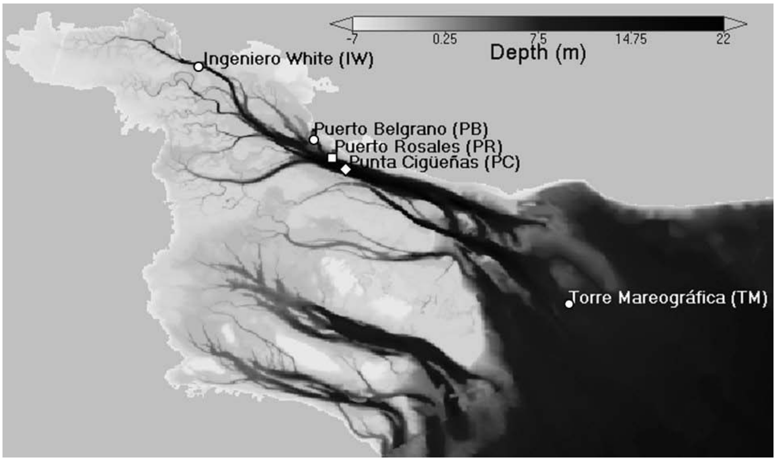

FIGURE 2: *Location of the tidal (circles), current (diamond) and meteorological (square) observation stations in the Bahía Blanca estuary.*

Harmonic analyses performed on the water level time series using the POL/PSMSL Tidal Analysis Software Kit 2000 (TASK-2000) resulted in 62 tidal components for each tidal gauge. Observed and predicted with the tidal components, water levels for each tidal gauge present a high coefficient of determination (R^2) from 0.87 for TM station to 0.91 for IW station.

Mean water level (Z_0) for the three permanent tidal stations shows a difference of 0.70 m between the two most widely separated stations (55 Km). This difference will be addressed later in the modelling chapter. The tidal components of greater amplitude are given in Table 2, the list is led in all the stations by the semidiurnal components, followed by the diurnal components whose influence decreases headwards. The Formzahl coefficient (F) provides a quantification of the degree of influence of the semidiurnal and diurnal components. It consists of the division of the sum of the two main diurnal tidal components (K_1 and O_1) by the sum of the two main semidiurnal components (M_2 and S_2). According to the result of the F ratio, tides can be considered diurnal (F$>$3), semidiurnal (F$<$0.25) or mixed (0.25$<$F$<$3). In Bahía Blanca the F ratio decreases as the tidal wave progresses up the estuary, being 0.26 at TM station and 0.19 at IW station. Tides in the Bahía Blanca estuary can thus be classified as semidiurnal. As the tide heads inland, the weight of the so-called compound tides or overtides

(M_4, MS_4 and MN_4), generated by nonlinear interactions of primary constituents, increases. This effect takes place in shallow water areas, also increasing the tidal range. These new constituents are important for turbulence, tidal asymmetry and mean water level (Wang et al. 2002).

According to Wang et al. (2002), tidal asymmetry can be assessed as a comparison of the amplitudes and phases of the dominant M_2 component with the overtide M_4. A relative phase ($phi = 2\phi_2 - \phi_4$) between 0° and 180° indicates longer duration of the flood than ebb (flood-dominant) and vice versa. After analysing the data for the three tidal gauges in Bahía Blanca, an evolution from the flood-dominated coast to the ebb-dominated head can be observed. In Table 3 the values obtained for the years 2000-2003 analysis are summarised. The α parameter calculated as division of the M_4 amplitude by the M_2 amplitude indicates the degree of asymmetry, this parameter increases headwards due to a larger increase of the M_4 component than the M_2.

TABLE 1: *Location, observation period and source of hydrodynamic and atmospheric data.*

Station	Longitude	Latitude	Period	Source
TM	61º43'12" W	39º08'56" S	23/08/99 - 31/12/03	CGPBB
PB	62º06'15" W	38º53'48" S	23/08/99 - 31/12/03	CGPBB
IW	62º16'08" W	38º47'26" S	23/08/99 - 31/12/03	CGPBB
PR	62º04'00" W	38º53'00" S	01/01/01- 01/12/01	IADO
PC	62°03'09" W	38°56'40 "S	17/03/97 – 21/04/97	SHN

TABLE 2: *Principal tidal components for each tidal gauge (H=amplitude; φ=phase).*

Tidal Component	Speed (deg/hr)	Puerto Ing. White		Puerto Belgrano		Torre Mareográfica	
		H (cm)	φ (º)	H (cm)	φ (º)	H (cm)	φ (º)
Z_O	0.000	263.544	0.000	245.635	0.000	193.273	0.000
M_2	28.984	169.123	186.072	153.523	177.906	115.864	157.094
L_2	29.528	25.475	255.364	22.471	245.987	15.977	220.692
N_2	28.440	23.983	103.593	21.080	95.833	15.239	76.690
M_4	57.968	22.764	178.277	16.141	184.234	6.396	171.968
S_2	30.000	21.589	307.350	20.632	298.669	16.843	274.215
K_1	15.041	21.151	61.178	21.481	54.527	20.345	45.066
O_1	13.943	15.528	0.701	16.186	355.431	15.426	344.967
MU_2	27.968	14.523	291.531	14.015	282.016	12.590	264.735
NU_2	28.513	10.954	137.915	9.637	130.029	7.632	113.261

3.2 Currents

The dataset collected at the Punta Cigüeñas oil pipe monobuoy (Figure 2, Table 1) by the Navy Hydrographical Service (SHN), consisting of current speed and direction every 15 minutes during 35 days, was the most representative dataset available in the Bahía Blanca estuary. The average value for the whole series is 0.64 $m\,s^{-1}$ with a maximum of 1.33 $m\,s^{-1}$ and the directions are practically opposite pointing to 295° and 115° respectively (Figure 3). Ebb tide maximum velocities are slightly higher. In order to validate and calibrate the model for different periods, the harmonic components of the dataset were obtained for each component of the velocity using the TASK-2000 software. Component U intensity of the velocity is almost double the V component, due to the orientation of the Principal Channel. The most important harmonic component for velocity is the M_2 which is responsible for more than 40% of tidal velocity, followed by the N_2 component that represents an 8% of the intensity of the currents. The tidal components are able to explain 82% of the total variability of the currents measured during the sampling period, the remainder of the variability may be due to variations in mean sea level and to atmospheric effects.

TABLE 3: *Principal tidal components for each tidal gauge (H=amplitude; φ= phase).*

	Puerto Ing. White	Puerto Belgrano	Torre Mareográfica
M_2 H (cm)	169.123	153.523	115.864
M_2 φ (º)	186.072	177.906	157.094
M_4 H (cm)	22.764	16.141	6.396
M_4 φ (º)	178.277	184.234	171.968
α	0.135	0.105	0.055
Phi	193.867	171.578	142.22

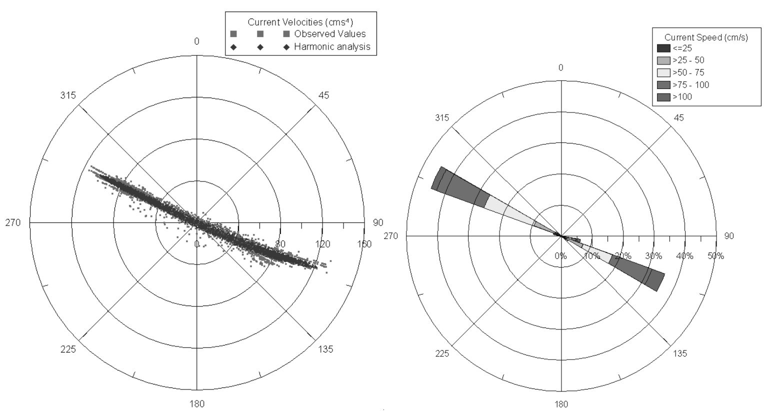

FIGURE 3: *Current intensities and directions observed and obtained through harmonic analysis for the same period (Left) and observed current values analysis (Right).*

3.3 Freshwater inputs

Main natural fresh water inputs in the Bahía Blanca estuary are located at the innermost part of the Principal Channel. Basically, only Sauce Chico River and Napostá Grande creeks have a considerable discharge in the study site (Heffner et al. 2003) with average annual flows of 5.80 and 2.68 $m^3\ s^{-1}$ respectively. Nevertheless, high runoff peaks have been registered, for instance in the Sauce Chico River several times discharges over 50 $m\ s^{-1}$ have been observed (Perillo and Piccolo 1991). Although the source of both rivers is located in the Sierras de la Ventana mountain range, both catchment areas are not subjected to the same processes and present different discharge peaks during the year. Sauce Chico River has two main peaks, one in autumn (Feb-Jun) and a stronger one during spring (Aug-Dec). However, the Napostá Grande Creek flow shows peaks in winter (Apr-Aug) and spring (Aug-Dec), both with similar intensities. The spring peak in both rivers corresponds to the rainiest period in the Bahía Blanca region. Sauce Chico presents a flow pattern very influenced by the local rain pattern on Bahía Blanca region while Napostá is more influenced by the precipitation near the Sierras de la Ventana (Piccolo et al. 1987). Figure 4 shows the monthly average flow from data collected during the 1993-1999 period provided by Aguas Bonaerenses Sociedad Anónima (ABSA). There are other creeks flowing into the Bahía Blanca estuary, such as Galván, Saladillo de García and Maldonado creeks with an overall flow lower than the Napostá Grande creek (Piccolo et al. 1987).

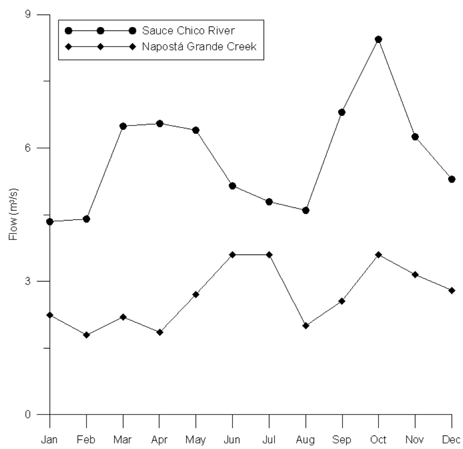

FIGURE 4: *Napostá Grande Creek and Sauce Chico rivers monthly average flow.*

3.4 Atmospheric influence

The SMN (National Meteorological Service, 1985) in Capelli and Campo (2004) described the study area as dominated by the pressure systems from the West and Southwest and the presence of a semi permanent high pressure over the South-western Atlantic centred at 30°. The circulation around the Western edge of the anticyclone results in strong winds from the NW and N. When winds blow from NW, they produce lower water levels than those predicted by the astronomical tides; on the other hand, when wind blows from SE water is piled up in the inner areas of the bay increasing the water level. This phenomenon is locally known as sudestadas. According to Perillo and Piccolo (1991) and Piccolo and Diez (2004) the predominant winds from N and NW produced an advance of the low water time, a delay on the high water time and reduced the predicted levels both for high and low tide.

Thirty minute data observed at Puerto Rosales meteorological station (Table 1, Figure 2) by the IADO for the period January-November 2001 showed prevailing winds blowing from Northern sectors, mainly from the North and Northeast. Maximum intensity recorded on this period was 23.2 $\mathrm{m\ s^{-1}}$ and an average value of 6.2 $\mathrm{m\ s^{-1}}$. During most of the time intensities do not surpass 10 $\mathrm{m\ s^{-1}}$.

4 HYDRODYNAMIC MODEL: SETUP AND RESULTS

The implementation of a validated hydrodynamic model would serve for applications that would aid in coastal management and would serve as a part of the DPSIR approach. The MOHID hydrodynamic model (http://www.mohid.com/) was used to force a Bahía Blanca 2-Dimensional domain model covering the study site. The most relevant characteristic is the order of magnitude of the horizontal and vertical dimensions. In our case study the horizontal extent is 70 km, while the average depth is 10m.

4.1 Modelling grids

To study the different hydrodynamic processes involved in Bahía Blanca, two different spatial scales covering the main area of interest were adopted with different resolutions: 0.01° and 0.002° horizontal step covering from the coordinates -61.41W, -39.38S to the inner estuary (Pierini 2007), hereafter referred as LoRes grid and HiRes grid, respectively. The main interests in developing different resolutions have been, on the one hand, to study different processes in diverse detail according to the available data and, on the other, to maximise the model runtime. Bathymetric data used for composing the different model domains came from the GEBCO digital atlas, a one minute global bathymetric grid database (IOC, IHO and BODC 2003) and from CGPBB data with a waterline obtained after evaluating 6 sets of Landsat 5 TM and Landsat 7 ETM data resulting in a high density bathymetry (50 m x 50 m) (Pierini 2007).

4.2 Tides

Tide enters each bay of the Bahía Blanca estuarine system by its Southern margin finding all the channels reduced in width and most of the intertidal areas emerged. As the tide advances, water starts to cover the intertidal area amplifying the submerged area. As a result of the interaction of the tidal wave with the shallow depths of the channels, overtides importance increases. Maximum tidal amplitudes are thus found at the innermost areas for each of the three different channels, and the absolute maximum was found in Principal Channel. During high tides, the connections between the three different bays reach their maximum width favouring water exchange between bays. When the tidal wave retreats, water scours from the inundated intertidal areas through tidal channels into the main channels, the time to evacuate those waters is long and so this process is still going on when a new tidal wave is already entering the system, as can be appreciated on the last image of the sequence (Figure 5). Due to the vicinity of the TM tidal gauge to the boundary of the Bahía Blanca domain, the model forcing consisted of imposing those tidal components along the entire coastal open boundary.

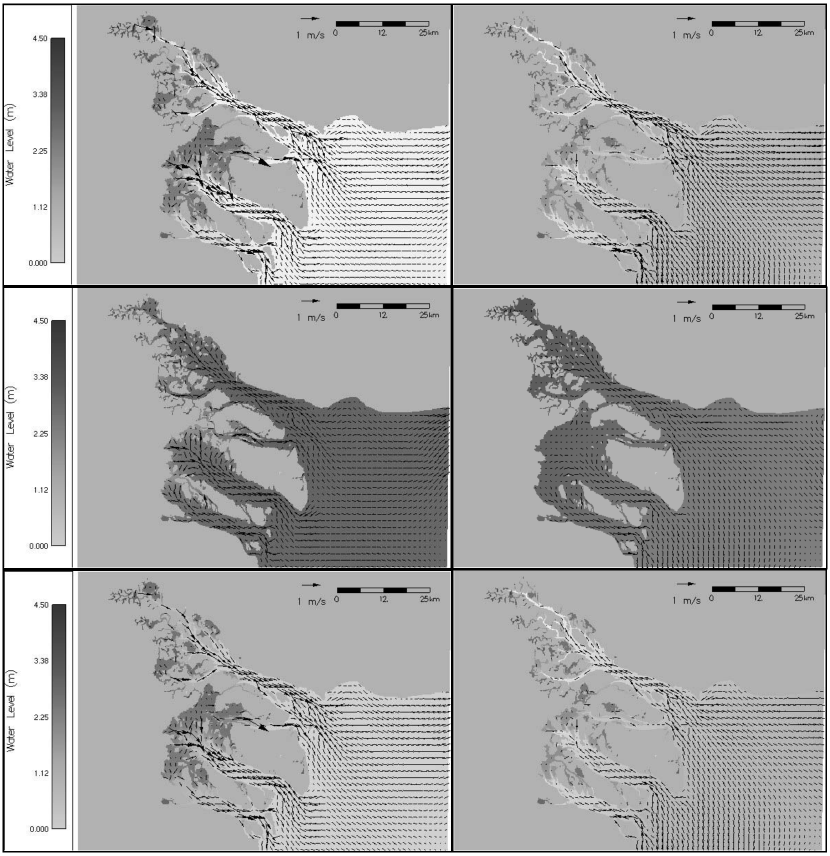

FIGURE 5: *Water levels for the HiRes domain during a period of 12 hours.*

The Bahía Blanca Estuary Principal Channel mean sea level, as stated in the data analysis, presented a permanent sea level difference of 70 cm between the most distant tidal gauges. That tidal amplification could not be explained by the increase of mean sea level due to tidal dumping. Using simultaneous validation of the water levels and the current intensities at Punta Cigüeñas (PC) station, it was concluded that using the mean sea level of the outermost station, TM, the right quantity of water was not entering the system, as the obtained currents at that station were reduced compared to the observations. On the contrary, when adding these 70 cm to the mean sea level of the TM station, both water levels and current intensities at PC station presented a high coefficient of determination.

Applying these conditions to the model, correlations between the expected water levels due to astronomical tides and model results obtained were performed obtaining correlation coefficients above 0.93 (Table 4). To further evaluate the comparison between the observed tidal levels obtained through harmonic analysis with the predicted levels obtained through modelling, the additional indicators suggested by Willmott (1982) were assessed (Table 4).

TABLE 4: *Model performance indicators at each tidal gauge.*

Station	Torre Mareográfica	Puerto Belgrano	Puerto Ing. White
Correlation (r)	0.985	0.987	0.966
Coef. Determination (r^2)	0.971	0.974	0.933
MB	-0.048	0.104	-0.054
RMSE	0.162	0.220	0.369
Skill	0.992	0.991	0.981

Mean bias (MB) indicates the average difference between the observed and predicted values. Predicted amplitudes vary in (bias/total amplitude*100) 1.37, 2.35 and 1.08% of the total amplitude at TM, PB, and IW stations, respectively. Root Mean Square Error (RMSE), or standard deviation, is the square root of the variance which represents that 95% of the model predictions do not differ (in absolute value) from the observations by more than 2xRMSE. The skill index is intended as a descriptive measure, and is both a relative and bounded measure which can be widely applied to make cross-comparisons between models. It provides similar information to the coefficient of determination, in the sense that it gives a measure of model performance, but it penalises models with greater bias. If the skill index is 1, the model presents an optimal predictive skill. Skill values for each station are close to 1, indicating a high degree of model performance.

4.3 Currents

Currents on Bahía Blanca estuary are mainly caused by tides. In addition, the importance of the geomorphology for the tidal asymmetry is reflected in the residual currents. Tidal currents at the mouth of the estuary are the result of the cumulative processes of the outer general circulation and the distortion produced by the estuary hydrodynamics. Flood velocities are generally larger than the ebb velocities outside the estuary due to the direct relationship between velocity and depth.

4.3.1 Instantaneous currents

The Principal channel, according to Piccolo et al. (1987), is dominated by ebb currents, maximum surface values are 0.80 $\mathrm{m\,s^{-1}}$ and 1.40 $\mathrm{m\,s^{-1}}$ for flow and ebb currents, respectively. Very similar values were obtained by Gómez et al (1996) integrating vertically the observations, with values of 1.05 $\mathrm{m\,s^{-1}}$ and 1.30 $\mathrm{m\,s^{-1}}$ for flow and ebb respectively. In the coastal area, measurements obtained during the Austral campaign in 1993 observed maximum velocities of 0.6 $\mathrm{m\,s^{-1}}$ with velocities over 0.3 $\mathrm{m\,s^{-1}}$ being exceeded for more than 30% of the time (Cuadrado et al. 2002).

Model results (Figure 6) present an ebb-dominated inner estuary in opposition to a flood-dominated coastal area in agreement to both the relative phase values (phi) obtained in Section 3.1 and Cuadrado et al. (2002). In Verde Bay maximum values are found near its mouth due to the channelling of the ebbing waters; in Falsa Bay, in addition to the mouth, it presents maximum values near its head in the region where water from the surrounding intertidal area scours and in the Principal Channel values are high along the main axis of the channel.

During flood, the high intensities on the three bays are located along each main channel slowly covering the intertidal areas where velocities relax. During ebb along the main axis of the Principal Channel values range between 0.7 m s^{-1} and 1.4 m s^{-1}, the maximum values being found near the upper reaches; maximum values of similar intensity are observed in Falsa Bay near its head and near the mouth of Verde Bay. On the three bays, velocities during ebb tide are higher than during flood. Low intensities are mainly found in the higher reaches of the intertidal areas. During flood, water piles up in the entry of the channels, with speeds of nearly 1 m s^{-1} being observed in the mouth and average speeds along the channel axis of 0.4 m s^{-1} on average. Peak velocities observed in the model on the intertidal areas during flow are generally below 0.3 m s^{-1}. It can be concluded that the model reproduces the principal features described in the literature.

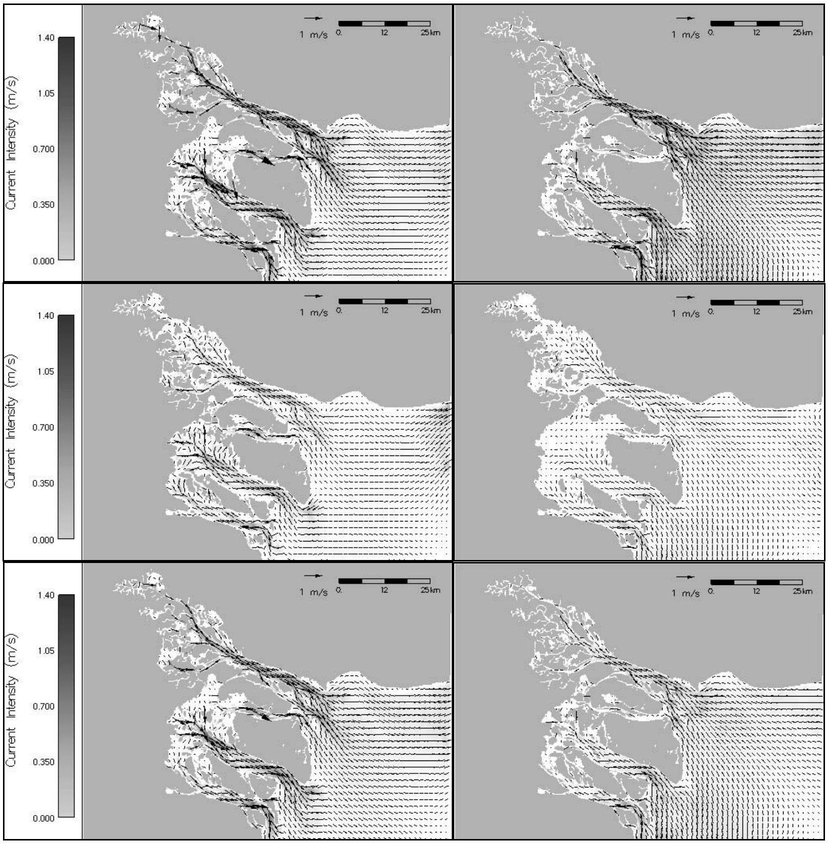

FIGURE 6: *Instant velocities for a 12 hour period in the Bahía Blanca estuary HiRes domain.*

4.3.2 Residual currents

The residual currents are a significant factor regarding the import and export of dissolved and particulate substances of the estuarine system. Two different residuals are relevant in terms of characterising the estuary, namely residual flux velocity and residual velocity. The residual flux velocity is calculated by the vertical integration of the specific residual water fluxes occurring during a defined period and is expressed in $\mathrm{m\,s^{-1}}$ and influences the transport of dissolved properties in the water column (i.e. nutrients). Figure 7 Top presents results corresponding to a month simulation average. As was pointed out by Perillo et al. (1987) and Pierini (2007) for the head of the Principal Channel, net transport in deeper areas is seawards while in shallower waters is landwards. The model reproduces this pattern. In the mouth of the Principal Channel due to its particular geomorphology with channels crossing intertidal areas, a recirculation pattern can be observed. As a consequence, the dissolved components would be transported into the intertidal areas and recirculated in the Principal Channel mouth retarding their export to the open ocean.

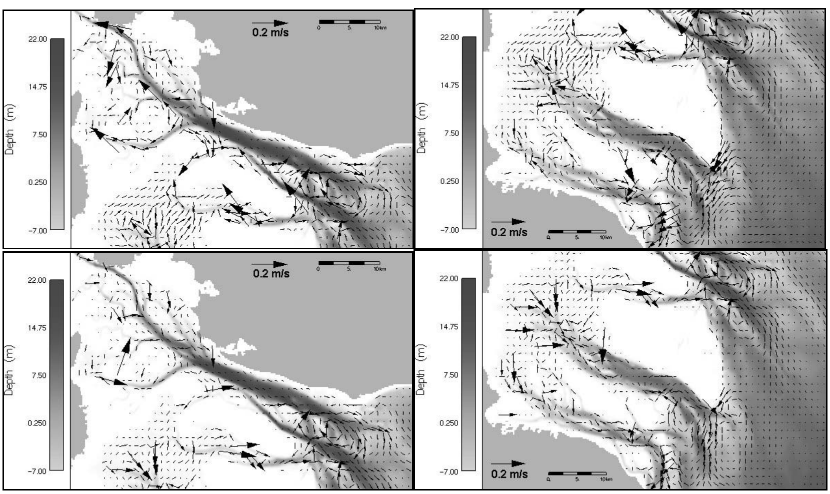

FIGURE 7: *Residual flux velocity (Top) and Residual velocity (Bottom) in the inner (Left) and the outer (Right) area of the Bahía Blanca estuary.*

The residual velocities are the net velocities obtained after balancing currents in different directions and they are of particular importance regarding particulate matter transport such as sediments. The most remarkable results (Figure 7 Bottom) are the net transport from the intertidal areas into the main channels. A recirculation pattern can also be found at the mouth of the Principal Channel. According to these results, the intertidal channels would be exporting sediments and other particulate matter into the main channels favouring erosion processes in the intertidal areas, although some recirculation at the Principal Channel mouth would slow down the sediments release into the coastal area (Pierini 2007).

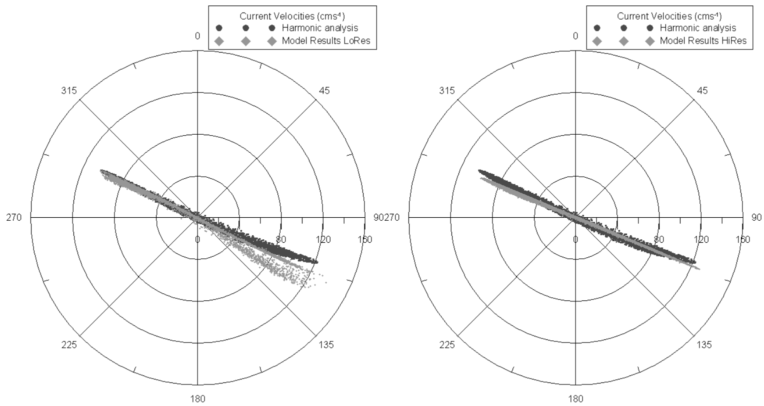

FIGURE 8: *Intensities and direction/s of the currents in Punta Cigüeñas stations obtained through harmonic analysis (dots) and modelling (diamonds) for LoRes and HiRes domains.*

4.3.3 Currents Validation

The main tidal current validation has been performed in the centre of the Principal Channel in the vicinity of the Punta Cigüeñas (PC) buoy (Figure 2, Table 1). After adding 70 cm to the mean sea level observed in TM station, it has been found that the model is able to reproduce the current intensities and directions (Figure 8). The coefficient of determination obtained when comparing both intensities is 0.72. In the HiRes domain this value increases to 80%. Model results and observed values present maximum peaks of intensity around 1.30 $\mathrm{m\,s^{-1}}$ during ebb conditions and nearly 1 $\mathrm{m\,s^{-1}}$ during flood conditions.

4.4 Wind effect

In order to study the effect of wind over the water levels observed, a month was simulated with different wind conditions. According to data analysis performed by Cuadrado et al (2005), predominant directions are NW, NNW and SE with a velocity range between 5 and 10 $\mathrm{m\,s^{-1}}$. A reference simulation with no wind was used to compare with simulations forced with constant wind conditions of 10 $\mathrm{m\,s^{-1}}$ coming from different directions (N, NW, S & SE). Maximum differences occur at the higher reaches of the estuary when winds blow along the Principal Channel axis, piling up water when it blows from the South and decreasing water levels when it blows from the North. In Table 5, mean sea levels are summarised for the three tidal stations analysed in this study.

5 SUSPENDED SEDIMENT DATA ANALYSIS

Despite the relevance of sediments accretion-erosion processes in the evolution of the geomorphology and hydrodynamics, suspended sediment data for the Bahía Blanca estuary are scarce and the origin of the sediments in the water column is not well-established. From the socio-economic point of view, the importance hinges on the need to maintain a nominal depth of 13.7 m for harbours and navigational channels (Perillo and Sequeira 1989). The Port Construction and Navigational Ways National Direction (DNCPVN, its Spanish acronym) dredges around 300,000 $m^3\,yr^{-1}$ in a 2 km stretch of El Toro Channel to maintain a 10 m nominal depth.

TABLE 5: *Wind direction and resulting mean sea level for each tidal gauge.*

Wind Direction	Torre Mareográfica	Puerto Belgrano	Puerto Ing. White
No Wind	1.93	1.93	1.94
North	1.84	1.64	1.62
Northwest	1.83	1.67	1.54
South	2.03	2.22	2.34
Southeast	2.01	2.25	2.44

Sediment balance is related to accumulation or erosion processes, also connected to the residual velocities. In the same sense, Ginsberg and Perillo (2004) concluded that the duration of asymmetry controls the trend of erosion in the channel system while the velocity asymmetry is responsible for the net sediment transport. Flow asymmetries characterized by shorter flood duration and higher flood current maximum (flood dominance) induces landward directed sediment transport while shorter ebb periods and greater ebb current maximum (ebb dominance) cause net outward sediment transport (Lanzoni and Seminara 2002).

The lack of major sediment point sources in the system, due to the reduced river flow, and the small contribution of sediments from the open ocean implies that the channel margins and intertidal areas should be regarded as the main source of sediments (Perillo and Sequeira 1989, Ginsberg and Perillo 2004). They considered sediment transport controlled by the degree of tidal asymmetry and trends on erosion governed by the degree of time asymmetry. According to Perillo and Cuadrado (1990) an erosion process is ongoing in the study area, sediments are eroded from the channel margins and eventually exported to the continental shelf. The materials affected are mainly very fine sand (38%), silt (37%) and clay (25%); with the sand transported as bedload and the other fractions in suspension (Aliotta et al. 2004).

As occurs with hydrodynamics, wind direction and intensity has an effect on suspended sediments in the estuary. Perillo and Cuadrado (1990) found in the nearby coastal area, Northeast of Bahía Blanca estuary mouth, homogeneous sediment concentrations between 70 and 90 $mg\,l^{-1}$ with North wind that decreased to 10 to 20 $mg\,l^{-1}$ when the wind shifted to a SSE direction. The origin of those sediments was found related to the Bahía Blanca sediment plume.

The Austral campaign on November 1993 covered the coastal vicinity of the Bahía Blanca showing a clear sediment front along the coastal area of the Bahía Blanca estuary (Figure 9, left), also observed in recent satellite pictures (Figure 9 right). In the Austral campaign, maximum concentrations of over 60 $mg\,l^{-1}$ are related to the Principal Channel while coastal values are between 20 and 25 $mg\,l^{-1}$ in agreement with the values estimated by Cuadrado et al. (2002). According to determinations done by NEDECO-ARCONSULT (1983) referred to in Perillo and Sequeira (1989), mean values for the middle reach of the estuary are around 90 mg l^{-1}. In the vertical, sediment concentrations present a nearly homogeneous distribution, with distance from the coast being the main factor determining the observed concentrations. This implied that a 2D model could be used to simulate the sediment transport and to test different hypotheses of the origin and fate of the cohesive sediments.

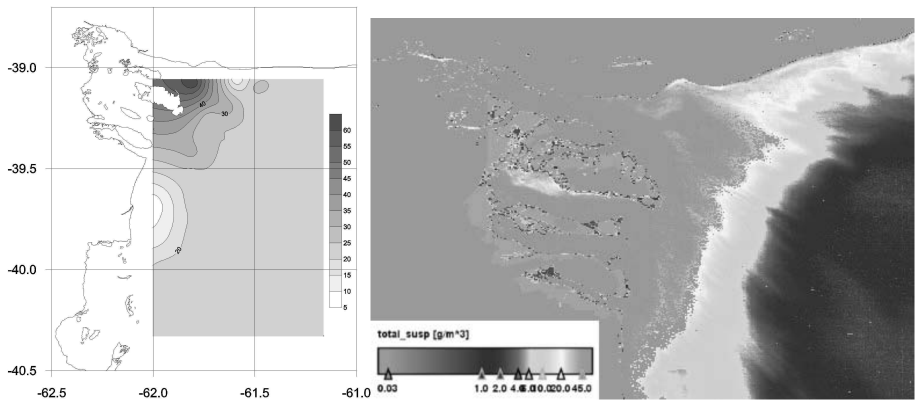

FIGURE 9: *Total suspended matter observed on the outer area of Bahía Blanca estuary measured during the Austral campaign at the surface in mg l^{-1} (left) and MERIS satellite image dated on 10/08/2004 (right).*

6 BAHÍA BLANCA ESTUARY COHESIVE SEDIMENTS CONCEPTUAL MODEL

Based on model tests, historical data and a bibliographic survey, the conceptual model to be tested assumes that water column sediments are resuspended during the high tides on the tidal flats and transported to each bay's main channel (Figure 10). In this manner, erosion is taking place due to the sediment loss into the open ocean although the erosion process is a slow process. This theory would also explain the wide ranges of values for turbidity and sediments in the inner areas of the system where tidal flats occupy larger areas in comparison with the main channel. Thus, depending on the stage of the tidal cycle when the data are collected and on the wind conditions, data would vary considerably on the inner stations with maximum values which may be up to six times larger than the minimum values observed.

The 0.01^{o} resolution grid was forced by the TM station tidal components taking into account the difference in the mean sea level. In the seabed, two different sources for the cohesive sediment layer were initially assumed. The intertidal area is considered as the main contributor of sediments to the water column, with a total thickness of available sediment of 0.2 m, on the other hand sediments in the channels and deeper areas were considered with a total thickness of 0.05 m that might undergo erosion and deposition processes. The sediments were also divided into a total of 15 layers; the first 8 were defined empty to allow consolidation of sediments, the next five represented each a 15% of the total sediments and the last two consisted of the 12.5% of the total sediments.

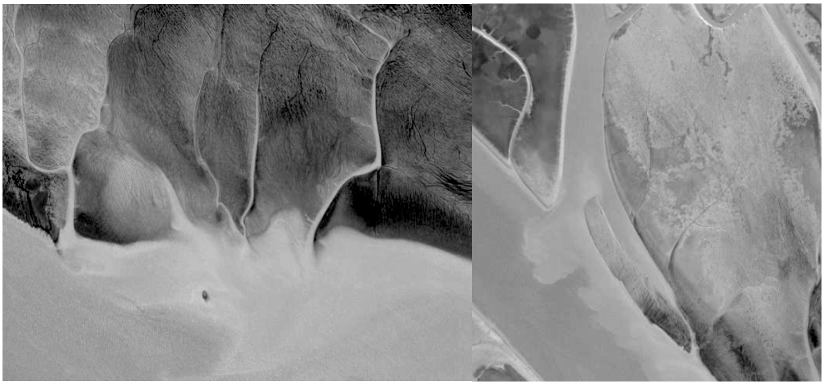

FIGURE 10: *Sediments loads from the intertidal areas into the Principal Channel of the Bahía Blanca estuary (Google Earth Captures).*

In addition, different critical shear stress profiles were considered on the intertidal and submerged areas. Sediments on the bottom of the channels were considered to be harder and also requiring higher stress intensity for resuspension. For the submerged area, an exponential increase of critical shear stress from 60 to 125 Pa was considered, while for the intertidal, a homogeneous value of 1.35 Pa was considered. The equations governing the erosion and deposition processes that would modify the original values are described in Chapter 5 of this volume.

The values assigned to the erosion and depositon parameters have been taken from those published by Perillo and Sequeira (1989) to model the cohesive sediment transport in the middle reach of the Bahía Blanca Estuary (Table 6). The settling velocity was computed by NEDECO-ARCONSULT (1983) as $1.25\text{x}10^{-3}\ \text{m s}^{-1}$. Perillo and Sequeira, based on the work of Hayter (1985), calculated a value for the first centimetre of newly deposited mud of $0.225\ \text{N m}^{-2}$. The same authors assumed that mud characteristics were similar to those in San Francisco Bay where Mehta (1985) obtained a value of $0.1\ \text{N m}^{-2}$. In this study a dry density for the sediment of $2300\ \text{kg m}^{-3}$ was used. To calibrate our model, different values for the erosion rate parameter have been evaluated, finally choosing a value of $10^{-5}\ \text{kg m}^{-2}\ \text{s}^{-1}$.

Those conditions are defined for the interface of the sediment with the water, also known as the fluff layer. Under this layer, a 3D sediment model that would provide this layer of sediments when they are eroded was deployed. The model boundaries are forced by the two rivers Napostá Creek and Sauce Chico River as the unique land source of sediments with a constant concentration of $25\ \text{mg l}^{-1}$, and a constant value for the ocean boundary of 21.6 mg l^{-1} (Cuadrado et al. 2002). Also, an initial value of $50\ \text{mg l}^{-1}$ was considered as the concentration in the water column for the entire system.

TABLE 6: *Parameters used to model the erosion and deposition of cohesive sediments.*

Parameter	Symbol	Value	Source
Critical shear erosion	τ_E	0.225 N m^{-2}	Perillo and Sequeira 1989
Erosion rate	E	$1 \cdot 10^{-5}$ kg m^{-2} s^{-1}	
Critical shear deposition	τ_{cd}	0.1 N m^{-2}	Perillo and Sequeira 1989
Constant settling velocity	w_s	$1.25 \cdot 10^{-3}$ m s^{-1}	Perillo and Sequeira 1989
Manning coefficient		0.020	

7 SEDIMENT MODEL RESULTS

The Bahía Blanca model, using the assumptions described above, was run for a period of five years until the bottom sediments and the water column concentrations stabilised, followed by a four year model run. A series of stations (Figure 11) running along the Principal Channel of Bahía Blanca Estuary was selected to evaluate the defined sediment model reaction to the forced hydrodynamics.

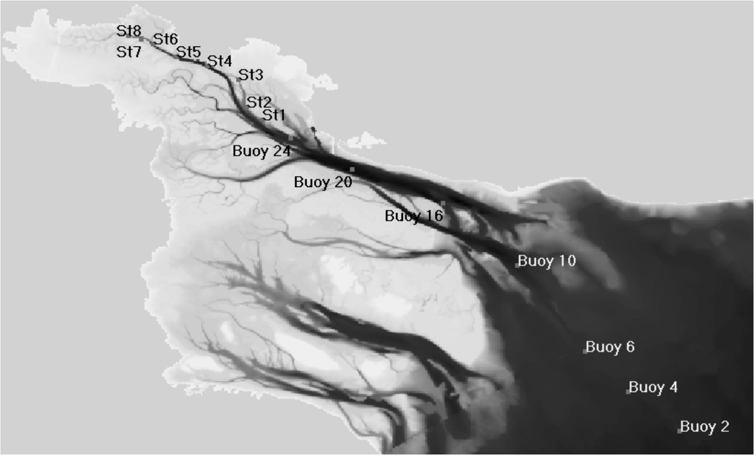

FIGURE 11: *Location of the monitoring stations in the Bahía Blanca estuary model.*

The model results show a decrease in suspended sediments from the head to mouth of each bay. Sediments are resuspended during the flood and transported through the intertidal channels into the main channels of each bay during the ebb tide. During the ebb tide, maximum concentrations are reached in the water column. During spring tides, the range of sediment concentrations on the water column is wider than during neap tides. The influence of spring and neap tides is detected even in the most distanced station Buoy 2. The transport of these sediments off each channel produces a sediment front along the Bahia Blanca coast (Figure 12). This feature was noticed in the MERIS satellite image (Figure 9, right) with similar concentrations to the observed in the model results.

The average values for the three year simulation depicted in Figure 13 follow a spatial distribution in which the innermost estuarine area cohesive sediment concentrations are stable, while seawards concentrations steadily decrease reaching constant values on the open ocean area. The maximum value is found in Station 3 due to its location in a narrow channel surrounded by intertidal areas parallel to the Principal Channel. According to our model results, the Falsa Bay, due to its particular geomorphology, presents the highest registered values. During ebbing tides, sediments from all the tidal channels converge, increasing their concentration. Unfortunately, data to calibrate and validate simply do not exist for the other bays that form the estuarine system along with the Principal Channel.

In terms of sediment erosion, a comparison in sediment height performed between the initial and last instant of the last three years simulation period shows that the Bahía Blanca estuary is under an eroding process in agreement with Perillo and Sequeira (1989). Erosion takes place mainly on the sides and heads of the tidal channels, while an area with significant deposition has not been found. In agreement with the results presented above, the head of the Falsa Bay is the area that presents a higher degree of erosion (Figure 14).

FIGURE 12: *Cohesive sediments concentration ($mg\ l^{-1}$) in the water column during flood (left) and ebb tide (right).*

8 DISCUSSION AND CONCLUSION

The Bahía Blanca estuary is the most important deep water harbour system of Argentina and to maintain its navigability it needs periodical dredging due to the permanent erosion by the action of tides enhanced by the action of wind and waves. In order to evaluate the possible impacts due to dredging and the dumping of dredge spoils at the disposal areas, it is very important to gain knowledge of system dynamics. In an estuary with a high ratio of intertidal areas, hydrodynamics would be very sensitive to water level variations. Due to this importance during the present study, the sources of variations in water level have been identified, analysed and quantified for the Bahía Blanca estuary.

The main contributors to the hydrodynamics system are the astronomical tides, explaining more than 85% of the water level variation. However, other atmospheric phenomena, such as pressure and wind, can provoke modifications in water elevations.

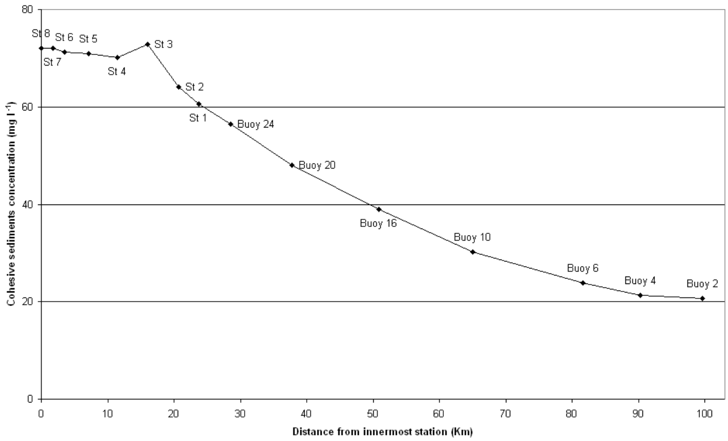

FIGURE 13: *Average values for four years simulation in each of the model monitored stations along the Principal Channel main axis.*

FIGURE 14: *Sediment height difference between the last three years simulated with an erosion rate of* 10^{-5} $kg\ m^{-2}\ s^{-1}$.

The circulation in the Bahía Blanca estuary has been described as dominated by a semidiurnal, quasi-stationary tidal wave (Perillo and Piccolo 1991) and this makes it a tide-governed estuary. The tidal wave travels from the South and enters each of the bays by the Southern margin. The particular geomorphology of the Bahía Blanca estuary produces the increase of the overtides components which produce the tidal amplitude increase. As an effect, tides evolve from the outer ocean from flood dominated to ebb dominated increasing this dominance with the distance from the mouth. As a result, higher current speeds are found during ebb for each of the bays. Maximum values during ebb tide are found near the mouth of the estuary where waters collected by the system of channels meet, and along the main axis of the channel. During flood, maximum values are found near the mouth of the bays where water entering piles up. On the other hand, minimum values are found on the elevated intertidal areas.

One of the great advantages of using modelling tools in integrated coastal zone management is that datasets collected at different periods and sampling intervals can all be integrated into one tool to reproduce periodical phenomena, i.e. tides, currents. Also, modelling allows isolating and discriminating single processes from all the available information. Data modify the conditions of the model, and model results can provide a feedback for the accuracy and consistency of the data. For these reasons, models should not be regarded as data ”sink” tools only, as they are also able to test the available data consistence.

In the Bahía Blanca case, modelling served to confirm an inconsistency on the MSL used for the TM station. In this case, the simultaneous use of observed water levels and currents, although they were not from the same period, and the posterior mathematical analysis and validation through a hydrodynamical model resulted in the conclusion that mean sea levels for the tidal gauge stations needed (re)calibration. This is one of the major outcomes of the present work.

The model hydrodynamics indicates a residual circulation into the head of the tidal channel of dissolved properties; on the contrary, tidal channels export particulated matter. This fact is relevant for water quality and sediment studies. Dissolved properties would tend to accumulate in the innermost areas of the channels while sediments would be washed away. The latter phenomena would be less in the Principal Channel, due to the recirculation pattern found at its mouth. These features are also in agreement with the lagrangian test performed where tracers located in the innermost area of the Principal Channel present longer residence times.

Although more detailed work should be performed on this matter, the model appears to respond to wind forcing as the literature has described. Water levels increase in the inner areas when winds blow from the South-Southeast sector and decrease on average when they blow from the opposite direction. This effect is more intense when the wind blows along the bays main axis.

In this chapter, a conceptual sediment model has been established, in which due to the absence of external sources of sediments, the sediments found in the water column have their origin in the Bahía Blanca estuarine system. Tidal currents transport the eroded sediments from the channel flanks into the Principal channels (Perillo and Sequeira 1989). These sediments would be transported by the ebb currents and exported into the adjacent coastal shelf forming a sediment front along the Bahía Blanca coast along with the exported sediment from the other bays of the Bahía Blanca estuarine system.

Some assumptions were made in order to achieve the present results. The intertidal areas were considered the main source of sediments into the system with a higher resuspension potential of sediments than in the subtidal areas. The subtidal areas were a residual source of more compacted sediments that would need higher stresses to be eroded, while in the intertidal areas, the stress needed would be lesser and constant with sediment depth. This 3-dimensional geomorphological model was forced by a 2-dimensional vertically averaged hydrodynamical model with a resolution of 0.01°.

The results obtained through the simulations agree in general with the conceptual model and the analysed data. The model has been validated mainly in the Principal Channel of the Bahía Blanca due to the lack of data for the rest of the system. Cohesive sediment concentrations decrease steadily from the innermost area of the channel to the open ocean. The decrease in sediment concentration is mainly due to the higher proportion of intertidal areas in the inner areas of the system while the proportion of these areas is lesser mouthwards. In its mouth, the Principal Channel, Falsa Bay and Verde Bay formed a sediment and hydrodynamic front that is more intense during ebb tides and enhanced during spring tides which indicates net erosion in the system.

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EVOLUTION OF SALINITY AND TEMPERATURE IN BAHÍA BLANCA ESTUARY, ARGENTINA

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1 INTRODUCTION

The main driving forces of circulation in a large number of estuaries are river flow at the head of the estuary and changes in the sea level at the mouth of the estuary, which in turn determine the distribution of water properties like salinity and temperature, as well as the distribution of any other tracer. In Bahía Blanca estuary there are other forcing actions like wind stress and surface heat balance and they respond differently to these forcing actions. Bahía Blanca estuary is in Argentina and the most dynamic in terms of physical and biogeochemical processes. It is a very important ecosystem in the region where it is situated due to the intense human activity in its waters and along its margins. In the last 30-40 years the estuary was studied mainly from a biological and geological point of view.

There are not many studies or publications regarding the hydrology and physical processes of the estuary. Despite this fact, a prior physical process study (Perillo et al. 1987, Piccolo and Perillo 1990) reveals some of the main features of Bahía Blanca. Observations of water level, current velocity, temperature and salinity properties were made at several stations in the estuary during the last years. It was determined that the type of tide at the mouth of the estuary is semidiurnal and it was observed that the astronomical tide is the main forcing driving water circulation in Bahía Blanca estuary. The tidal wave propagation in the study area has the characteristics of a damped progressive wave. According to this study Bahía Blanca estuary was found to be a vertically homogeneous estuary. Nevertheless, some channels may exhibit the characteristics of a partially mixed estuary, depending on the freshwater input.

The importance of the freshwater sources in the estuary dynamics and of its seasonal effects remained a subject of study until now. The estuary has also been studied through numerical modelling. These studies were performed to investigate topics such as the tidal propagation in the estuary (Palma 1995, Etala 2000, Pierini 2007), the Lagrangian transport of particles (Pierini 2007) and studies of residual suspended sediment (Perez and Perillo 1998). The estuary has several freshwater sources, Sauce Grande River being the most important and Napostá Creek. This last freshwater source is connected to the Atlantic through a channel located in the central area of the estuary, Principal channel. This channel is only partially known in terms of temperature and salinity processes, even though it is ideal to perform this kind of studies in Bahía Blanca estuary. The description of salinity patterns provides a basis for predicting the behaviour of other soluble substances, being suitable to study since the salt is a natural tracer.

This study takes a first step to better comprehend the interaction between seawater, freshwater and temperature within this system and to the understanding of its dynamics. These studies require a large amount of field data, of salinity and water temperature, as well as the implementation of numerical models that combine hydrodynamic and transport modules. This contribution uses two complementary and interconnected approaches to study the water temperature and salinity patterns in Bahía Blanca estuary, combining field measurements with numerical modelling results.

The main purpose of this chapter is to determine the horizontal patterns of salt and water temperature in two stations along the Principal channel and to evaluate the importance of the main forcing mechanisms: tides and incoming river flow. To achieve this objective, a short description of the first annual observational program of the hydrological properties in Bahía Blanca estuary will be given. Salinity and water temperature data were measured in (Cuatreros Harbour) the harbour of Puerto Cuatreros and Ingeniero White Harbour along the Principal Channel in the inner part of estuary. These measurements have been taken weekly since 1974 to the present day. An estimation of the river flow for the survey periods was also performed. Results will be presented and discussed, that reflect the influence of superficial heat transfer and tidal forcing on the spatial and temporal distribution of the referred hydrological properties.

Another aim of this work is to implement a transport model in a 2D mode for the entire area of the estuary, with a closer look at its inner area. The model used was Mohid-Water Modelling System, a finite volume model that combines hydrodynamic and transport modules, describing its assessment through calibration and validation against several different data sets.

Due to the estuary´s complex geometry and the number of calibration stations used, this goal constitutes a very challenging task. The model was calibrated using as a first approach a qualitative comparison of the temporal evolution of sea surface elevation (SSE) data measured in 1999/2003 at three locations. When a good match is obtained for all stations, the model's accuracy is evaluated through the determination of the root mean square (RMS) error and also through the comparison between amplitude and phase of the main tidal constituents determined from harmonic analysis of the observed and computed data. The validation procedure was performed using two independent data sets, which includes observations of current velocities (1997 data) and SSE values.

A mathematical model is by definition an attempt to approximate and reproduce real phenomena. The approximations and parameterizations used for the synthesis of the model lead to discrepancies and deviations of model results from observations. The optimization of the model performance is a complicated task and before using a model for operational applications the model should be verified, calibrated and validated. However, there is no widely accepted procedure for carrying out these tasks. Model calibration and validation appears in various forms, dependent on data availability, characteristics of water body, and most of all, the perceptions and opinions of modellers (Hsu et al. 1999). The aim of this paper is to present the calibration and validation of the salt and the temperature for Bahía Blanca estuary.

2 STUDY AREA

A map of the Bahía Blanca estuary location is shown in Figure 1. Principal channel is located in the middle of complex area of Bahía Blanca estuary, a mesotidal and shallow (mean depth of about 1 m relative to the local datum) estuary situated in the southeast of the Buenos Aires province, in Argentina. The study area from the mouth of the estuary to the head is approximately 79 km long, has an average width of about 200 m and a mean depth, along its longitudinal axis, of about 10 m. The tides are semidiurnal, with M2 as the most important constituent, representing more than 90% of the tidal energy. The estimated tidal prism for the estuary head at extreme neap and extreme spring is $6{,}5\,\mathrm{x}10^{9}$ m^3 and $1{,}0\,\mathrm{x}10^{10}$ m^3, respectively, with an average value of $8{,}4\,\mathrm{x}10^{9}$ m^3. The total estimated freshwater input for the estuary is very small (about $4{,}4\,\mathrm{x}10^{4}$ m^3 during a complete tidal cycle) when compared to the tidal prism both at the mouth or at the beginning of Principal channel.

In spite of the small contribution of the rivers in terms of water input, when compared to the tidal prism, they may have a long-term influence on the residual transport. Previous studies by Perillo and Piccolo (1991) and Pierini (2007) revealed that the tide is strongly distorted as it progresses upstream from the mouth towards the end of the channels of the estuary, due to changes in channel's geometry and bathymetry. The general characteristics of the tidal wave are those of a damped progressive wave. Nevertheless, in shallow areas the tidal wave assumes the main characteristics of a standing wave. From a dynamical point of view, Principal Channel may be considered the most important area of the estuary, because the strongest currents are observed here, reaching values higher than 2 $\mathrm{m\,s^{-1}}$. The estuary's other channels are mainly shallow and tidal flat areas, contributing to a strong damping of currents.

3 MODEL CONFIGURATION

The MOHID - Water Modelling System (Leitão et al., ReferencesMOHID) is a barotropic finite volume model, designed for coastal and estuarine shallow water applications, like Bahía Blanca estuary where flow over complex topography, flooding and drying of intertidal areas, changing mixing conditions are all important. MOHID allows an integrated modelling approach of physical and biogeochemical processes. A complete description of the model's physics can be found in several works by Martins et al. (2001) or Leitao (2003).

MOHID has been configured for the Bahía Blanca estuary. The bathymetry of the estuary is extracted from data obtained by the Bahía Blanca Port Consortium. More recent bathymetric data, obtained from recent dredging operations in several channels and remote sensor images, were also used (Pierini 2007). Bathymetry is probably the most important among many factors that affect the flow properties in shallow systems like Bahía Blanca estuary. Previous modelling experience indicates that bathymetry controls the spatial variability of current magnitude and direction. Thus, an accurate bathymetric representation is one of the most important and fundamental requirements for successful modelling (Pierini 2007). This is particularly true

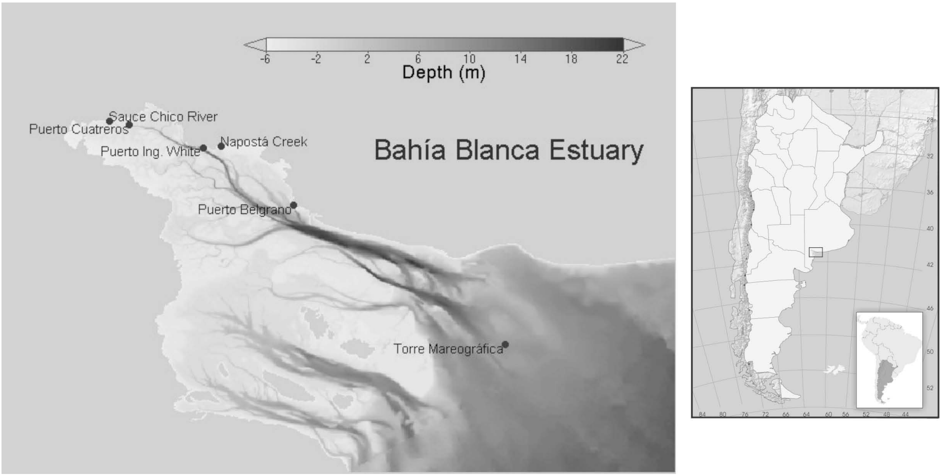

FIGURE 1: *Bathymetry, location of the tidal gauges and discharges within the estuary.*

for the Bahía Blanca estuary, where the bathymetry is very complex. Furthermore, the grid must accurately represent the bathymetric characteristics of the estuary and guarantee model stability.

The task of defining the grid-depth relations for the computational grid points can be extremely time consuming and tedious. The flow simulation in this complex domain requires the use of a high resolution grid. The model grid must be sufficiently refined to resolve the essential features of the depth and the geometry variations, but as the grid resolution is refined the total number of grid points and the computation time both increase geometrically. For this domain the ideal cell dimension would be around 50m, but the compromise solution was to develop a grid with dimensions of 200m, resulting in 119 cells in the x-direction (eastward) and 68 cells in the y-direction (northward) (Figure 1). In this case the narrowest channels had their width exaggerated, but by decreasing their depth it was guaranteed that they maintained their water volume.

Changes in the bathymetry may be expected due to sediment dynamics and to the dredging operations performed last years. The domain was designed to resolve the estuary's dynamics and not the dynamics of the near coastal ocean. Bahía Blanca estuary is an estuary where the tidal prism spreads toward the main channels and extensive tidal flats. This causes a decrease of the tidal prism through the channel affecting the dynamics of the whole study area. The Rio Sauce Chico river and Arroyo Napostá creek (Figure 1) supply fresh-water to the system. Thus, in order to simulate this inflow/outflow, the model uses as boundary conditions water flow, salinity and water temperature time series computed using a two-dimensional (depth-integrated) application of the model for all the study area.

The 2D application uses a variable spatial step Cartesian grid with higher resolution in the study area, calculating time-varying discharge, salinity and water temperature time series. The model was forced using tides (ocean open boundary) and the landward boundary was forced using river flow values typical for each run. At the oceanic and landward boundaries, the water temperature and salinity are fixed in each simulation, with typical values for each period. At the surface, heat fluxes were imposed. The model uses the heat fluxes parameterizations described by Chappra (1997). Time step is 60 seconds and the horizontal eddy viscosity is 50 $m^2\ s^{-1}$. For the transport model the initial conditions are salinity and water temperature fields, obtained by interpolation of data collected in the study area. A value of 50 $m^2\ s^{-1}$ was adopted for both salt and heat diffusion coefficients. The two-dimensional application is fully described in Campuzano et al. (2008). Coefficients of horizontal viscosity and diffusivity are set to 50 $m^2\ s^{-1}$. Initial conditions for the hydrodynamic model are null free surface gradients and null velocity in all grid points. Initial conditions for the transport model are mean monthly values of salinity and water temperature (typical values for each run).

As a first approach, a model covering the Argentina Continental Shelves to whole estuary and surroundings ("grandfather model", Figure 2) and forced with tidal components provided by the generic tidal FES95.2 model (Le Provost et al. 1998). In order to simulate water level at the boundaries a network of "imaginary" tidal gauges fringing the model boundary is set as shown in Figure 2 for the primary model. The model automatically performs triangulations between the different "imaginary" tidal gauges to impose the tide on every cell of the boundary. At the landward boundary freshwater inflow was imposed water inflow/outflow, salinity and temperature fluxes were also prescribed. On the offshore open boundary and at the river boundary constant values of salinity and water temperature were prescribed (S_{sea} = 35 and S_{river} = 0.5, water temperature varies from run to run). The hydrodynamic was spun up from rest over 4 days ($\sim$8 tidal cycles). This is considered a fair adjustment period for the hydrodynamics. The spin-up period is not included in the results and the initial state of a run refers to the end of the spin-up period.

3.1 Temperature and Salt transport models

In most estuaries with a significant freshwater discharge, salinity may serve as an ideal natural tracer for calibration of transport processes. In these environments the tidal current, the freshwater discharge, the density circulation, as well as the turbulent mixing processes affect salinity distribution. Therefore, the salinity distribution reflects the combined results of all these processes, and in turn it controls the density circulation and modifies the transport processes. Assuming that the barotropic flows (tidal and freshwater flows) have been calibrated and validated, the procedure to calibrate the salinity transport model is to match the observed and the computed salinity time series in Puerto Cuatreros and Puerto Ingeniero White. In this calibration procedure, the analysis of predicted distributions of salinity is used to guide the adjustment of calibration constants through comparison with the typical horizontal salinity distribution observed in Bahía Blanca estuary. Once the salinity transport model is considered calibrated, the transport processes may be considered well represented by the model. Therefore, the calibration of the heat transport model is related only to the heat and radiative fluxes parameterization.

A set of salinity and water temperature data measured between 1/4/2003 and 1/8/2004 is available for comparison with model results. These data include long time series of salinity and water temperature measured weekly at Puerto Cuatreros and Puerto Ingeniero White. The freshwater continuous inflows through the upstream boundaries are not known. The imposed flowrates at the rivers boundaries resulting from mean monthly value procedure were the following (Table 1)

The freshwater was specified with salinity equal to 0.5, freshwater temperature for the Napostá Creek and Sauce Chico River as 10 °C. In the calibration of the heat transport model the sensible and the latent heat fluxes coefficients are also used as calibration parameters and were considered constant in all the simulations. It was found that the agreement between the predicted and the observed temperature is good. Figures 3 and 4 show the comparison between the predicted and observed salinity and water temperature, respectively, for 2 stations (Puerto Cuatreros and Puerto Ingeniero White). The patterns observed are essentially dependent on the tidal transport. The RMS values were computed for each station, and are presented in each plot.

The agreement between the predicted and the observed salinity may be considered good for all the stations, the salinity time evolution and amplitude variation being well represented by the model. The maximum absolute RMS value was determined for station Puerto Cuatreros, with a value of 0.125, which represents about 12% of the local salinity amplitude. The RMS value for the other station is also around 10% of the local salinity amplitude. In general there is a good agreement between the predicted and the observed water temperature values (Figure 3 and 4). The RMS values are typically about 21% of the local water temperature amplitude. From these results it may be considered that the transport processes in the Bahía Blanca estuary are well represented by the transport models. And the heat exchanges between the atmosphere and the water surface are very important processes too.

4 CONCLUSIONS

The transport models of heat and salt were successfully implemented for the Bahía Blanca estuary. Results show that the calibration of the models was successfully carried out, showing a good agreement between measurements and model results. The validation tests showed that the models can reproduce an independent observed data set. The developed and applied models to the study area appear to use an adequate bathymetry. However, differences still do exist, and they might be the result of several factors, including: inaccurate definition of the bathymetry in the model for that region; very narrow channels not well resolved by the model horizontal grid; and uncertainties in the field data. The results show that the models can accurately reproduce the barotropic flows and simulate adequately the salt and heat transport in Bahía Blanca estuary, even in conditions of freshwater inputs from the rivers and heat transfer between the atmosphere and the water surface. The models can, therefore, be considered as a new important tool for future studies of Bahía Blanca estuary dynamics and water quality.

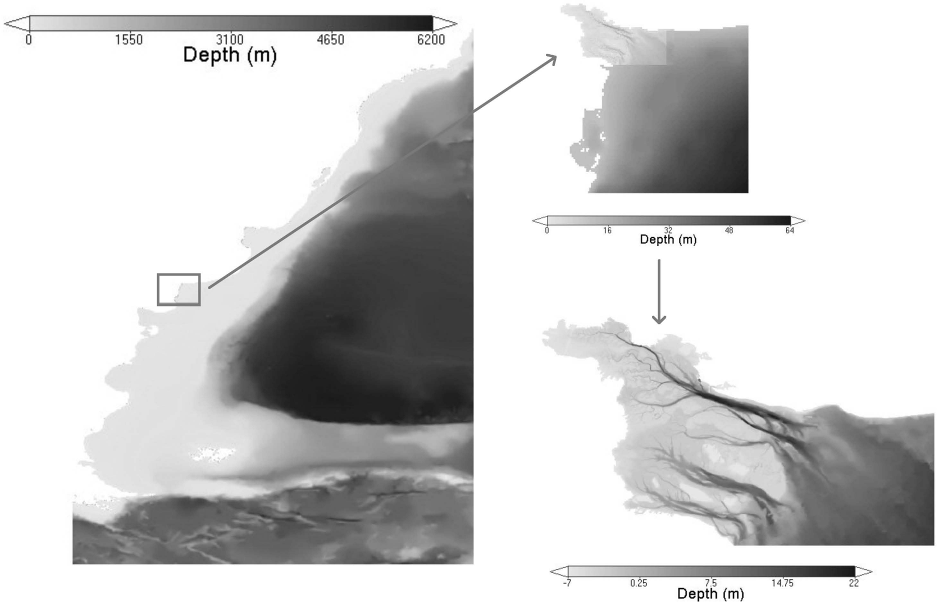

FIGURE 2: *Nested models used for Bahía Blanca Model. On the left is the primary model (Argentina model), on the right above is the secondary model (El Rincon model) and lower right is the tertiary model (Bahía Blanca model).*

TABLE 1: *Monthly mean freshwater flows*

Month	Arroyo Napostá creek (m^3s^{-1})	Río Sauce Chico river (m^3s^{-1})
January	2.25	4.35
February	1.80	4.40
March	2.20	6.50
April	1.85	6.55
May	2.70	6.40
June	3.60	5.15
July	3.60	4.80
August	2.00	4.60
September	2.55	6.80
October	3.60	8.45
November	3.15	6.25
December	2.80	5.30
Average	2.68	5.80

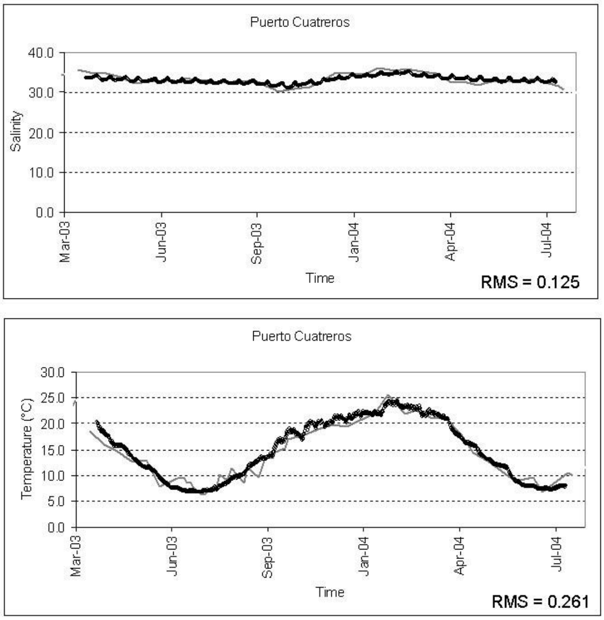

FIGURE 3: *Puerto Cuatreros salinity and temperature representation: observed (grey line) and modelled (circles).*

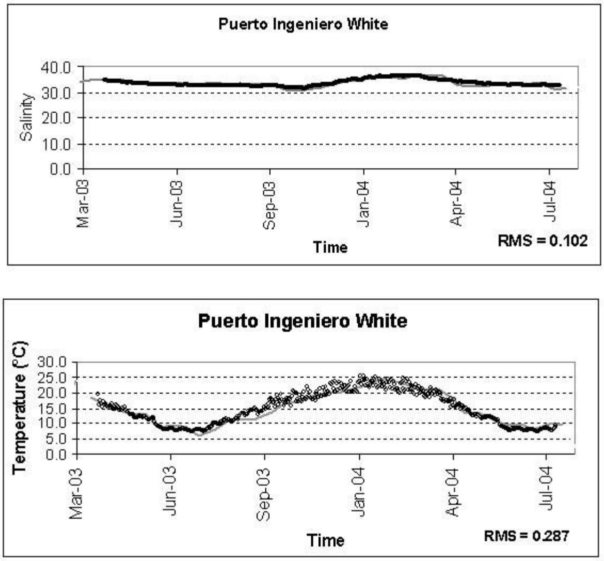

FIGURE 4: *Puerto Ingeniero White salinity and temperature representation: observed (grey line) and modelled (circles)*

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THE APPLICATION OF MOHID TO ASSESS THE POTENTIAL EFFECT OF SEWAGE DISCHARGE SYSTEM AT BAHÍA BLANCA ESTUARY (ARGENTINA)

J.O. PIERINI, F. CAMPUZANO, J. MARCOVECCHIO AND G.M.E. PERILLO

1 INTRODUCTION

The Bahía Blanca urban wastewater treatment plant (UWWTP) discharges the wastewater from the city into the Bahía Blanca Estuary through a tidal channel located 4 km south from the closest city, Ingeniero White, and 1.7 km south from the mouth of the Napostá Creek. There are four UWWTPs discharging in the Bahía Blanca Estuary, with the highest flow from the Bahía Blanca UWWTP at 65000 $m^3\ d^{-1}$, corresponding to 77% of the total (Heffner et al. 2003). Wastewater is an important source of freshwater in the Bahía Blanca estuary comprising 23.3% of the overall freshwater, with contributions of a similar magnitude as the freshwater inputs from the Napostá Grande Creek, 86400 $m^3\ d^{-1}$ (Carrica 1998) and the Sauce Chico creek, 164000 $m^3\ d^{-1}$.

Sewerage systems in Bahía Blanca city are designed with overflow structures that discharge into local waterways when the capacity of the system is exceeded. Overflows occur when the hydraulic capacity of the system is exceeded due to heavy rains. During dry periods, overflow can also take place due to blockages or pump failures. Aguas Bonaerenses Sociedad Anónima (ABSA) manages the main network. The aim of this research was to analyze the potential risks of sewage in the estuary and the installation of a new UWWTP on the coast in the inner Bahía Blanca estuary near bathing waters, i.e. the Maldonado Municipal pool.

2 STUDY APPROACH

The problems associated with wastewater discharges in estuarine and riverine areas are of growing engineering interest, since the increase of pollutant loads in the last decades may lead to serious environmental impacts. In order to quantify river and estuary pollution problems it is necessary to predict the concentration distribution of the pollutant loads in the area under study under different conditions. Numerical models constitute a powerful tool for the study of diffusion and dispersion of pollutants. Model simulations reproduced two different sewage discharges (West and East UWWTPs) in the inner part of Bahía Blanca estuary (Figure 1). The actual UWWTP discharges into a tidal channel, located 5 km away from Puerto Ingeniero White (Latitude: 38°48'34.51" S Longitude: 62°13'19.70" W) in the vicinity of the Napostá River mouth (Figure 1). Official data made available by ABSA estimate an UWWTP discharge of 0.575 $m^3\ s^{-1}$ on average, although on the other hand Piccolo and Perillo (1990) estimated a discharge of 10 $m^3\ s^{-1}$ and sometimes higher than this value. Maldonado Channel is a waterway mainly used when the Napostá River reaches flows over 30 $m^3\ s^{-1}$, preventing floodings by deviation of part of the water. This channel is used during torrential rains, maximum flow under average conditions is 3.6 $m^3\ s^{-1}$.

Samples were collected from five sites along the principal channel of the Bahía Blanca Estuary. Pollutants were characterized in the sewage of Bahía Blanca to determine its physical, chemical, toxicological and microbiological character (MBB 2004). The analyses included: organic and inorganic nutrients, microbial faecal indicators and toxicants. Although the project was not a study of the stormwater impacts, we looked for the pollutants common to both to distinguish the impacts of the overflow during rain events. During the ECOMANAGE Project we simulated the potential impact of sewage discharges and the installation of a new primary treatment plant in the inner estuary. But for the new location of a Treatment Plant there are no data or knowledge of the environmental impact. The next stage of the ECOMANAGE project will be to determine the environmental water-quality of Bahía Blanca estuary in relation to the pollutants found in the untreated sewage. We found that the most cost-effective approach was to test samples for faecal coliform contamination during the event at all locations and times. Faecal coliform (thermo-tolerant coliform) tests were completed in 2003 to identify areas of greatest contamination. The more expensive and time-consuming analyses were then used as confirmation tests for the presence of human faecal contamination.

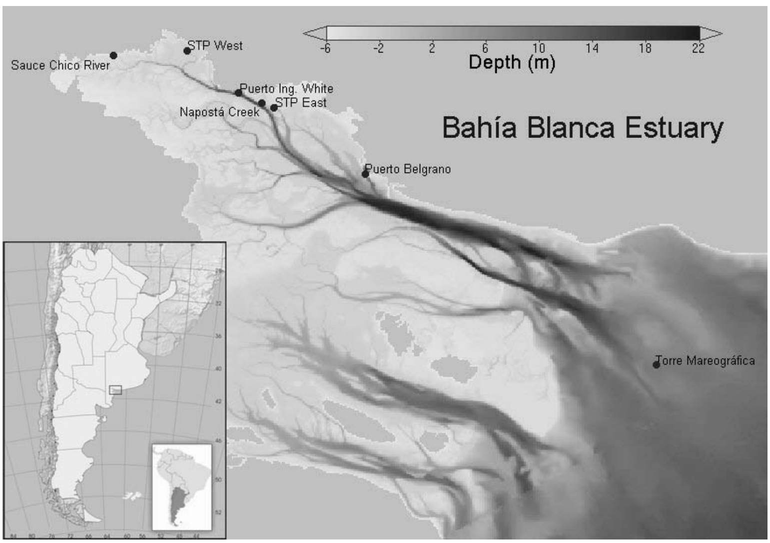

FIGURE 1: *Location of the Sewage Treatment Plant (West and East STP), water discharge and tidal gauges within Bahía Blanca estuary.*

Knowing the environmental water-quality in the recipïent, we were able to compare environmental impact caused by the two treatment plants. Human faecal contamination in the waterway was used to assess the public health hazard and potential risk assessment based on pathogenic indicators (*Escherichia coli*). The *Escherichia coli* abundance was then used in the risk assessment model to determine potential human public health risks over the inner part of estuary. Risk assessment requires knowledge of the extent of human exposure, such as recreational waters (Maldonado pools). While we observed children as well as complete families swimming within the study area, a quantitative assessment of the use (exposure to hazard) of the waterway was beyond the scope of this study.

This study took into account the two treatment plants focusing on the potential impacts and hazards over the inner part of Bahía Blanca estuary as well as the feasibility of using a hydrodynamic model to interpret the pathogenic contamination due to sewage inputs.

3 MATERIAL AND METHODS

3.1 Study Area

The selected area (Figure 1) is located in the southwest of Buenos Aires Province (Argentina). This area is a hydrographic basin of 19000 km^2. It is a mesotidal estuary characterized by a complex tidal channel network separated by islands, wide low marshes and tidal flats. Furthermore, the tidal flats are covered by dense nets of minor tidal channels and creeks. All major channels open to the inner continental shelf form ebb deltas, some of which are largely modified by present day dynamic conditions. The study is concentrated on a sector of the main channel (Figure 1) located in the northern part of the estuary. This channel is approximately 70 km long and 3 km wide in its lower reaches. The average depth of the channel is 10 m and the circulation is dominated by a quasi stationary tidal wave (Perillo et al. 2004). Mean tidal range varies from 2 m at an offshore Oceanographic Tower to 3.5 m at Puerto Ingeniero White (Figure 1). Prevalent winds for the area are mainly from the Northwest and North. These continental winds are strong and flow approximately parallel to the axis of the channel, thus affecting the circulation and vertical mixing in the estuary. The main pathogenic inputs were two sewage treatment plants, located in the northern part of the estuary, equipped with activated sludge treatments (Figure 1). Treated wastewater was monthly sampled at the five sampling stations of the outfall near Napostá Creek. These sites were selected based on their levels of bacterial contamination or hydrodynamic circulation. Water samples were stored at 4 ^{o}C during shipment and analysed within 36 h.

3.2 Hydrodynamic Model

The MOHID hydrodynamic model (Coelho et al. 2002, Santos et al. 2002) was used to force a model covering the study site (from the coordinates (-61.41W, -39.38S) to the inner estuary). In order to study the different hydrodynamic processes involved in Bahía Blanca it was needed to adopt different spatial scales. As a result, the model of Bahía Blanca consists of a set of three nested grids: (1) Level Argentina: covers the Atlantic coast of South America (-70W -40W, -60S -20S) with a resolution of 0.06^{o}; (2) Level Rincón: a secondary model covering the area known as El Rincón with 0.02^{o} resolution (-62.6W -60.5W, -42.8S -38.68S); (3) Level Bahía Blanca: a third submodel covering the main area of interest, the Bahía Blanca with 0.005^{o} resolution (covering from the coordinates (-61.41W, -39.38S) to the inner estuary. The main interest in developing different resolutions has been to study different process in more detail according to the available data and to optimize the runtime of the model. Bathymetric data used for composing the different model domains come mainly from two sources: (i) from CGPBB (Consorcio de Gestión del Puerto de Bahía Blanca) with a waterline obtained from evaluation of 6 sets of Landsat 5 and Landsat 7 data resulting in a high density bathymetry (50 m x 50 m) for the Level Bahía Blanca (Pierini 2007); and, (ii) from the GEBCO digital atlas, a one minute global bathymetric grid database (IOC, IHO and BODC, 2003) for the other areas (Figure 2). The Argentina model was forced with tidal components provided by the generic tidal FES95.2 model (Le Provost et al. 1998). In order to simulate water level at the boundaries a network of ”imaginary” tidal gauges fringing the model boundary on the primary model. The model automagically performs triangulations between the different ”imaginary” tidal gauges to impose the tide on every cell of the boundary. This model then provides boundary conditions consecutively to the nested submodels. In this study, wind and wave conditions were not considered.

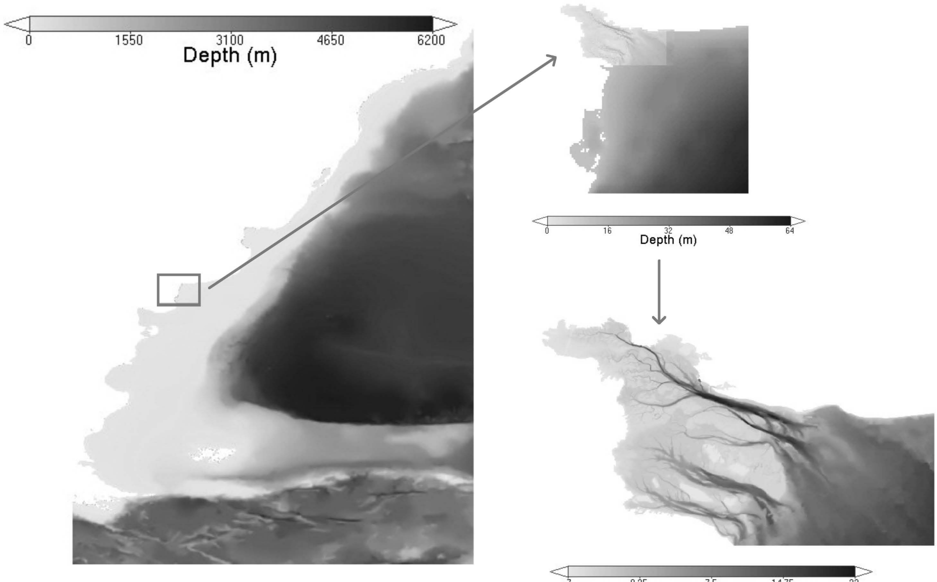

FIGURE 2: *Nested models used for Bahía Blanca Model. On the left is the primary model (Argentina model), on the right above is the secondary model (El Rincon model) and lower right is the tertiary model (Bahía Blanca model).*

3.3 Lagrangian model

The model application couples two different transport models; while the hydrodynamics and environment properties as temperature, salinity and solar radiation are calculated by a eulerian approach, the discharges and evolution are lagrangian oriented. Particle properties are affected by their environment.

4 RESULTS

The variation of *E. coli* concentration from day 0 (starting date of the outbreak) to day 90 (when the simulation was stopped) is reported in Figure 3 for all sample sites. An increase of the contamination was observed during the first days corresponding to the dispersion in the estuary. After 90 days concentrations were well established and were only dependent on the tide and decay rate. The effect of the semidiurnal tide was well modelled. The maximal concentration was observed every 12 h at high tide. The *E. coli* water concentration was highly variable, depending on the tide. In this simulation, the maximal water concentration at 17-7-2003 was equivalent to 10^5 CFU/100 ml (St 4, Figure 3 and 4), after 90 days, the spatial distribution was observed in the other samples sites. The model indicated minimal and maximal concentration of *E. coli* for the five sites. For example, at St 3, all the *E. coli* results obtained by analysis were in accordance with the model, as no sample was found to be contaminated by more than 4560 CFU/100 ml except at the discharge station (Table 1). All the results obtained fitted the values calculated with the model.

TABLE 1: Escherichia coli *observed and modelled with MOHID.*

	***E. Coli* concentration (CFU/100 ml)**					
	17 – Jun – 2003		**30 – Jul – 2003**		**10 – Sept - 2003**	
St	**Obs**	**Mod**	**Obs**	**Mod**	**Obs**	**Mod**
1	2300	2200	4400	4560	1000	1500
2	400	320	1700	1780	1400	1510
3	1700	1700	2100	2212	3600	3655
4	100000	100000	180000	184000	240000	243000
5	2500	2500	2300	2300	2000	1922

5 DISCUSSION AND CONCLUSION

For the integration of the wastewater issue in the coastal management in Bahía Blanca estuary, and more widely in all sensitive coastal ecosystems a systemic methodology based on modelling and simulations can allow the investigation of several different technical solutions which are evaluated using environmental and health criteria. Coastal management for sustainable development requires the investigation of alternative strategies which should optimize: resource preservation, water quality, technical aspects and human uses. Once the urban wastewater has been discharged it is diluted by the estuarine water. The resulting mixture flows towards the main channel of the estuary, a further dilution gradually occurring as a result of mixing with the estuarine water.

The modelling approach embedded in an ecosystem approach is designed to provide to the public a physical and environmental understanding of the processes, as well as to help decision makers to analyze the public demands and to evaluate the environmental investments. The narrow application of a modelling system and production of the results is not sufficient because of the complexity of the situations and the nature of the challenges. The use of optimizing models for systems analysis as linear programming could be used to evaluate and validate some technical solution. This form of numerical approach will only work if everyone involved in the process understands the benefits and limitations of its application and agrees with its uses.

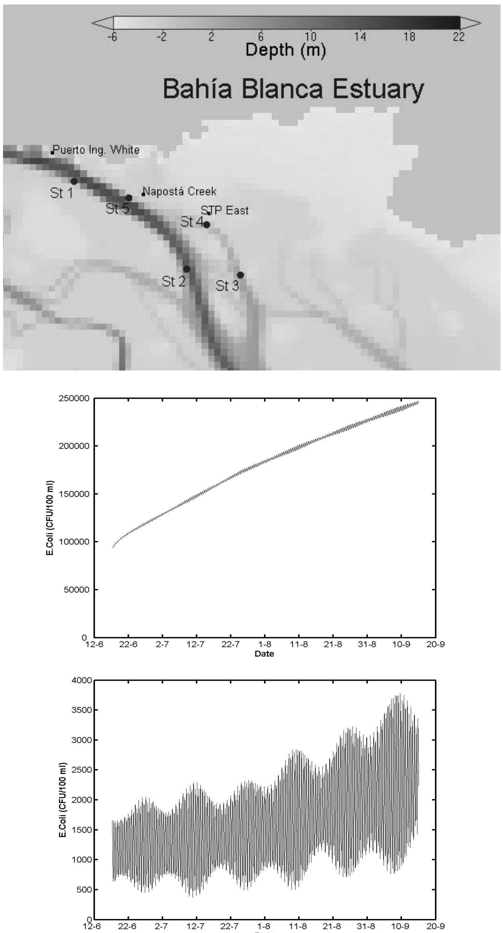

FIGURE 3: *Location of the* E. coli *samples near STP East and temporary evolution in St 4 and St 3.*

Mathematical models were previously shown to be efficient to evaluate sewage impact on bacterial contamination (Kashefipour et al. 2002, Pierini 2007). The use of mathematical modelling could help to interpret the large *Escherichia coli* variations observed along the estuary, principally from the new sewage discharge in the inner part of estuary, near Maldonado

Municipal pool. Moreover, this location requires the construction of a very efficient, sensitive and expensive tertiary process in the treatment plant. In case of malfunctioning - 10^5 CFU/100 ml - the effluent could generate a large polluted area. Consequently, the outflow must be (re)located to an always submerged point, several meters below the low tide level where the currents are continuous as in the main channel.

The model was capable of predicting complex two dimensional flow features and provided a rational tool for environmental decision making. The Estuarine impact assessment using a two-dimensional circulation and transport model has been implemented. The MOHID model described herein is expected to be a useful tool in the efforts to reduce the pollution in the Bahía Blanca estuary. Several projects may have strategic options which are not amenable to simple comparative analysis. In these cases, analytical techniques must help the decision-maker by showing the implications of the various options with the necessary multi-disciplinary perspective, rather than presenting a prescription for action. This recognizes that such complex investment decisions are inherently political. The strategy is then based on the decision maker's capacity to understand the environmental challenge and the human uses and also, to accept the results of the modelling approach which might deeply modify the initial project.

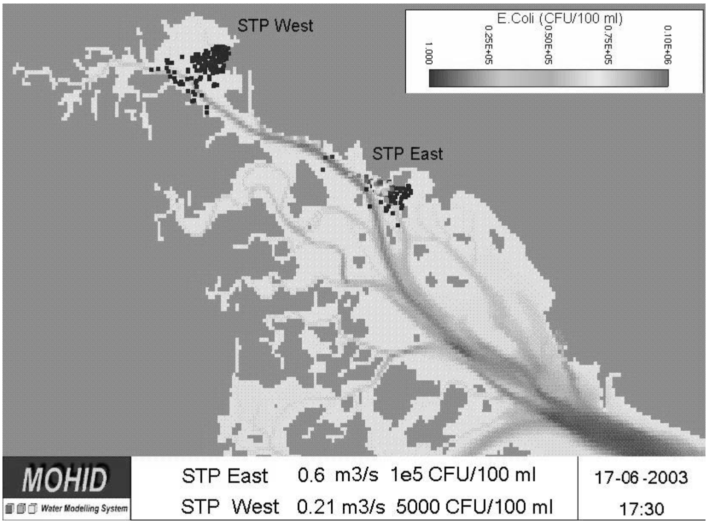

FIGURE 4: *Spatial distribution of* E. coli *with STP West and East discharges in Bahía Blanca Estuary.*

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MOHID OIL SPILL MODELLING IN COASTAL ZONES: A CASE STUDY IN BAHÍA BLANCA ESTUARY (ARGENTINA)

J.O. PIERINI, J. MARCOVECCHIO, F. CAMPUZANO AND G.M.E. PERILLO

1 INTRODUCTION

Half the world's production of crude oil is transported by sea (Clark 1992). A significant amount of oil is spilled into the sea from operational discharges, collision and grounding of tankers, well blow-outs and pipeline-breaks. Forty-eight percent of marine oil pollution is due to refined products and 29% to crude oil. Tanker accidents contribute 5% of marine pollution (Fingas 2001). Due to global economic growth, the demand for petroleum products is on the rise; hence one might also expect an increase in oil spills, especially along tanker routes. In Bahía Blanca estuary there are two monobuoys near the coast to discharge or load oil tankers. In recent years the international community has become increasingly aware of the risks due to major accidents occurring near populated and environmentally sensitive areas (i.e. the Prestige oil spill in 2002 (Galicia, Spain), the Hebei Spirit oil spill in 2007 (Taean, South Korea). There is also a growing need to ensure that health, environmental and safety issues are addressed as an integral part of social and financial development. Petroleum products that enter the marine environment have distinct effects, depending on their composition, concentration and the elements in the environment that are taken into consideration. Some effects can be related to transformations of the chemical composition of the environment and alterations in its physical properties, the destruction of the nutritional capital of the marine biomass, danger to human health, and changes in the environmental biological equilibrium.

The Bahía Blanca estuary coastal waters are noted for their heavy oil tanker traffic and, as a consequence, the risk of an oil spill occurring in these coastal waters is very real. The Presidente ILLIA oil tanker accident, in the middle part of Bahía Blanca estuary on May 19th, 1992, when the vessel due to strong SE wind spilled about 700 m^3 of crude oil over the sea water, is just one recent example of an accident with serious environmental consequences for the Bahía Blanca estuary coastal zone (Figure 1).

Numerical models are in principle able to predict the evolution and behavior of oil spilled in coastal zones, regardless of the atmospheric conditions, hence the vast interest in them. Mathematical modelling thus is a very powerful tool for management assessment after an oil spill accident, particularly for determining mitigating measures and to help monitor accident evolution. The latest information technologies have provided us with new tools that are capable of efficiently processing the great quantity of information needed to support accidental hydrocarbon spill management. These modelling tools are indispensable in the forecasting of oil slick evolution at coastal areas, as they allow measures to mitigate the negative impacts associated with hydrocarbon spills to be put in place. When integrated with other geographic information tools, the information provided by the model results can be analyzed easily and aid decision making.

Spill modelling is important to predict the trajectories and oil fate for devising suitable combating mechanisms. Hypothetical spill trajectories for different scenarios should be hindcast for the Bahía Blanca estuary, but as no real spill data exist to validate the model, results would still be uncertain. Concern exists about the 300 km^2 Bahía Blanca, Bahía Falsa and Bahía Verde natural reserve being hit by an oil spill event because of its sensitivity and ecological and economical importance (i.e. tourism).

Measurements of tides, currents, winds and hydrography close to the spill location were made available in order to validate and calibrate a hydrodynamical model for Bahía Blanca estuary. MOHID model was used to simulate trajectories (Leitão et al., Mateus and Fernandes, this volume). The present work is the first modelling study for a hypothetical oil spill in the coastal waters of Bahía Blanca estuary. The objectives of this study were: (i) to simulate hydrodynamics of the coastal waters; (ii) to generate a trajectory for a potential spill; and, (iii) to generate a training system to provide an oil spill response in Argentinian harbors and make decision support available in a short time to be applied to the contingency plans.

2 IMPACTS OF OIL SPILLS

The petroleum products that enter the marine environment have distinct effects, depending on their physical and chemical composition, and the environmental elements that are considered. The mechanisms of toxic action depend on the petroleum's characteristics. The toxicity of the various fractions of the pollutants is directly related to the distilled products, on a short-term basis, and related to the slow-action products, on a long-term basis. It is also related secondarily to the products degraded either biologically, through the action of bacteria, or through physical-chemical processes. Petroleum pollution can be detected through the modification of the environmental conditions and can be described by transformations of the chemical composition of the environment and alterations in its physical properties, destruction of the biomass nutritional budget or changes in the environmental biological equilibrium.

From the physical point of view, oil directly influences the marine environment, since gas transfer mechanisms are impaired by the presence of a pollutant layer on the surface. Self-purification processes are thus reduced. These processes can worsen by the increased oxygen consumption by growing micro-organisms, depending on the quantity of biodegradable organic matter present. This oxygen deficit could even create conditions for anaerobic life, giving rise to the death and disappearance of certain species and permitting the fermentation of organic residues.

From a biological point of view, the environmental effects of oil are varied and complex. While some are immediately obvious, others only manifest themselves after a long period. The degree of the effect is therefore different, whether in the animal or in the plant kingdom. In the case of crude oil, the volatile components and the aromatic compounds are the most toxic. In addition to possible direct intoxication resulting from the inhalation or ingestion of petroleum products, there is an indirect risk to humans from the consumption of certain marine animals (fish, crustaceans, etc.) that have been in close contact with the oil. The pollution's noxious effects can also be felt indirectly through its environmental and economic impacts; damage to biological resources (flora and fauna), affecting biodiversity; deterioration in seawater; shoreline quality and recreational waters, with negative effects on human and economic activities.

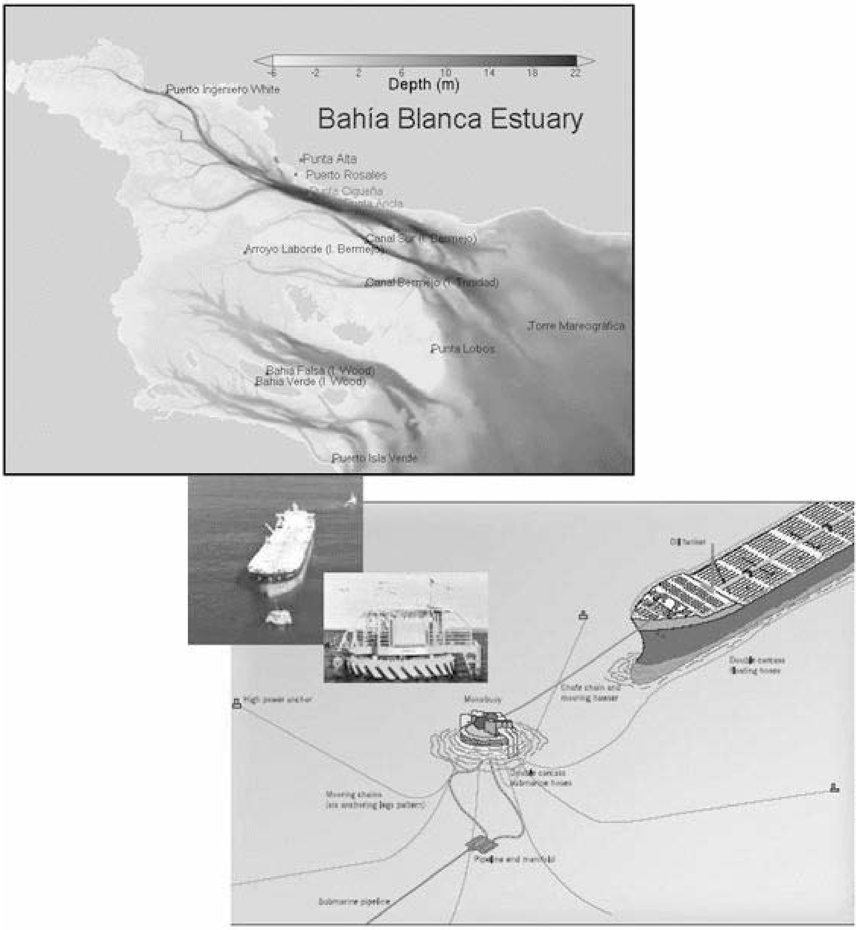

FIGURE 1: *Location of Punta Cigueña and Punta Ancla monobuoys, tidal gauges, bathymetry and single-point mooring system for Bahía Blanca estuary.*

3 CASE STUDY: MONOBUOY ACCIDENT MODELLING

In the middle of Bahía Blanca estuary, there are two monobuoys (Punta Cigueñas and Punta Ancla) where crude oil is loaded and discharged. The MOHID model was used to evaluate the fate of spilled oil under different wind conditions (NNO, SSO, NNE and SSE).

It is the first time that a simulation of these characteristics is performed in Bahía Blanca estuary and the results provide useful information to implement contingency plans and areas to guide the oil booms to another area could be considered, but require a discussion on the acceptance to sacrifice another area. The oil characteristics for the simulations were obtained from the typical NRN (Neuquen Rio Negro) oil composition, which is the type of oil loaded to the ships at the buoys (Table 1). For this experience it has been simulated an accidental discharge of 45 m^3 of oil type NRN (Table 1). The accident simulation starts at Punta Cigueña monobuoy (38°56'40.30" S and 62° 03'8.99" W) being subject to estuarine hydrodynamics and wind fields that play an important role in the oil transport processes. This accident simulation was performed to analyse their influence and also to test the surface oil slick transport model. The MOHID 2DH hydrodynamic model was used to quantify the current velocities in the coastal zone during the accident. Figure 1 shows the geographical extent, and the bottom topography (Pierini 2007).

The same hydrodynamical model configuration as applied in the case of bacterial pollution was employed. Figure 2 displays the instantaneous maximum tidal current velocities (during ebb and flood) at the accident area, produced by the calibrated and validated hydrodynamical model. The hydrodynamical model was used to calculate the tidal current velocities during the hypothetical accident period with the different wind scenarios. The wind directions (NNO, SSO, NNE, SSE) and intensities (average 10 $m\,s^{-1}$) used in the simulations were representative of the study area (Piccolo and Diez 2004). Figure 3 presents the spatial distribution of the oil with different wind directions. The wind clearly appears to have a significant influence on the oil transport process. The oil spill in Figure 3 can be seen floating along and over the coast, and out along to Principal channel. The coastal city of Punta Alta and Puerto Rosales harbor, lie near the oil spill.

- The region of Puerto Rosales Harbor, especially between the monobuoys, as well as the inner part of Bahía Blanca estuary, run a high degree of risk that increases during discharge or loading of tankers.
- The whole northern coast of the Bahía Blanca estuary, and especially the area to the east of Principal channel, including the Natural Reserve of Bahía Blanca, Bahía Falsa and Bahía Verde runs a high degree of risk.
- The Puerto Rosales area (near Punta Cigueña monobuoy) runs a high degree of risk with increasing winds from a SSE direction.
- The areas of Southern coast runs a low degree of risk which increases during periods with prevailing Northerly winds.

modelling could be taken a further step with the development of operational models that integrate meteorological models and wave models to force hydrodynamical models thus forecasting the hydrodynamical conditions during a possible event.

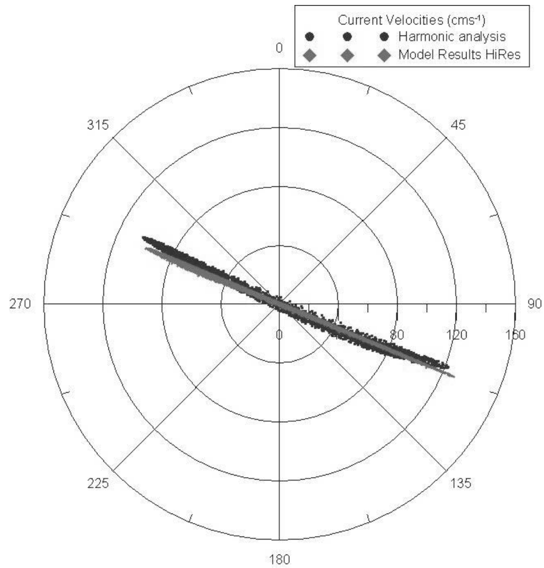

FIGURE 2: *Maximum tidal current velocities (during ebb and flood) at the accident area (Punta Cigueña).*

TABLE 1: *Properties of the oil type NRN (Source: OiltankingEbytem S.A., personal communication).*

Property	Value
Density (15 ºC)	844 kg m^{-3}
API	36.06
Water	0.05 %
Salts	35 grm^{-3}
Sulphur	0.40 gr %
Vapour Tension	2.91 lbm^{-2}
Viscosity (37.8º)	710 St
Pour Point	-9

In a precautionary approach, the MOHID model oil application for Bahía Blanca could help in defining management responses to an oil slick incident by identifying the areas where collection and protection operations would be most urgent. Moreover, in combination with socioeconomic information it could help defining a future integrated coastal zone management. It would be necessary in the future to improve the monitoring of the spills and to improve access to meteorological data in addition to an evaluation of a possible increase in the coastal observational network, to cover data gaps. Last but not least, we are greatly indebted to OiltankingEbytem S.A for their kind collaboration in this work.

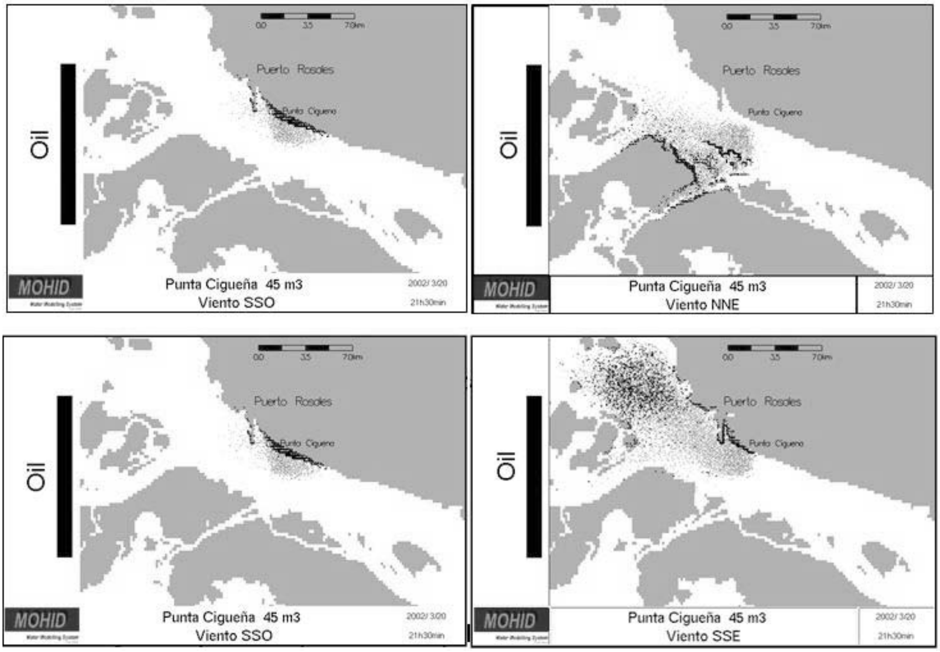

FIGURE 3: *Spatial oil spill distribution with different wind directions at the accident area (Punta Cigueña) and impact zones.*

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GROUNDWATER FLOW COMPONENTS TO THE GLOBAL ESTUARY MODEL OF THE AYSÉN FJORD

M. YARROW AND M.M. OLIVEIRA

1 INTRODUCTION

The Aysén Fjord receives freshwater mainly from the Aysén river watershed (Q $\sim$ 545 m^3 s^{-1}). A key goal of ECOMANAGE was to study the importance of the groundwater medium in the system, its recharge and the contribution of the groundwater medium to the river flow. To achieve this goal the surface flow hydrograph separation method is used in order to separate baseflow and direct runoff. Because of the high latitude and mountainous terrain of the Aysén Basin, it is important to consider the impact of snowfall and snowmelt on hydrographs and on groundwater recharge. The applied techniques have been presented in Oliveira et al. (this volume).

Two watersheds within the Aysén basin were chosen for application of the snowmelt model and the hydrograph separation method because they are gauged rivers and are close to weather stations. An additional consideration was a significant elevation range. Weather stations in the Aysén basin are generally located below 700 m elevation and in order to fully understand hydrographs in the basin it is necessary to consider the dynamics of snowfall and snowmelt at higher elevations. Each watershed was divided into three elevation bands of approximately 500 m each. This elevation range was suggested in the documentation for the Snowmelt Runoff Model (Martinec, 1994). Furthermore, each watershed was divided into zones according to the dominant gradient in annual precipitation. For the Coyhaique watershed, the annual precipitation decreases from the fluviometric station, eastward toward Argentina. The Rio Claro watershed receives higher overall precipitation, decreasing from the headwater to the fluviometric station. Using routines in ArcView GIS 3.3, the area of each of these precipitation-elevation zones was calculated, as well as an estimation of the percent areal forest coverage. Table 1 and Table 2 characterize the snowmelt zones created for each watershed.

2 SNOWMELT MODEL INFORMATION

Table 3 lists parameters and variables required by the snowmelt equations and gives information about their sources and values. These equations can be found in USACE (1998)), and are also reproduced in Yarrow and Oliveira (2006). In the watersheds modelled, the equations for forested and partially forested areas were used. In the case of zone F (>1200 m; >1500 mm annual precipitation) in the Rio Claro watershed, the rain-free, open area equation should be used. However, the use of this equation requires two additional variables that are unavailable for this area; furthermore, because the zone only misses the canopy cover threshold by 2 percent, it is deemed that the equation for partially forested areas will suffice. Forest cover data came from the spatially-explicit 'Catastro del Bosque Nativo' database generated by CONAF-

CONAMA in 1997 and provided to ECOMANAGE project by SERPLAC. Information about the physiognomy of the vegetation in the watersheds was quantified using the Table 4 and each vegetation type was weighted by its area within each zone. Canopy cover is important in the calculation of long-wave radiation, short-wave radiation, and convection melt. Table 5 shows the categories of forest density used to determine which of the simplified energy-budget equations can be used.

TABLE 1: *Snowmelt model zones based on elevation and annual precipitation for Rio Claro watershed. The percent forest cover within each zone is indicated in parentheses.*

Elevation Zone (altitude m)	**1000-1500**	**1500-2000**
1 (226-699)	A (11.5%)	B (39.3%)
2 (700-1199)	C (74%)	D (68.9%)
3 (1200-1722)		F (8.4%)

TABLE 2: *Snowmelt model zones based on elevation and annual precipitation for Coyhaique watershed. The percent forest cover within each zone is indicated in parentheses.*

Elevation Zone (altitude m)	**Precipitation (mm yr^{-1})**		
	>800	**400-800**	**<400**
1 (282-799)	A (22.4%)	B (31.2%)	C (68%)
2 (800-1249)	D (75.3%)	E (30.6%)	F (34.4%)
3 (1250-1697)	G (10.9%)	H (10.3%)	I (12.5%)

3 DETERMINATION OF THE SNOW COVERED AREA

MODIS satellite images were classified in ENVI 4.2, using the middle infrared (mir) and blue bands (the bands used for the normalized difference snow index). A maximum likelihood supervised classification routine was used; the regions of interest for the supervised classification were based on a GIS coverage from SERPLAC indicating areas of permanent snow. Three MODIS images were used, representing winter (July 12 to 28, 2005), spring (November 12-28, 2005), and summer (March 6-21, 2006) (Figure 1). A sinusoidal curve was fit to these few data points using the snow coverage for each elevation zone in the two watersheds. Using the equations thus produced, an attempt was made to use the daily proportion of each elevation zone covered by snow and a weighting factor for snowmelt. This makes intuitive sense: as the area covered by snow in each zone decreases during the spring, the production of snowmelt water should decrease even as the rate of melt increases. However, the implementation of this snow-cover-area weighting factor was problematic. On one hand, Figure 2A and B indicate that significant inter-annual variation can occur in these basins, thus the acquisition and analysis of MODIS images becomes labor-intensive. On the other hand, use of a weighting factor was shown to lead to significant snow accumulation between years in the Rio Claro watershed. In the absence of information indicating that there is net snow and ice accumulation in this watershed, it appeared judicious to not apply the snow-cover-area weighting factor until more information is available.

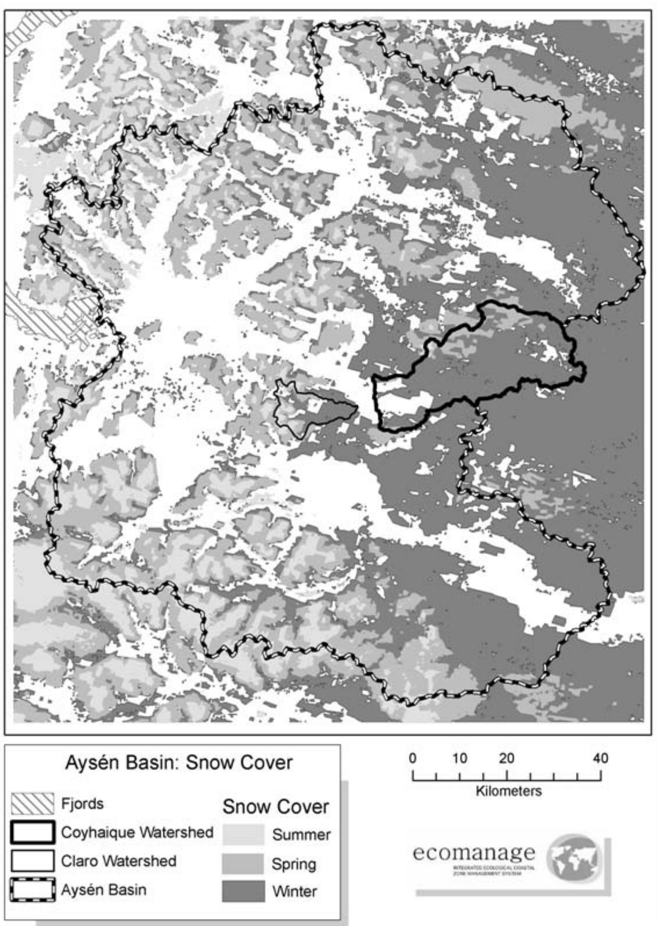

FIGURE 1: *Snow cover winter - summer 2005-2006 (From MODIS imagery).*

As recorded data were not available on snowfall, snow depth, and snow melting rates for the Aysén region, the snowmelt model could not verified. A rudimentary calibration was undertaken by (1) comparing the snow-covered area generated from the MODIS images with the presence of snowpack in the elevation bands and (2) modifying the precipitation adjustment factors so that long-term precipitation was greater than the streamflow of the given watershed. The lack of a rigorous calibration of the snowmelt model should be taken into consideration when interpreting the final results. A comparison of Figure 2A and B shows that snow accumulation should be a more important process in the Rio Claro watershed, where up to 1800 mm of precipitation fell as snow in the highest elevation band. This significant accumulation of snow in the Rio Claro watershed also meant that snow remained on the ground longer into the summer than in the Coyhaique watershed, where the snowpack had melted between mid-October and early December.

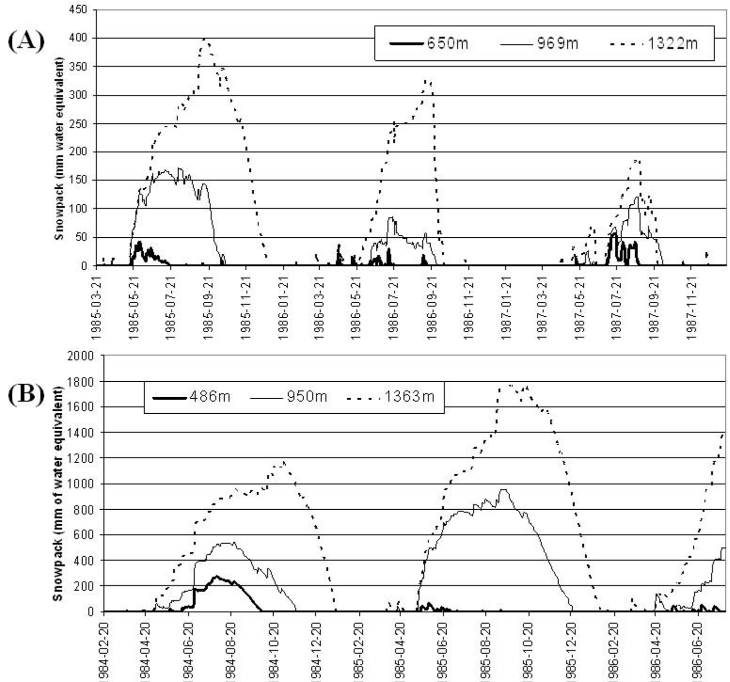

FIGURE 2: *Modeled snowpack, (A) Coyhaique watershed. (B) Rio Claro watershed. Three vertical bands are shown; values are in mm of water equivalent.*

4 CONSIDERING SNOWMELT IN THE HYDROGRAPH SEPARATION METHODOLOGY

The hydrograph separation model, DECHIDR_VB, described in Oliveira et al. (this volume), was run with the following settings: number of days from hydrograph peak until end of direct runoff (n) = 2 d; Minimum precipitation required to begin new recharge episode = 0.1 mm d^{-1}; no control over balance between precipitation and runoff; model considers precipitation in order to start a new episode. In order to allow a comparison between dynamics with and without snowmelt, two runs were made for each period, using data on Rio Claro watershed. A visual comparison of Figure 3A and B indicates that with the calculation of snowmelt, the interpretation of the hydrograph changes. In Figure 3A there is a steady contribution of snowmelt, with peaks of precipitation. It is likely that the daily snowmelt is not so constant. However, the snowmelt model uses monthly instead of daily radiation and thus differences between daily melt rates are probably less than what would be observed in the field. Nonetheless, the presence of this snowmelt water informs the hydrograph separation model that new runoff-recharge episodes should begin. Without this snowmelt signal (Figure 3B), the peaks in the hydrograph are considered to be come from baseflow - due to the fact that there is no precipitation as rain. Thus, the inclusion of snowmelt can (1) help explain the high streamflows and hydrograph peaks that occur in the absence of rain events and (2) help avoid the over estimation of baseflow when using the hydrograph separation method.

TABLE 3: *Source of parameters and variables used in snowmelt model.*

Parameter or variable	Source or References	Value (if applicable)	Considerations
Daily mean air temperature, ºC	DGA		Average of daily min and max values
Dewpoint Temperature, ºC	Relative humidity, temperature (DGA)		Relative humidity for the Coyhaique Alto station, taken from the Ñirehuao station
Precipitation, mm	DGA		Snowfall often underestimated in rain gages: if precipitation falls as snow, it is multiplied by a factor of 1.3 (Larson and Peck, 1974)
Temperature – Elevation Adjustment factor	SRM User Manual 1998	-0.65 °C/ 100m	Average temp for elevation zones is adjusted from nearest weather station
Precipitation adjustment factor	SERPLAC		Precipitation isohyets used to adjust the precipitation falling at nearest station. Number of rain days unchanged.
Wind (v)	DGA		Daily wind from Ñirehuao used for the Coyhaique watershed, average monthly values used for the R. Claro watershed
Solar radiation	Monthly average hours of sun (DGA)		Solar radiation calculated on a monthly basis using Excel worksheet provided by IST.
Snow albedo (α)	Tague & Band 2004	α = 0.85* (0.94Age^0.58)	Albedo decay function based on the number of days since last snowfall
Mean elevations	DEM (SERPLAC)		Watershed divided into 3 elevation zones; mean elevation calculated in ArcView
Forest cover (F)	Land use GIS theme (SERPLAC)		Vegetation structural classes converted to forest cover categories with the weighting factors found in Table 4
Basin convection-condensation melt factor (k)	USACE 1998	0.5	Range is 0.3 to 1. A low value was used because of high forest and mattoral cover in study watersheds.
Basin shortwave radiation melt factor (k')	USACE 1998	1	Range of parameter 0.9 to 1.1: a central value was used because aspects are evenly distributed in study watersheds

5 RESULTS FOR RIO CLARO WATERSHED

The Rio Claro watershed covers 108 km^2 to the west of the town of Coyhaique. The Rio Claro flows toward the east, emptying into the Rio Simpson. With its headwaters coming from the central Andean range, this watershed has higher pluviosity and generally steeper slopes than the larger Coyhaique watershed. For the purpose of the snowmelt model, the watershed was divided into five zones, as seen in Table 1. The available data allowed for the snowmelt model and DECHIDR_VB to be run for the following periods: May 1985 to December 1986 and December 2003 to August 2005.

The results obtained (Table 6 and Figure 4) show that the average baseflow for this period represents close to 2/3 of the total streamflow. This is perhaps a surprisingly high value for a watershed that has relatively steep slopes and thus a large potential for high runoff rates. However, it must be considered that almost 40% of this watershed is covered by forest and that, in general, the soils are quite sandy (average of 88% sand in forest soils) (Cruces et al. 1999). Furthermore, the soils in the Aysén basin are often thin and underlain by weathered bedrock or glacial deposits (Hepp 1996), thus it makes intuitive sense that much of the streamflow in the Rio Claro watershed is contributed by shallow groundwater. However, it is important that the results be interpreted with caution as there is uncertainty associated with the snowmelt model and the hydrograph separation method has the limitations described in Oliveira et al. (this volume).

The results obtained also indicate that snowmelt is an important component of runoff and baseflow. It appears that snowmelt regularly makes up between 50% and 70% of the water available for transport to the river system during the spring months. This is important because it represents a significant time lag in watershed-level hydrodynamics that could ultimately play a role in the chemistry and sediment loads of the river water. Sueker et al. (2000) indicate that during the snowmelt season in the Rocky Mountains, 42-57% of streamflow was provided by snowmelt runoff.

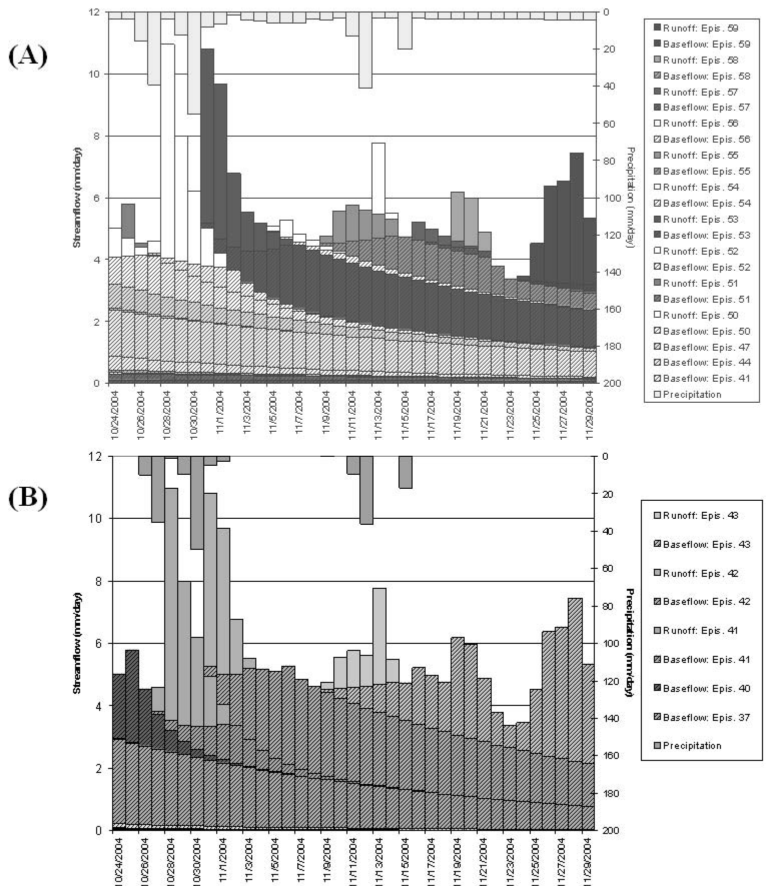

FIGURE 3: *Rio Claro watershed hydrograph separation detail (A) considering snowmelt (B) not considering snowmelt. Units: mm* d^{-1}. *The shaded areas represent baseflow discharge, while the solid colors represent surface runoff.*

6 RESULTS FOR COYHAIQUE WATERSHED

The Rio Coyhaique watershed covers 616 km^2 to the east of the town of Coyhaique, where it flows into the Rio Simpson. The eastern margin of the Coyhaique watershed borders with Argentina and includes areas classified as Patagonian steppe. The eastern part of the watershed is quite dry (average annual precipitation of 325 $mm\,yr^{-1}$) and it receives higher precipitation between May and August (winter). For the purpose of the snowmelt model, the watershed was divided into nine zones, as seen in the Table 1. The available data allowed for the snowmelt model and DECHIDR_VB to be run for the following periods: January 1985 to April 1986 and January 2000-February 2003.

Table 7 and Figure 5 show that streamflow in the Coyhaique River is comprised of 80% baseflow discharge. This is higher than the Rio Claro watershed and indicates that there is relatively little runoff. Again, it is worth mentioning the high permeability of the soils in the Aysén basin. This fact, coupled with the flatter topography and low precipitation of the Coyhaique watershed could explain this high baseflow value. The snowmelt appears to make up a relatively small portion of the streamflow, about 26%. Snowmelt still influences the hydrograph of the Coyhaique river, but this influence is more temporally limited as compared to the Rio Claro watershed. In general, it is apparent that the Coyhaique watershed exhibits a strongly seasonal pattern where monthly streamflow and snowmelt rates are highly correlated (r = 0.77, n = 32). On the other hand, the monthly precipitation as rain calculated by the snowmelt model is not significantly correlated with streamflow (r = -0.08, n = 32). This is an interesting anomaly and likely has to do with high evapotranspiration during the summer and the fact that much precipitation falls as snow during the winter.

7 CONCLUSIONS

The surface flow hydrograph separation method (requiring precipitation and streamflow data) coupled with a daily snowmelt model that accounts for the time lag and distinct pattern in which snow becomes available for runoff or groundwater recharge, was developed and applied in the Aysén basin. This model was based on available information about the topography, vegetation cover, and several meteorological and climatic variables. It is assumed that calculated baseflow is an estimation of the recharge that occurred in the system. Groundwater recharge estimates varied between the Rio Claro watershed (average baseflow is 3.65 $mm\,d^{-1}$) and the Coyhaique watershed (average baseflow is 1.15 $mm\,d^{-1}$). This reflects the steep gradient in precipitation caused by an orographic effect as prevailing winds pass across the Andes. In all likelihood, this difference also reflects differences in topography and vegetation cover. The presented results allow hydrologists (and others) to obtain a good estimate in the absence of solid, local data on snowfall and groundwater.

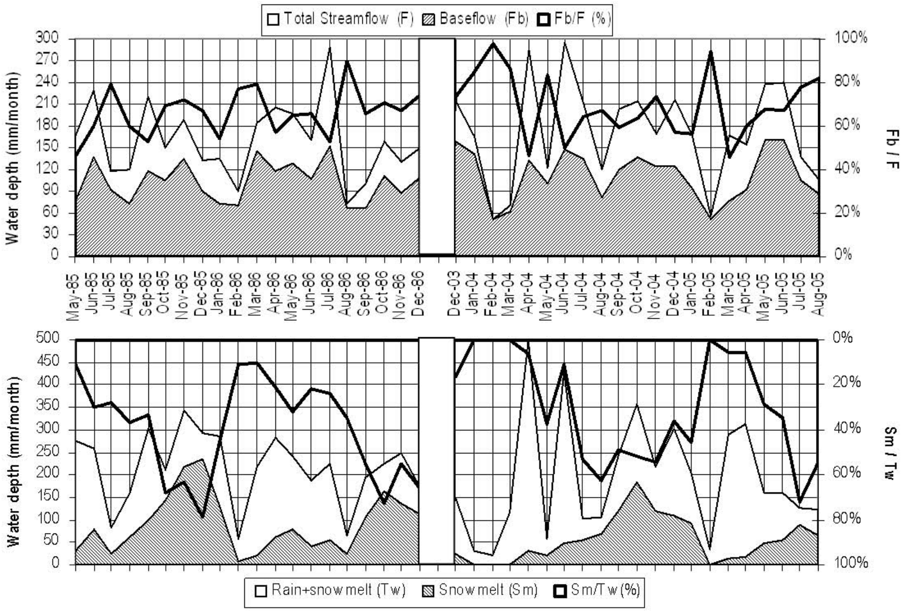

FIGURE 4: *Monthly values for total stream flow, precipitation and associated values in Rio Claro watershed.*

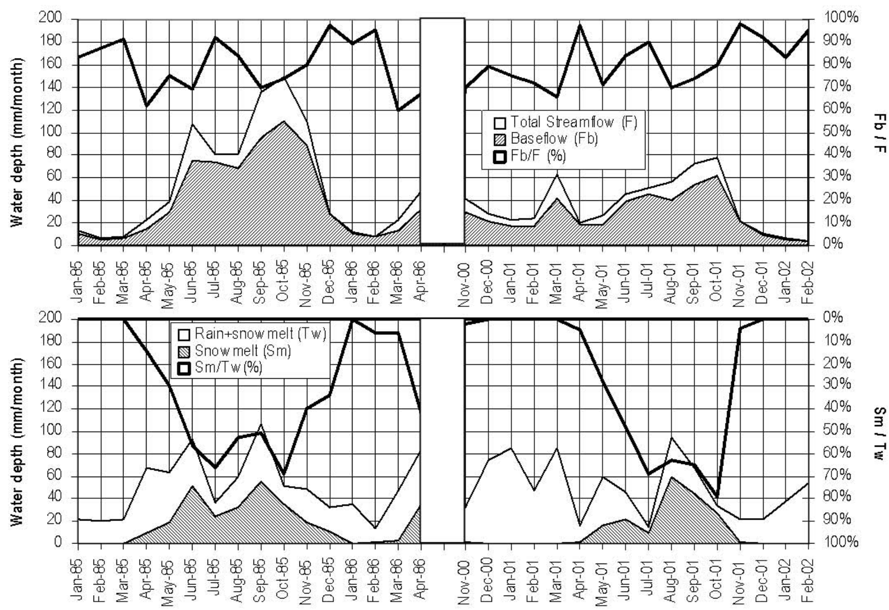

FIGURE 5: *Monthly values for total stream flow, precipitation and associated values in Coyahaique watershed.*

TABLE 4: *Weighting factors for structural vegetation classes.*

Structural vegetation class	Dense forest	Semidense forest	Open forest	Pine plantation	Semidense mattoral	Open mattoral	Rock outcroppings, pasture
Weighting factor	1	0.7	0.4	0.8	0.3	0.2	0

TABLE 5: *Categories of forest density (Source: USACE 1998).*

Descriptive category	Heavily forested	Forested	Partly forested	Open
Mean canopy cover (%)	>80	60-80	10-60	<10

TABLE 6: *Average monthly values ($mm\ month^{-1}$), Rio Claro watershed.*

Total Streamflow (F)	Baseflow (Fb)	Runoff (Fd)	Rain+snowmelt (Tw)	Rain (R)	Snowmelt (Sm)	Fb/F	Sm/Tw
166	107	58	204	131	73	68%	35%

TABLE 7: *Average monthly values ($mm\ month^{-1}$), Coyhaique watershed*

Total Streamflow (F)	Baseflow (Fb)	Runoff (Fd)	Rain+snowmelt (Tw)	Rain (R)	Snowmelt (Sm)	Fb/F (%)	Sm/Tw(%)
45	35	10	50	35	15	80%	26%

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ESTIMATION OF LOADS IN THE AYSÉN BASIN OF THE CHILEAN PATAGONIA: SWAT MODEL AND HARP-NUT GUIDELINES

M. YARROW AND P. CHAMBEL-LEITÃO

1 INTRODUCTION

The Aysén Basin presents a fascinating and challenging case for determining nutrient loads and understanding the ecohydrological processes that influence these loads. Three principal challenges were faced by the ECOMANAGE team in calculating nutrient loads: 1) nutrient cycling in the Chilean Patagonia differs significantly from temperate systems in the Northern Hemisphere, 2) the Aysén Basin has complex topographic, climatic, edaphic, and ecological characteristics that can not necessarily be easily modeled with current watershed models, 3) although ECOMANAGE was able to obtain all relevant existing data for the basin, certain 'data holes' persist. This chapter delves into the approach taken to confront these challenges and presents the results obtained and lessons learned in the process. Although the basin is often considered 'pristine', the nutrient loads calculated here show an important impact from human activities. The combination of a nutrient accounting and balance tool (Harp-Nut guidelines) with a semi-distributed watershed simulation model (SWAT2005) provides an integrated and flexible approach for estimating nutrient loads and allowing the simulation of possible future scenarios.

The main results of this work have been: (1) export estimation of Nitrogen and phosphorous to the Aysén Fjord using SWAT model and Harp-Nut guidelines, (2) comparison of SWAT model and bibliographic values on nitrogen cycling, (3) the development of associated tools to SWAT model facilitated the evaluation of nitrogen cycle, and (4) organic nitrogen in the rain is an important input in this basin.

2 DESCRIPTION OF AYSÉN BASIN

The Aysén River basin is located between 45^o - 46^{o}16' S and has a surface area of 11,456 km^2. The topography of the basin and the spatial distribution of precipitation within the basin are quite heterogeneous. Soils tend to be thin and are largely composed of volcanic ash (Hepp 1996). More detail about the geography, ecology and social history can be found in Yarrow and Torres (this volume).

Currently, the water quality of the basin's rivers does not present any marked problems and the waters have been classified as oligotrophic (Oyarzo 2006). However, as salmon aquaculture in Southern Chile grows by leaps and bounds, the Technological Salmon Institute has projected that the number of hatcheries in the basin will grow from about 18 to 114 (CONAMA 2006). Furthermore, the regional Agriculture and Livestock Service (SAG) has supported an intensification of livestock production in the basin - one estimate is that cattle will double within a decade. Thus, for regional planners to understand the implications of economic drivers on the water quality of the basin, it is important to develop tools and apply models to the basin for the calculation of nutrient loads.

3 NUTRIENT DYNAMICS IN PATAGONIAN FORESTS AND STREAMS

Perhaps the most outstanding feature of the Southern Patagonian forests is their isolation. Currently, a major effect of isolation is that rates of wet and dry nutrient deposition remain similar to historic rates, while in the northern hemisphere these rates have been severely altered. Inorganic N wet deposition to forested ecosystems in Southern Chile is limited; concentrations range from 0.015 to 0.071 mg l^{-1} (Hedin et al. 1995, Perez et al. 1998, Godoy et al. 2003). On the other hand, organic N in rainwater and especially cloudwater can contribute significantly to the total N deposition (Godoy et al. 2003). Net N mineralization has also been studied in Patagonia and reported values of between 12 to 37 kg N ha^{-1} yr^{-1} have been reported, significantly less than forested systems with anthropogenic N inputs (Pérez et al. 2003a). Denitrification can be generally characterized as low in well-drained forested soils and higher in wetter floodplain areas. Pérez and colleagues (2003a) show that denitrification in temperate forests of Chiloé Island are low: roughly 0.2 kg N ha^{-1} yr^{-1}. Volatilization of nitrogen does not appear to be important in the nutrient cycle of Patagonian forests, according to an N tracer study by Perakis and Hedin (2001). To synthesize the above: tight N cycles have been documented in several Patagonian temperate forests and would be expected in the Aysén Basin as well. This would result in low nutrient loading in the river network. However, the introduction of cattle and sheep into the basin has altered watershed dynamics in ways that are not yet fully understood. Figure 1 summarizes what is known about the fluxes and pools of the N cycle in Patagonian forests.

4 APPLICATION OF HARP-NUT GUIDELINES

Harp-Nut guidelines were developed to quantify and report on the individual sources of nitrogen and phosphorus discharges/losses to surface waters (Source Orientated Approach). These results can be compared to nitrogen and phosphorus figures with the total riverine loads measured at downstream monitoring points (Load Orientated Approach), as load reconciliation. Figure 2 shows, for the Aysén Basin, the delineation of monitored area and the non-monitored area. This delineation was made using the Aysén River gage station. The SWAT model will be the main tool that will be used to estimate diffuse loads.

4.1 Data Used

In general, there is little major industry within the Aysén Basin. Within the monitored part of the basin, the main point sources include mining, two small slaughterhouses, numerous gravel extraction sites, fish processing plants, fish hatcheries, and wastewater treatment plants. To support growth in the aquaculture sector, fresh-water hatcheries have sprouted up around the Aysén watershed, with more in planning stages. Although a recent CONAMA (2005) report listed 21 hatcheries, several of these hatcheries were not yet operational, leaving 15 hatcheries with reliable data. Harp-Nut Guideline 2 procedure 2 was used with the following assumptions: (a) the feed conversion ratio (FCR) is 0.6 as mentioned in Guideline 2 for fingerlings; (b) the yield of N and P from treatment is based on the estimative figures given in Guideline 2 and applied to all hatcheries employing some form of wastewater treatment (usually decanter); (c) the weight of individuals produced by the hatcheries is considered to be equal to the weighted average of all smolt and alevin production reported by SERNAPESCA for 2005 and 2006 (21.9 g individual^{-1}); (d) information on P and N content in feed and organisms was extracted from Guideline 2 and a report from Fundación Chile (1997).

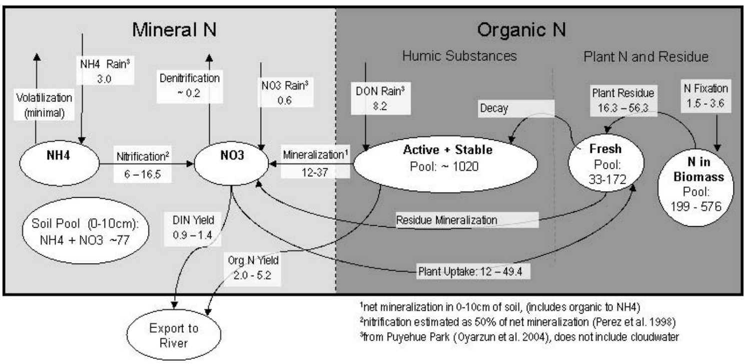


FIGURE 1: *Diagram of N cycle in SWAT2005 with pools and process rates for Patagonian forests taken from the literature. Flows and Pools are in kg N ha^{-1} (Diagram adapted from SWAT2005 Theory).*

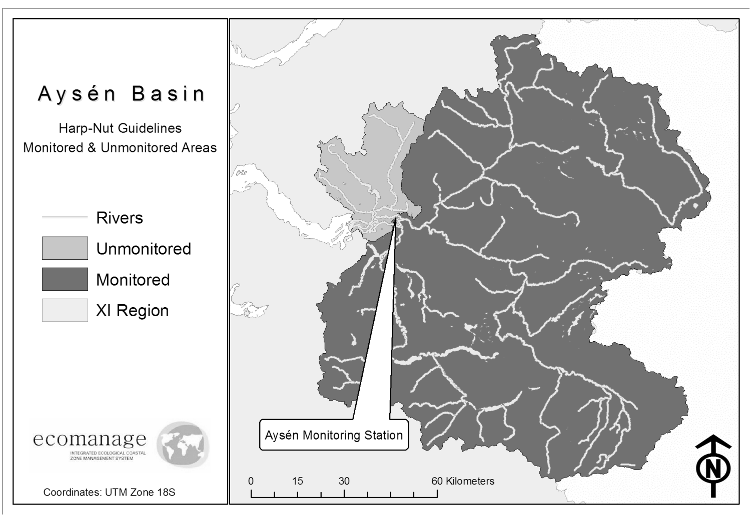


FIGURE 2: *Location of the monitoring station used in the application of the Harp-Nut Guidelines.*

CONAMA provided four composite samples of effluent from Comercial Mañihuales (a food processing plant) as well as average monthly effluent flow rates. Salmones Antartica has two plants that are considered here. Calculations were based on method 3 of Guideline 3, however due to few data points the effluent flows and nutrient concentrations were averaged and then multiplied by 365 to get annual loads.

Harp-Nut Guidelines 4 & 5 deal with sewage treatment. There are five waste water treatment plants (WWTP) in operation in the Aysén watershed. Only about 4% of the population in the Aysén watershed is not hooked up to a municipal waste treatment system. For the two main WWTPs Coyhaique and Puerto Aysén, information about the average daily discharge and monthly quarter quality monitoring was available (CADE-IDEPE 2004). Nutrient loads for the three other plants were estimated via person equivalency information found in the OSPAR guidelines. A 50% treatment efficiency was used for these three smaller plants in absence of concrete local information. Information about rural and urban population sizes in the Aysén Basin was taken from the 2002 National Census. OSPAR population equivalency information was used to calculate N and P loads from rural households.

Harp-Nut Guideline 7 deals with the measured nitrogen and phosphorus loads in a given basin. A primary problem encountered in the application of this guideline is the lack of important variables (N-total, Organic N and P-total) and the sparse temporal resolution (at best seasonal). Additionally, some of the data points represent outliers with no further information about why they may have originated. As a way to deal with the uncertainty introduced by this data set, we have consulted several other sources about nutrient concentrations in the rivers of southern Chile. A study by Oyarzun et al. (2004) found that in a micro-watershed in the X Region of Chile, nitrogen exported via runoff was made up of close to 89% organic nitrogen, while the nitrate fraction contains only 6% of exported N. The work of Perakis and Hedin (2002) agrees with Oyarzun et al. (2004) in that nitrogen export from 100 first-order streams in unpolluted temperate South American forests is mainly in the dissolved organic form. Given the large geographical coverage of the Perakis and Hedin study, it is highly likely that small streams in the forested areas of the Aysén Basin conform to this pattern.

On the other hand, two other studies indicate that the small ratio of nitrate: total N export mentioned above cannot be simply extrapolated to larger bodies of water or watersheds with non-forested landuses. A study by Oyarzún and Huber (2003) compared stream nutrient concentrations in forested and pasture-covered watersheds in the X region of Chile. This study indicated that NO_3 makes up a higher proportion of the total N in watersheds dominated by livestock activities. Finally, water quality data collected for a MOP-DGA study (1996) in the inlets and outlets of Lake Riesco and Lake Elizalde (located within the Aysén Basin) indicated that on average, nitrate made up 40% of the total-N, and orthophosphate made up 37.5% of the total-P. Because the MOP-DGA sampling was focused on lacustrine systems and included a relatively short time series, the authors have decided to combine this data set with results from the three other publications mentioned previously. For nitrogen, a ratio of nitrate:total N of 0.219 was used while a ratio of phosphate:total P of 0.375 was used for Phosphorus.

These ratios were then used to convert the measured concentrations $NO_3 - N$ and $PO_4 - P$ into Total-N and Total-P, respectively. This conversion represents a major assumption as to the relative proportions of the different N fractions within the Aysén River, values that are likely to change seasonally and through time, as different productive activities are undertaken in the watershed. However, this is the only way forward using the currently available data.

In order to arrive at nutrient retention estimates, the German approach described in Harp-Nut guideline 9 was used. The area of the areas of lakes and reservoirs with the Aysén Basin was calculated in ArcView® 3.3 to be 164 km^2. Additionally, fluviometric data from Guideline 7 was used to calculate retention using the equations described in this guideline. The parameters in said equation were derived empirically from 100 river basins in Europe.

4.2 Results of Harp-Nut Application

Due to uncertainty associated with the measured values of N and P for the Aysén monitoring station, the values of the Harp-Nut ”load-oriented” approach should be interpreted with caution (Table 1). Nevertheless, the correspondence between the load-oriented and source-oriented approaches indicates that despite challenges with data completeness, the general patterns and magnitudes of nutrient loading in the Aysén Basin have been correctly identified. By using a nitrate: total N ratio that represents a range of studies approximating the different conditions found in the Aysén Basin, the load-oriented estimate is 3438 tons N yr^{-1}, which is only 8% higher than the source-oriented estimate. The inclusion of N loads from the un-monitored part of the basin brings the total annual N load to the Aysén fjord to 3641 tons yr^{-1}. The source oriented approach quantifies N loading by sector. Table 1 shows that point sources of nitrogen are relatively low in the Aysén Basin - on the order of 4.5% of the total N loads. Diffuse N loads from ranching and land-use change are estimated to be 15% of total N loads. Thus, according to the application of the Harp-Nut guidelines, just under 20% of all N reaching the Aysén Fjord is of human origin.

Because less literature on P loads in Patagonia was encountered, the load-oriented estimate is based on a $PO_4 - P$: Total-P ratio from the MOP-DGA study (1996). The correspondence between the source-oriented and load-oriented approaches is lower for P than for N. A more representative sample of P fraction in the basin would likely provided improved results. Nevertheless, general patterns emerge. The phosphorus loads showed more human influence as compared to nitrogen, with 38% of P loading deriving from human sources. Again, point sources of P are low, totaling 7.3% of total P export. This indicates that diffuse sources of P, such as ranching and landuse change comprise just under one third of P loading to the Aysén Fjord. It is interesting that the SWAT model indicates that P is lost much more readily than N to the river network when considering grazing. More needs to be done to verify this pattern, but given that P is often a limiting nutrient for agricultural activities in the basin, this finding is of some interest (Hepp 2004).

TABLE 1: *Results of the application of the Harp-Nut Guidelines to the Aysén Basin.*

Harmonised Quantification and Reporting Procedures for Nutrients (HARP)
Quantification procedure for "Source Orientated Approach" and "Load Orientated Approach"

Aysén Basin			tons/year		
Operator	**Quantification sequence for discharges and losses of nitrogen**	Guidelin	MON	UNMON	TOTAL
+	Quantified nitrogens losses from aquaculture	2	52	1	53
+	Quantified nitrogen discharges from industry	3	4	13	16
+	Quantified nitrogen discharges from sewage treatment works	4	81	24	106
+	Quantified N losses from households not connected to	5	8	1	9
+	Quantified diffuse anthropogenic nitrogen losses	6	572	49	622
+	Quantified natural nitrogen background losses	6	2899	368	3267
=	**Sum of nitrogen losses/discharges (Source Orientated Approach)**		**3617**	**456**	**4072**
-	Quantified nitrogen retention in surface waters	9	431	0	431
=	**Total estimated transport of nitrogen at the monitoring point (derived from the Source Orientated Approach)**		**3186**	**456**	**3641**
	To be compared with:				
	Load Orientated Approach - Total	7	**3438**		
Operator	**Quantification sequence for discharges and losses of**	Guidelin	MON	UNMON	TOTAL
+	Quantified phosphorus losses from aquaculture	2	14	0	14
+	Quantified phosphorus discharges from industry	3	0.6	3.59	4
+	Quantified phosphorus discharges from sewage treatment	4	22	8	30
+	Quantified P losses from households not connected to	5	1	0	1
+	Quantified diffuse anthropogenic phosphorus losses	6	204	4	208
+	Quantified natural phosphorus background losses	6	392	27	419
=	**Sum of all phosphorus losses/discharges (from Source Orientated Approach)**		**633**	**42**	**676**
-	Quantified phosphorus retention in surface waters	9	101	0	101
=	**Total estimated transport of phosphorus at the monitoring point (derived from the Source Orientated Approach)**		**532**	**42**	**575**
	To be compared with:				
	Load Orientated Approach - Total	7	**421**		

5 SWAT MODEL APPLICATION

The SWAT model was used to: (a) estimate the background levels of N and P loads in the Aysén Basin (b) estimating diffuse N and P loading by changing landuse to reflect current conditions and utilizing a SWAT grazing routine reflecting recent stocking rates. These model runs were used in the application of the Harp-Nut Guidelines. Additionally, work was done to adjust the SWAT nutrient modules so that they would reproduce patterns of nutrient cycling found in the Chilean Patagonia.

5.1 Data Used

The landuse information used in the Aysén SWAT model came from two sources. First, for the current landuse the Registry of Native Forest (CONAF-CONAMA 1999) was utilized. This database includes a large number of land cover classes, subdivided by physiognomy in the case of forest classes. This information was converted (and simplified) into a format usable by SWAT. Built-in SWAT land cover classes were used for agricultural areas, pastures and rangeland, pine plantations, and urban areas. It was not clear how to simulate the native forest communities of southern Chile using the limited array of SWAT tree classes. Thus, five new SWAT classes were created based on an extensive literature review and expert opinion. Finally, due to the large proportion of the basin that is either above the vegetation limit or is best described as rocky outcroppings, a new Urban SWAT class was created: ROCK that was highly impervious but that exported few nutrients. In the case where it is important to simulate hydrodynamics or nutrient loading without considering anthropogenic influence in the basin, a vegetation map generated by Gajardo (1994) was used. This map is essentially a hypothesis of the original vegetation types occurring in Chile based on physiological traits of the dominant species and on pollen cores.

Data from 6 weather stations within the Aysén Basin and one station to the south of the basin have been incorporated into SWAT. The significant periods for which no data is available for all or some of the weather stations represents a further obstacle. Holes in the precipitation data were filled by extrapolating from the stations with the highest Pearson's correlation coefficient value (not always the nearest station). Due to the extreme spatial heterogeneity in precipitation in the Aysén Basin, the simple algorithm used by AVSWATX to assign precipitation stations to subbasins was not adequate. Two steps were needed to produce stream flows that approximated observed flows. The first step was the addition of four 'synthetic' precipitation stations, especially in those areas of the basin that get over 3000 mm of rain. This approach involved multiplying an observed precipitation station by a factor to bring the annual average precipitation to the desired level. Precipitation stations were then manually assigned to subbasins in order to approximate the isohyets produced by the DGA of the XI region. Figure 3 shows the result of the work to create reasonable precipitation input for SWAT.

Because of the cold climate, there is little agriculture in the Aysén Basin. However, extensive pastures in the valleys have been established through the burning of the original forest. The Agriculture and Livestock Service (SAG) has several programs that provide incentives for local landowners to improve their pastures, which are often infertile and unproductive. Pasture seeding and rehabilitation programs have been declining in recent years, while the trend has been towards more fertilization. Phosphorus and sulfur can increase forage production by 2 to 3 times (Hepp 2004). Thus, although the basin is not currently a major producer of meat for export, the SAG sees potential for expansion in this area, based on fertilization and improvement of existing pastures. A census generated by the SAG for the Aysén Basin listed 46,350 cattle and 30,725 sheep present 2004. The importance of diffuse agricultural sources of nutrients in the Aysén Basin has been sparsely documented. Observations made by ECOMANAGE suggest that few vegetative or geological barriers prohibit nutrient rich grazing byproducts from reaching the river system. Values taken from the literature were used to convert information on cattle stocking rates into parameters used by the SWAT management module (Manske 2004).

5.2 Model Calibration

Although SWAT2005 includes an autocalibration tool, many authors recommend using this after a manual calibration. In the case of Aysén, manual calibration alerted us to many of the particularities of the basin that needed to be addressed in the model. The first step was to run the SWAT model with the data described previously. This served as a baseline run, to which subsequent modifications were compared. Calibration for river flows was carried out using four fluviometric stations: (1) Simpson River (2) Mañihuales River, (3) Blanco River, and finally (4) Aysén River. An issue that became readily apparent was that the algorithm used by the AVSWAT-X interface to assign precipitation stations to subbasins was not optimal. For the period between 1996 and 2002, the measured average streamflow at the Aysén River was 537 $m^3\ s^{-1}$. This contrasts with a value of 375 $m^3\ s^{-1}$ with automatic assignment of precipitation stations and 531 $m^3\ s^{-1}$ with manual assignment.

A second issue that affects streamflow in this mountainous basin is snowfall. Because the DGA's weather stations in the basin are at low elevations where winter temperatures are tempered by prevailing winds off the Pacific Ocean, snow occurs infrequently in SWAT under the baseline setup. In order to enhance this key hydrologic process, we calculated 10 elevation bands for each subbasin allowing the temperature of the higher bands to fall below freezing for a longer period of time. This has the effect of increasing snow as a percentage of total precipitation from 0.5% in the baseline run, to between 15-40% in the runs with elevation bands (Figure 4). Although there is no empirical data about the percent of precipitation that falls as snow in the Aysén Basin, this range is reasonable given that a) a maritime climate on the leeward side of the Andes range produces heavy precipitation, much of it as rain and b) snow is common in the higher mountains and eastern plateaus but areas east of the main Andes range are significantly drier. The strength of a snowmelt signal in the flow data of the Aysén River varies interanually. Manipulating the TLAPS parameter (the temperature lapse rate -$^\circ$C km^{-1}) can produce more snowfall in the model, but this often reduces the quality of hydrodynamic results unless measured temperature and radiation data is available to accurately calculate snowmelt pulses.

It is important to note that the Blanco River subbasin is the largest contributor to the streamflows at the Aysén fluviometric station. This subbasin is very mountainous, has the highest precipitation in the basin, contains several major lakes and does not benefit from having precipitation or weather stations. Thus, it represents a challenge in the process of model setup. We found the best approach under these circumstances is to link changes the parameterization with physical processes that are known to be occurring in the subbasin. On the other hand, the Mañihuales subbasin has three precipitation and weather stations and consequently produced better model results.

5.3 Hydrodynamic results

The results from the autocalibration analysis were used as the basis for the final hydrodynamic setup of the Aysén SWAT model. Because the autocalibration minimizes the sum of the squares of the residuals for daily streamflow, the daily statistics improved significantly. However, in order to strike a balance between daily and monthly hydrodynamic results, some of the best values from the autocalibration were changed. Thus, the CN2 parameters were multiplied by 1.3 instead of 1.5, and the canmx parameter was increased to 10 in forested HRUs. The value of Lat_time in the final calibration was 3.5. According to a recent paper on model evaluation guidelines, NSE values greater than 0.5 indicate satisfactory model performance while values above 0.65 indicate good performance (Moriasi et al. 2007). Table 2 shows that during the calibration phase, the Aysén SWAT model had good performance at all stations except Blanco River. The difficulty in modeling the Blanco subbasin is evident by looking at the validation statistics. Poor model performance in this subbasin can affect the Aysén station downstream. Nevertheless, the monthly performance of the model at the Aysén station for the period 1996 to 2005 is satisfactory.

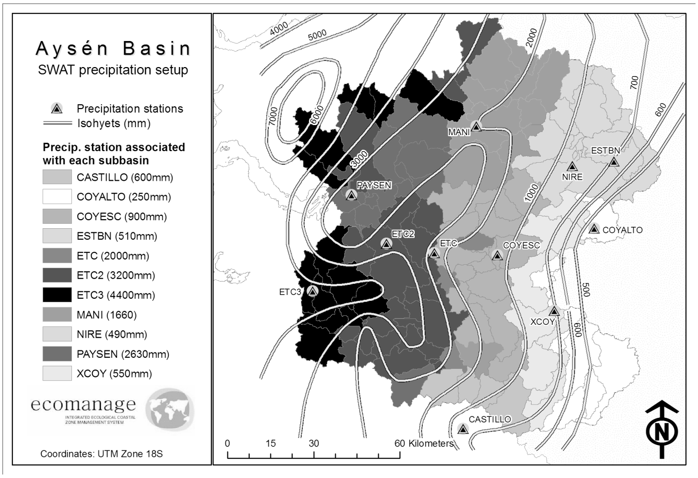


FIGURE 3: *Isohyets and the assignation of precipitation stations among subbasins.*

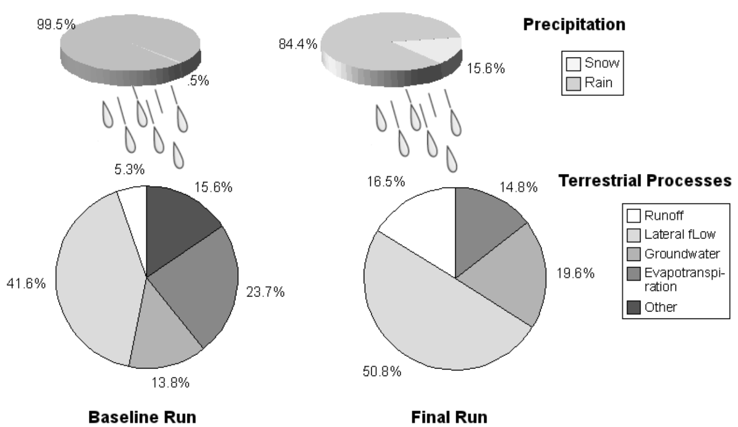


FIGURE 4: *Breakdown of terrestrial hydrological process (percent of total precipitation). The baseline run (no elevation bands/WATR landuse class) and the final run (elevation bands/RNGB instead of WATR landuse class) are shown.*

5.4 Nutrient dynamics in SWAT

The SWAT model was used to calculate annual nitrogen and phosphorus loads from diffuse sources. An initial comparison of the measured inorganic nutrient concentrations from the Aysén monitoring station and the annual nutrient loads estimated with SWAT led to the conclusion that diffuse loads were being significantly overestimated. This led to a more detailed examination of the nitrogen cycle used in SWAT and the inferences that can be drawn about N cycling in the Aysén Basin.

5.4.1 Wet deposition of nitrogen

As mentioned previously, wet deposition of nitrogen in Southern Chile differs from most areas of the Northern Hemisphere. It was found that the RCN parameter (concentration of N in precipitation) in SWAT2005 was code was fixed at 1 $mg\ l^{-1}$, significantly higher than measurements from Southern Chile (Godoy et al. 2003). Furthermore, the literature from Patagonia indicates that dissolved organic nitrogen (DON) can make up two thirds of wet deposition (Oyarzún et al. 2004). Thus, SWAT source code was changed to allow DON to be added as a component of wet N deposition (as well as NO_3 and NH_4).

After a spin-up period of 2 years, we compared the average daily NO_3 concentration at the subbasin that corresponds to the Aysén River monitoring station with 28 measurements of $NO_3 - N$ taken between 1997 and 2004 by the local authorities. The average measured value was 0.0475 $mg\ l^{-1}$ while SWAT NO_3 output for the corresponding reach was 0.0453 $mg\ l^{-1}$. The difference between modeled and measured values was not statistically significant (*t= 0.40, p=0.69, gl=27*). The same simulation set up, run with RCN = 1 gives a NO_3 concentration in the reach of 0.74 $mg\ l^{-1}$, which is higher than even the highest measured value.

5.4.2 Tree growth and residue production

SWAT outputs the biomass generated in each HRU, and though for agricultural landuses this figure may be directly compared to crop growth and harvest, the situation for forests is more complex. In SWAT2005, the default tree parameter sets (e.g. PINE) lead to a large fraction of annual biomass production removed as yield or converted to residue, resulting in minimal growth of persistent biomass such as trunks and large roots. If the primary objective for calibrating biomass is to achieve realistic nutrient cycling, then the total biomass of a forest system can be ignored and the focus can be placed on residue production. In Table 3, the ranges of litterfall for the three principal forest types in the Aysén Basin were gathered from the literature and compared to SWAT output after calibration. The most sensitive parameter modified was BIO_LEAF which controls the fraction of biomass converted to residue at the end of each season. Other parameters changed-according to values found in the literature-were the nitrogen and phosphorus uptake parameters, LAI, canopy height, rooting depth, and the harvest index.

TABLE 2: *Results for the calibration and validation phases of the final Aysén SWAT setup. Statistics: Root Mean Square Error/Coef. of Determination (R2)/ Nash-Sutcliffe Efficiency (NSE).*

	Station	Time period	Observed/ modeled flow ($m^3 s^{-1}$)	RMSE Day/Month	R^2 Day/Month	NSE Day/Month
Calibration	Aysén	Jan '96–Dec '99	531 / 535	321 / 136	0.48 / 0.75	0.47 / 0.70
	Mañihuales	Jan '96–Dec '99	176 /176	141 / 70	0.47 / 0.67	0.46 / 0.67
	Simpson	Jan '96–Dec '99	48 / 56	31 / 19	0.60 / 0.81	0.55 / 0.72
	Blanco	Nov '98-Dec '00	230 / 220	131/78	0.38 / 0.41	0.07 / 0.27
Validation	Aysén	Jan '01-Jul '05	579 / 513	278 / 189	0.45 / 0.53	0.35 / 0.42
	Mañihuales	Jan '01-Jun'05	180 / 161	122 / 71	0.50 / 0.69	0.48 / 0.62
	Simpson	Jan '01-Jun'05	62 / 52	37 / 25	0.48 / 0.66	0.42 / 0.55
	Blanco	Jan '01-Jun'05	284 / 234	145 / 112	0.38 / 0.36	0.00 / 0.02

TABLE 3: *New SWAT Land Cover Classes Used in Aysén SWAT Model.*

SWAT Code	Vegetation Type	Main Species	Range of litterfall ($mg\ ha^{-1}\ yr^{-1}$)	Simulated residue, range and mean ($mg\ ha^{-1}\ yr^{-1}$)
BCAY	Deciduous forest of Aysén	*Nothfagus pumilio*	2.0 - 3.6	1.9 - 4.2; 2.93
MCAM	Montane deciduous forest	*N. Antarctica, N. pumilio, Berberis spp.*	1.4 - 2.5	1.2 – 2.8; 1.94
BSNB	Montane evergreen forest	*Nothfagus Betuloides, Laurelia philippiana*	2.8 - 3.8	2.3 – 3.82; 3.19

Sources: Caldentey et al. 2001, Austin and Sala 2002, Vann et al. 2002, Pérez et al. 2003b

5.4.3 New nutrient output for SWAT/MOHID

One obstacle for the rapid exploration of nutrient cycling in SWAT is the absence of a comprehensive output file that includes all processes and pools for a given land cover class. It is possible to look at individual HRUs, but this is tedious when a SWAT setup has several hundred HRUs. SWAT2005 output also provides basin-wide yearly averages for some nutrient forms and processes, but this output does not allow the examination of a certain land cover. Using the MOHID programming philosophy and making use of MOHID post-processing tools, the source code of SWAT2005 was modified to create an output file for each landuse class that was the average of all HRUs of that class (Chambel-Leitão et al. 2007). In order to condense the large amount of data in these new *.ldu files, we created a macro in Microsoft Excel that calculates the average values over a user-specified time period for all the pools and processes in the SWAT model. These average values are then placed in a diagram as a way to facilitate the visualization and analysis of nutrient cycles.

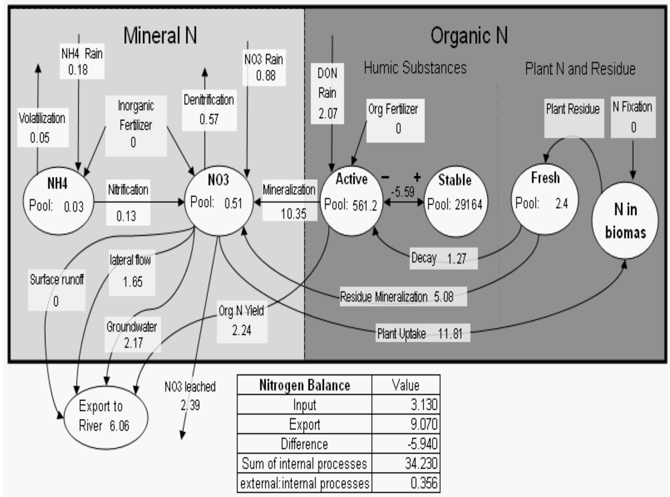
The deciduous forest of Aysén cover class (BCAY) was used as an example of this new output (Figure 5). Without changing SWAT source code, we were able to replicate the relative magnitude of some of the nitrogen processes and pools such as denitrification, volatilization, mineralization and plant uptake. However, some others were difficult to calibrate effectively. For one thing, it is difficult to limit NO_3 export via lateral flow and groundwater flow, even lowering the NPERCO parameter to a very low parameter allows significant export of NO_3. As this model has been used primarily in areas where NO_3 is the most important form of N exported, certain equations should probably be modified. Furthermore, the stable organic pool of nitrogen is high. The equation that initializes soil N content from organic carbon might overestimate the N in humus. However, this also indicates that more data is required to understand patterns of soil organic matter in the Aysén Basin. One result of calibrating the nitrogen cycle in SWAT was that the cycle became tighter, showing a low ratio of external: internal N flows.

6 DISCUSSION AND CONCLUSION

The concurrent use of the SWAT model and Harp-Nut Guidelines is a good approach in the face of uncertainty and in the current case-study, has produced good results. We conclude that roughly 20% of N loads in the monitored part of basin appear to come from anthropogenic activities, while 38% of P loads come from anthropogenic activities. Although this analysis indicates that aquaculture in the form of salmon hatcheries is currently contributing less than 4% of the total N and P in the basin, it is clear that a dramatic increase in the number of hatcheries could have an important impact on nutrient loads. Our analysis indicates that diffuse anthropogenic activities of N and P are very important in the Aysén Basin. Any increase in livestock production intensity should be accompanied by the implementation of best management practices (BMPs). The HARP-NUT results can be confirmed and improved by increasing the sampling frequency of water quality in the rivers to a monthly basis.

The work represented by this chapter and previously papers (Yarrow and Chambel-Leitão 2007a, 2007b) is useful in synthesizing current information about hydrodynamics and nutrient dynamics in the Aysén Basin and providing a basis for further work in the area. We have emphasized the terrestrial nitrogen cycle, because (1) diffuse nutrient sources appear to be very important in this basin, (2) local authorities have limited information on terrestrial nitrogen pathways and nitrogen species concentrations in freshwater systems. Further work could (1) examine denitrification rates in the pluvious western part of the Aysén. (2) study mineralization *in situ*; given high measured organic soil content in the basin, mineralization of organic nitrogen can greatly affect availability of inorganic N for vegetation., and (3) map patterns of soil organic content throughout the basin (currently this has been limited to productive areas). This work would improve SWAT results given that nutrient export in SWAT is relatively sensitive to soil organic matter.

We have taken steps to improve the modeling of watersheds dominated by relatively unpolluted temperate forests with SWAT2005. Our strategy has been to make small modification instead of adding more complex routines requiring additional parameterization or input data. We have seen results improve: the ratio of organic N to inorganic N in river water has decreased as we have calibrated and then modified the model. Apart from a few processes, the annual fluxes in the SWAT N cycle for the BCAY cover class corresponded to what were gleaned from the literature. A macro created in Excel allows diagrams (such as Figure 5) to be rapidly produced. SWAT results indicate that the internal N cycling in the basin is high compared to either inputs via precipitation or exports to the river system. This is consistent with published work in temperate forests of southern Chile (e.g. Pérez et al. 2003).



Nitrogen Balance	Value
Input	3.130
Export	9.070
Difference	-5.940
Sum of internal processes	34.230
external:internal processes	0.356

FIGURE 5: *Output of SWAT2005 after calibration and changes for BCAY forest type ($kg\ N\ ha^{-1}\ yr^{-1}$).*

It is important to mention the utility of using SWAT with the three wet deposition compartments (NH_4, NO_3, and DON) in (1) approximating the N cycle in remote areas of the Chilean Patagonia and (2) studying the potential effects of increasing anthropogenic N emissions worldwide and the interactions between climate change and biogeochemical cycles. We strongly encourage regional authorities to measure the different forms of N deposition within the Aysén Basin to confirm assumptions made in the current work and as a way to detect changes in the N cycle over time. Additionally, monitoring can confirm the importance of DON export from subwatersheds that have different levels of anthropogenic activities. In concluding, the results of the modeling and analysis in this chapter provide several concrete suggestions for future research and monitoring in the Aysén Basin. Furthermore, they suggest that rapid development of different economic sectors such as aquiculture and animal production could have a notable impact on nutrient exports. We recommend that management approaches and continued data collection be integrated among the regional authorities. As management plans are refined, they can be evaluated with the SWAT model and the kind of detailed analysis of the nutrient cycles undertaken in this chapter.

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HYDRODYNAMICAL VERTICAL 2D MODEL FOR THE AYSÉN FJORD

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1 INTRODUCTION

As a first approach to model the hydrodynamics and ecological evolution of a fjord, a 2D vertical model was implemented to evaluate the sharp gradients that occur on fjords with a high river discharge. The Aysén Fjord located on Region XI on the Chilean Patagonia is one of the study sites of the ECOMANAGE Project, along with Bahía Blanca and Santos estuaries. The Aysén fjord is one of the natural connections between the Andean mountains and the system of islands and fjords in southern Chile, called the Chilean Inland Sea. In this fjord, several extreme events take place which make it special from an oceanographic point of view: very large river discharges fed by rain and by melt water from the Andes glaciers accompanied by low temperatures provoking a very sharp gradient in the vertical density structure of the system. This feature is very important for the functioning of the system from the hydrodynamic and the ecological point of view.

2 NUMERICAL MODEL IMPLEMENTATION

The Mohid modelling system (Leitão et al., this volume), has been employed to simulate the baroclinic hydrodynamics of the Aysén fjord. Turbulent vertical mixing has been reproduced using the embedded GOTM model (Burchard 2002) using the parameterisation proposed by Canuto et al (2001) with the exception of the minimum Turbulent Kinetic Energy (k_min) value for the Mellor-Yamada turbulence model, most satisfactory results were obtained with a value of $5x10^{-5}$.

2.1 Domain discretisation

The Aysén 2D vertical model domain was obtained from a previous 2D horizontal domain obtained through interpolation of the digitalised nautical charts from the Chilean Navy's Hydrographic and Oceanographic Service (SHOA, its Spanish acronym). A transect following the main axis of the fjord was defined and the extracted depths were converted into a regular grid that covers the major bathymetric features of the fjord seabed, i.e. sea mounts and sea terraces (Figure 1). The horizontal distance covered by the domain was divided into a regular horizontal grid with 64 cells (Figure 2). The first 47 cells correspond to smoothed bathymetric data to avoid numerical problems and the last 17 were added for numerical stability with the imposed boundary conditions. Vertically the discretisation consisted of 30 layers divided into two vertical domains: a cartesian domain consisting of 16 layers from the bottom to a depth of 32 m; above which was defined a sigma domain with 14 layers up to the free surface (Figure 2). Both domains been have defined taking into consideration the known vertical stratification and in order for the model to be able to reproduce cell heights decrease from bottom to top (Table 1).

TABLE 1: *Layer thicknesses of the two domains in the 2D vertical model in meters. Mohid layers are numbered from bottom to top.*

Layer	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Cart.	50	50	20	20	20	20	20	10	10	10	10	8	5	5	5	5
Sigma	5	5	3	3	2	2	2	2	2	2	1	1	1	1		

2.2 Hydrodynamical forcing

On each of the sides of the model domain a different hydrodynamical forcing was imposed: tides on the open boundary and river discharge on the inland area. On the ocean boundary were imposed the tidal components obtained from the Chacabuco Harbour tidal gauge records for the years 2003-2004. Tidal analysis performed on the water level series using the TASK-2000 Software (Tidal Analysis Software Kit 2000, POL/PSMSL) was used to obtain 63 tidal components.

The Aysén River, the largest river discharging into the fjord by volume, located at the head of the fjord is the only one considered to force the model thus the influence of the lateral fresh-water inputs from other rivers are neglected in this model. In order to force river discharges with realistic flow values despite the lack of observed data, the Soil and Water Assessment Tool (SWAT) model (Arnold and Fohrer 2005) has been applied to the Aysén River catchment. An application of the SWAT model to the Aysén catchment (Yarrow and Leitão 2007) produced the monthly flow values summarised in Table 2. The river discharge has been implemented dividing its flow over the top six cells at the land boundary. Monthly river temperatures were obtained by adjustment into a curve of the temperature data collected by CADE-IDEPE (2004) and it was also assumed that the water discharged was pure freshwater (0.01 PSU).

To characterise the Aysén area atmospheric conditions, monthly values for air temperature, relative humidity and cloud cover data from Meteorological Annals of Chile colleted at Puerto Aysén for the years 1968-1969 (Ministry of Defence, Table 2) were imposed in the model Atmosphere Module. Additionally, winds measured at Mitahue Island (45°24'05.94" S, 73°44'25.22" W, Figure 3), located in the Moraleda Channel near the mouth of the Aysén fjord, during the CIMAR 7 campaign in 2002 were imposed only to calculate atmosphere-sea heat exchanges due to wind. Wind forcing was not taken into consideration for hydrodynamical forcing. Though the Mitahue Island is not located in the Aysén fjord, this dataset represents wind speeds and directions registered in the vicinity of the fjord, and could be considered as a typical channel location record. Recorded data show that the wind blows predominantly from the third and fourth quadrant, from North to Southwest, and during most of the time wind speeds do not exceed 5 $\mathrm{m\ s^{-1}}$.

TABLE 2: *Monthly averaged values for river properties (flow and temperature) and atmospheric properties (temperature, relative humidity and cloud coverage).*

Month	River Flow ($m^3 s^{-1}$)	River Temp (ºC)	Air Temp (ºC)	R Humidity (%)	Cloud Cover (%)
1	325.86	11.01	14.15	88.2	79.2
2	249.45	11.88	12.30	89.3	87.9
3	336.82	11.37	11.50	88.3	69.2
4	475.12	9.60	9.35	92.3	81.0
5	655.63	7.05	7.65	93.0	88.8
6	731.80	4.40	4.75	92.2	79.0
7	729.44	2.37	4.55	93.0	85.8
8	814.96	1.49	5.60	91.0	79.4
9	636.33	2.01	7.00	85.8	84.8
10	542.83	3.78	8.00	88.2	79.6
11	436.88	6.33	11.30	86.2	84.2
12	439.44	8.97	12.85	85.0	77.5

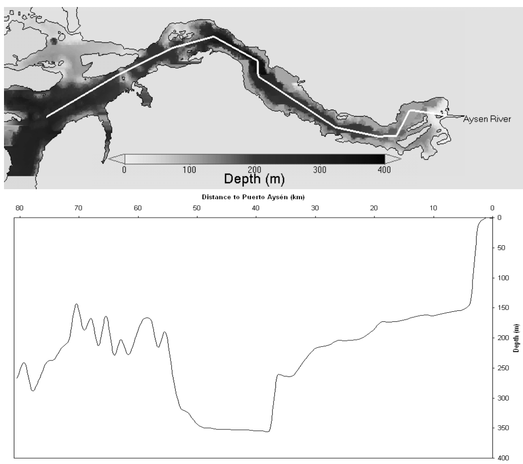

FIGURE 1: *Aysén fjord domain bathymetry with a transect along the main axis (Top) and the depth contour of this transect (Bottom).*

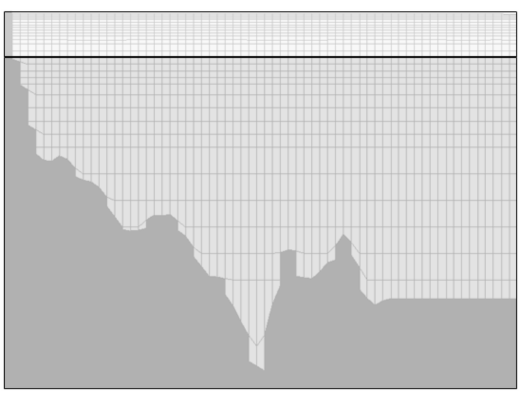

FIGURE 2: *Aysén Fjord 2D vertical model discretisation.*

FIGURE 3: *Mitahue Island geographical location (Left) and wind speeds and directions (Right).*

3 MODEL RESULTS

3.1 Hydrodynamics

The residual velocities are the net velocities obtained after averaging currents. The residual currents presented in Figure 4 were obtained from one year simulations. The residual currents present a clear three layer structure similar to the one described in the literature (Cáceres et al. 2002, Valle-Levinson et al. 2002). At the surface there is a strong seaward current immediately below there is landward. Below these two superficial layers there is a much thicker third layer where the residual current does not have a clear pattern. The two layers superficial flow is clearly induced by the density gradients caused by the input of fresh water at the head of the fjord. The fresh river water tends to lie on top of the denser sea water. This induces a seaward sea surface height gradient. This gradient generates a superficial seaward residual current, with the landward density gradient generating an undercurrent.

When the relative scale between the horizontal and vertical components of the vector is enlarged it can be seen how the interaction of the barotropic tide with the bottom topography creates internal tides. The main area of internal wave generation would be in the pinning points and the sill found at the entrance of the Aysén fjord. The irregularities on the seabed are mainly reflected in the vertical velocities as can be observed in Figure 5. This result shows that below the surface layers the model presents residual velocities with a turbulent structure highly conditioned by the internal tide. It should be kept in mind that the model was only forced with river discharge and tide. The residual current structure may be affected by other processes not considered in this implementation like wind forcing or interaction of the fjord circulation with the large scale circulation.

Instant maximum currents values are observed on the surface layer depending on the river discharge. In this surface layer velocities according to the model are comprised mainly between 0.1 and 0.3 m s^{-1} (Figure 6).

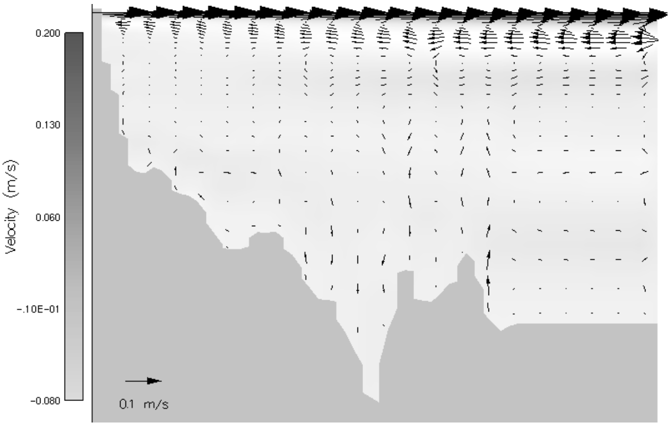

FIGURE 4: *Residual currents after one year simulation (arrows) and horizontal velocity (grey scale) showing the three layer circulation. The vertical vector component scale is 500 times the horizontal vector component.*

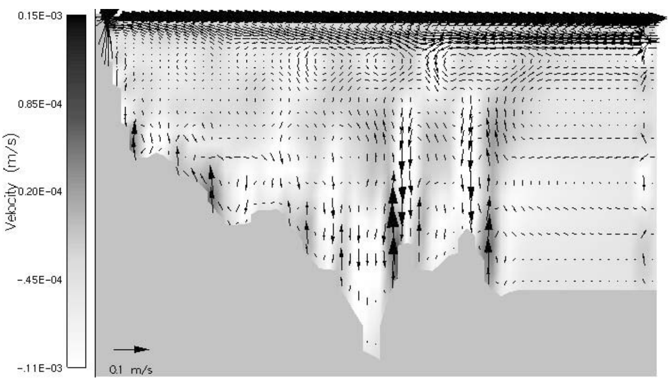

FIGURE 5: *Residual currents after one year simulation (arrows) and vertical velocity (grey scale) vertical vector component 1000 times the horizontal vector component.*

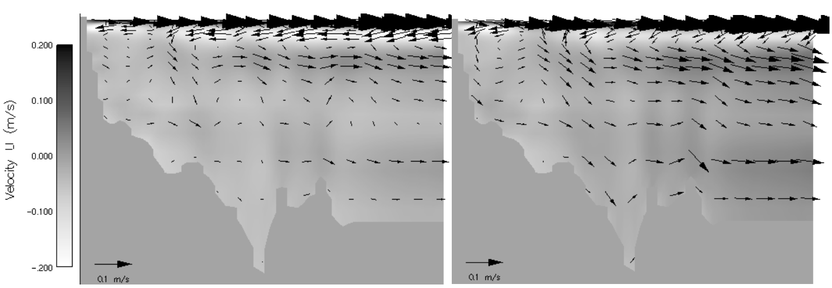

FIGURE 6: *Instantaneous currents during flood (Left) and ebb conditions (Right).*

3.2 Salinity and Temperature

Most of the data used for the Aysén study area come from the different CIMAR-FIORDOS oceanographic cruises (hereafter, CIMAR) that cover the whole system of southern Chilean fjords funded by the Chilean Navy's Hydrographic and Oceanographic Service (SHOA, its Spanish acronym). In addition to the CIMAR cruises, the Salmon Technological Institute (INTESAL, its Spanish acronym) which is part of the Salmon Industry Association, SalmonChile (`http://www.salmonchile.cl/`), made available temperature, salinity, density and oxygen vertical profiles data collected on the 10th of February 2007 on nine stations along the Aysén Fjord. Further descriptions of these datasets can be found on the chapter "Ecological Conceptual Model for a Southern Chilean fjord: the Aysén Fjord case study".

Vertical salinity gradients in the Aysén fjord are sharp due to the high amount of water collected by the fringing watersheds that reach the surface of the waterbody by their respective watercourses. River flow characterisation is not sufficiently defined due to the lack of observations in many of the watercourses. The largest one in discharge volume is the Aysén River located at the innermost part of the fjord. The other rivers for which discharge information is available are located in the vicinity of the Aysén River.

Haline stratification occurs in the first meters where values decrease from surface values between 2.70 and 10.60 PSU (inner and outer station values) to stable deep values of around 31 PSU at 80 meters (Figure 7), though the strong halocline is in the first few meters, i.e. the inner stations go from 2.7 to 25 PSU in 3 meters. Salinity values increase steadily seawards, this stratification is also maintained by the inputs from the lateral entrance of freshwater along the whole length of the fjord, with relatively low values of salinity at the mouth of the fjord. This intense halocline extends through all the fjord from 10 to 25 m deep; however its intensity varies in time due to the variability of the river discharge and tidal variability (Guzman and Silva 2002).

The salinity on the fjord presents a nearly steady situation during the whole year due to the high river discharges. The surficial low salinities are found also in the outer reaches of the fjord because due to the strong stratification mixing with deep water is minimal (Figure 8). Deep waters permanently maintain salinity values typical of the Chilean Inland Sea of around 31 PSU. In the horizontal, the interseasonal difference would consist in the extension of the fresh water plume seawards. Model results agree with the features described for the salinity; these features are shown in Figure 9.

Temperature in the fjord also shows a steep gradient in the surface waters. Below the thermocline typical values for the fjord along the year are around 10-11 ^{o}C. On the other hand, surface waters present a seasonal range due to atmospheric heating and cooling, of 4 to 14 ^{o}C, and the rivers temperature, between 1.5 and 12 ^{o}C (Table 2). This wide range of temperatures would be important for the ecological diversity and the appearance of HABs as is described in the chapter "Ecological Conceptual Model for a Southern Chilean fjord: the Aysén Fjord". Below the thermocline near the head of the estuary at around 25 m depth a temperature inversion is present . This phenomenon is more common during spring-summer conditions though it can also be observed in winter. In the innermost stations during spring-summer conditions the typical decreasing profile is disrupted at 25 m by a mass of colder water. Also in the data collected by SalmonChile on the 10th of February of 2007, under summer conditions, when the river discharge is lower than during wintertime show this watermass with different temperature trapped in the subsurface (Figure 10). The seasonal variation in surface temperatures is larger in the fjord water than in the nearby coastal water as can be seen in Figure 11. During summer temperatures are relatively similar over the study area whereas during winter a clear gradient occurs between the fjord waters and coastal water with differences of 5 ^{o}C.

Model results are quite in agreement with the observations (Figure 12) with different temperatures and inversions also observed in the model results. During summer (Figure 12 Right) subsurface water would correspond to warmer water that enters forced by the tide underneath the surface freshwaters. During winter (Figure 12 Top), water with higher temperature than the defined range, around 0.1 ^{o}C over the surrounded water, and this water mass would be flowing seawards. The mechanism proposed for this characteristic is that during winter due to the low temperatures on the surface and of the river discharge a cold thin layer is formed that covers a warmer water that maintains the heat due to the high stratification and as was commented with regard the residual currents their exit would be through the bottom layer of circulation loosing its temperature as travels through colder waters. However, this temperature minimum seems to not influence the density profile; in fact the Aysén fjord could be regarded as a nearly uniformly stratified watermass (Figure 13). The proposed explanation for this is the contrary effect that the salinity profile has on density.

4 DISCUSSION AND CONCLUSIONS

The occurrence of harmful algal blooms (HABs) has been related to years that presented temperatures higher than the average for the Aysén fjord. Three species found in the area are related to the production of toxins: *Alexandrium catenella* (PSP- paralytic shellfish poisoning), *Dinophysis acuta* and *D. acuminata* (DSP- diarrhetic shellfish poisoning). During warm years the proportion of toxic and harmful microalgae increases in the total phytoplankton population.

Toxic dinoflagellates appeared in temperature ranges between 10.5 and 14.5 ^{o}C. (Cassis et al. 2002). For this reason, it is relevant to characterise the hydrodynamics and processes regulating temperature and salinity. Model results and analysed data suggest a three-layered circulation with a top layer dominated by river discharges that presents temperatures and salinities different from the surrounding coastal waters with the differences being more intense during winter when river flows are larger. Under this thin top layer appears an incoming current that penetrates the system mainly tidally forced, providing waters with temperatures and salinities typical for the Chilean Inland Sea. The entrance of these waters results during winter in a tongue of warmer water that is trapped between colder layers and that during summer, as the surface is heated, flows outside as part of the third layer of circulation which is mainly seawards.

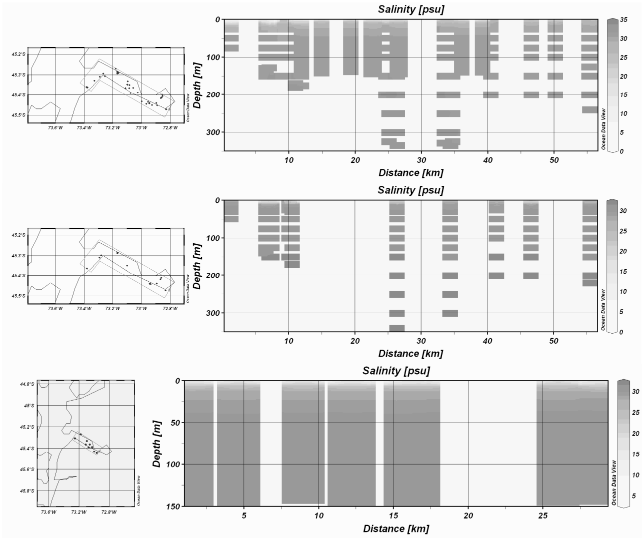

FIGURE 7: *Salinity profiles along the Aysén Fjord for the CIMAR spring-summer cruises (Top), winter cruises (Middle) and Intesal campaign in February 2007 (Bottom). Distance is measured seawards from the innermost station.*

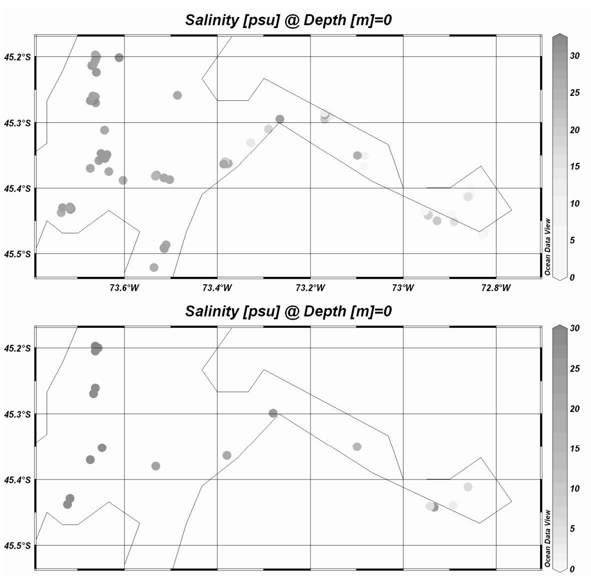

FIGURE 8: *Surface salinity in the Aysén Fjord and the adjacent part of the Moraleda Channel during the campaigns carried out in spring-summer (Top) and winter (Bottom) conditions.*

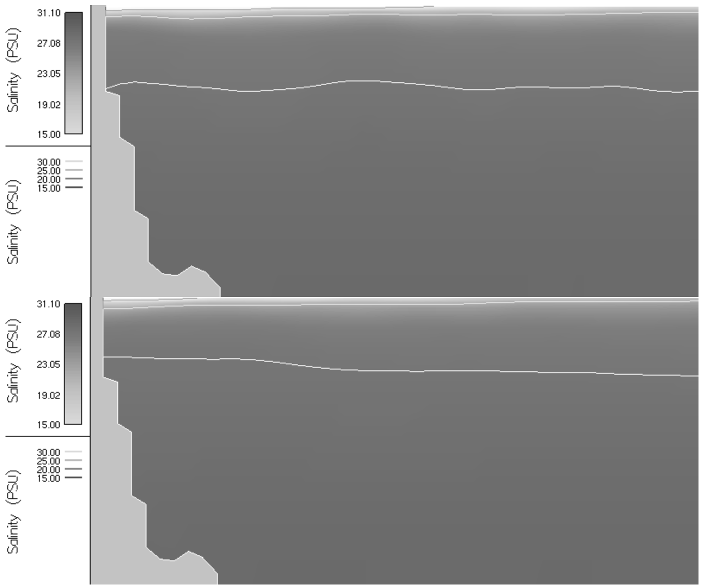

FIGURE 9: *Modelled salinities for the months of July (Top) and January (Bottom).*

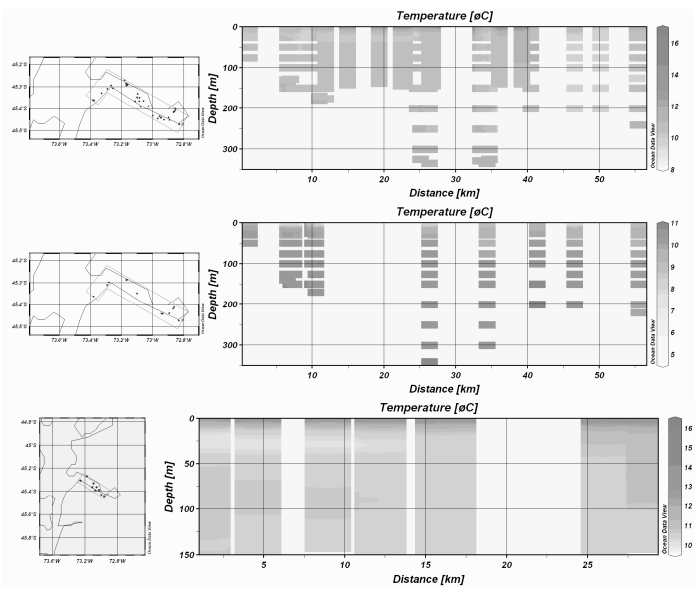

FIGURE 10: *Temperature profiles along the Aysén Fjord for the CIMAR spring-summer campaigns (Top), winter campaigns (Middle) and Intesal Campaign in February 2007 (Bottom). Distance is measured seawards from the innermost station.*

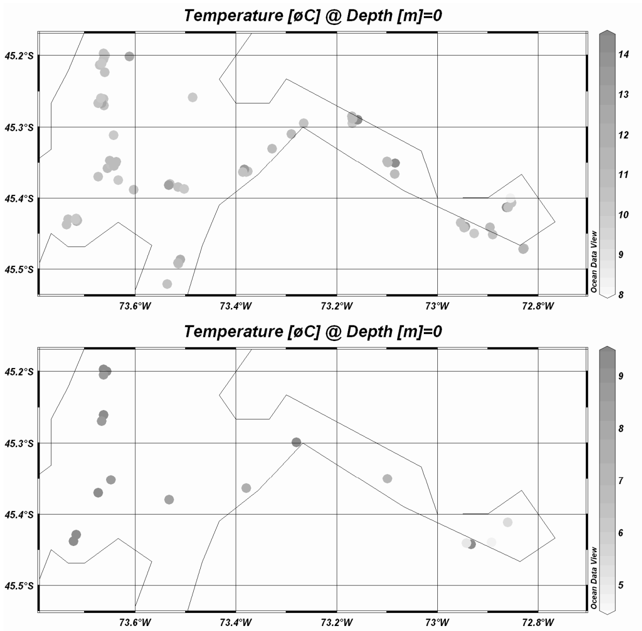

FIGURE 11: *Surface temperatures in the Aysén Fjord and the adjacent part of the Moraleda Channel for the campaigns carried out during spring-summer (Top) and winter (Bottom) conditions.*

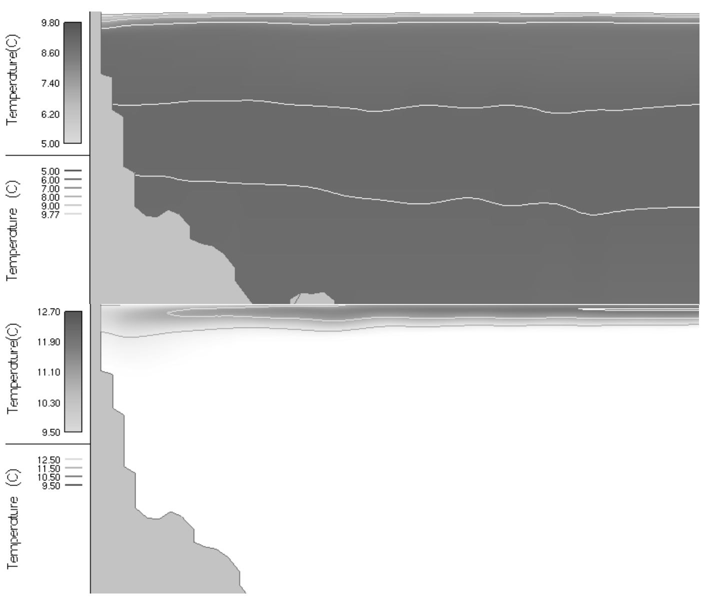

FIGURE 12: *Surface modelled temperatures for the months of July (Top) and January (Bottom).*

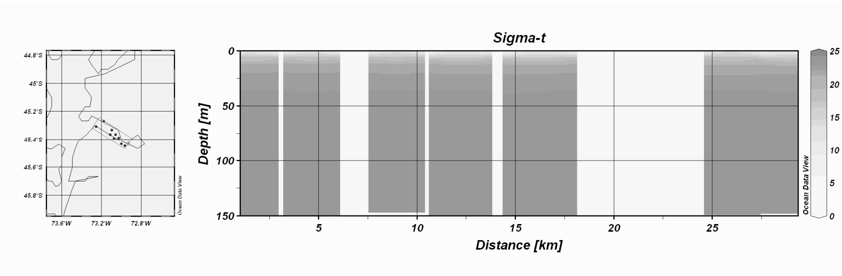

FIGURE 13: *Density profiles along the Aysén Fjord, distance is measured seawards from the first station (Source: SalmonChile).*

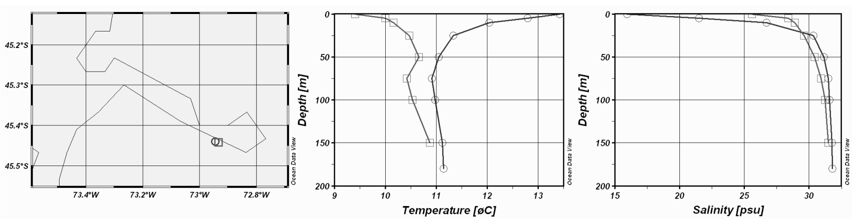

FIGURE 14: *Temperature and salinity profiles during a summer cruise (CIMAR 4, circles) and a winter cruise (CIMAR 9, squares).*

Figure 14 represents temperature and salinity profiles for two stations located at similar locations in the fjord collected during CIMAR cruises in different seasons. The differences in surface salinities and in temperature profiles can be clearly seen. Salinity values are higher during summer due to lower freshwater discharge while the vertical structure of temperature shows an intermediate water layer that breaks the typical temperature decrease.

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ECOLOGICAL CONCEPTUAL MODEL FOR A SOUTHERN CHILEAN FJORD: THE AYSÉN FJORD CASE STUDY

F.J. CAMPUZANO, V. MARIN, A. TIRONI AND P. CHAMBEL-LEITÃO

1 INTRODUCTION

The geographical location of each fjord conditions the ecological processes taking place in the system because the factors involved such as river discharges, including their origin and associated loads, and atmospheric conditions act locally converting them into singular study sites. In this paper we describe a conceptual and numerical ecological model for the Aysén fjord (Southern Chile), one of the study sites of the ECOMANAGE Project. The Aysén fjord is where the main human settlements of the XI Region of Chile are located and thus is a target of possible ecological perturbations including harmful algal blooms (HABs) by phytoplankton species harmful to salmon farming such as toxic dinoflagellates, which have been increasing their presence and distribution range in recent years (Muñoz et al. 1992).

2 MATERIAL AND METHODS

Most of the data used for the Aysén study area come from the different CIMAR-FIORDOS campaigns (hereafter, CIMAR). These campaigns consist of oceanographic cruises along the whole system of southern Chilean fjords funded by the Chilean Navy's Hydrographic and Oceanographic Service (SHOA, its Spanish acronym). The CIMAR campaigns are focused on different studies of the Chilean southern fjord system; the Aysén fjord was included in five different CIMAR campaigns. Most of these campaigns were divided in two cruises (E1 and E2).Table 1 summarises the period when each campaign took place and Table 2 the variables measured in each campaign. Due to the scarcity of data those data have to be grouped into spring-summer conditions and winter conditions. In addition to the CIMAR cruises, the Salmon Technological Institute (INTESAL, its Spanish acronym) which is part of the Salmon Industry Association, SalmonChile (`http://www.salmonchile.cl/`), made available temperature, salinity, density and oxygen vertical profile data collected on the 10th of February 2007 on nine stations along the Aysén Fjord ((`http://pronosticos.salmonchile.cl/`) (Figure 1).

2.1 Modelling efforts

The MOHID water modelling system Module Water Quality has been coupled to the hydrodynamical model described in the Chapter "Hydrodynamical vertical 2D model for the Aysén fjord" in order to sketch the ecological processes taking place in the Aysén fjord. The Water Quality Module is based on an eutrophication model developed by the EPA (U.S. Environmental Protection Agency) (Bowie et al. 1985). Initially, the model simulated the oxygen, nitrogen and phosphorus biogeochemical cycles, including both inorganic and organic forms, but recently the silica cycle has also been included. The concentrations of these nutrients along with other environmental factors such as light availability and temperature would determine the dynamics of the primary producers, in this case phytoplankton and diatoms, and indirectly the zooplankton dynamics as well. In order to model the ecological processes taking place in the Aysén fjord, it was to characterise the Aysén waterbody with two open boundaries corresponding to the Aysén River loads and the open entrance that connects the fjord with the Moraleda Channel in the Chilean Inland Sea. In order to force river discharges with realistic local values despite the lack of observed data, the Soil and Water Assessment Tool (SWAT) model (Arnold and Fohrer 2005) has been applied to the Aysén River catchment. As defined by their creators, the model consists of a river basin scale model developed to quantify the impact of land management practices in large, complex watersheds. An application of the SWAT model, on the Aysén catchment (Yarrow and Leitão 2007), produced the monthly nutrient values summarised in Table 3. The oxygen concentrations presented in the same table were obtained from seasonal data presented in CADE-IDEPE (2004). The river discharge has been implemented dividing its flow over the top six cells of the land boundary.

Cohesive sediment loads from the river were included on the model though observed data are absent. A constant concentration of 25 $mg\ l^{-1}$ was assumed for the river discharge. Concentrations along the fjord strongly depend on river flow, with the river plume presenting higher concentrations and extending further during winter, coinciding with high river discharges. The Aysén fjord initial conditions and for its coastal open boundary have been characterised by using vertical profiles from station 77 of the campaign CIMAR 1 (table 4). This station has been chosen because of its location near the mouth of the Aysén fjord and also because during the CIMAR 1 campaign were collected simultaneously more water properties. The boundary cells would be reading the boundary conditions from the interior, obtaining at each moment the immediate values resulting from the interaction of the initial values and the forcing.

The Mohid model atmospheric module has a global model of radiation that changes the emitting radiation from the high atmosphere that can be modulated by the local climatology. In the case of Aysén, relative humidity, air temperature and cloud cover was obtained from the Meteorological Annals of Chile for the years 1968-1969 at Puerto Aysén (Table 5) to characterise the Aysén fjord climatology. To favour the exchange of heat and gases taking place at the air-water interface also it was included winds collected during the CIMAR 7 campaign in 2002, as explained in the Hydrodynamical chapter. The model was previously run for a period of a year to adjust the initial values to the atmospheric, river and boundary forcing. The model results shown in this chapter correspond to the subsequent year of simulation.

3 DATA AND MODELLING RESULTS

3.1 Abiotic variables

Oxygen concentrations present their maximum values, close to 12 mg/l, at the surface due to exchange with the atmosphere favoured by the wind interaction and the entry of oxygenated river waters (Table 3). Though a minimum appears around 100 m, this could be explained by the deposition at the bottom near the river mouth of the organic loads at the head of the fjord where a platform stops at around 100 m. Through remineralisation processes oxygen is consumed nearly to depletion levels, under 2 mg l^{-1}. This mass of water with low oxygen levels is transported to the rest of the estuary at the same depth (Figure 2). Also is in the head of the fjord where material allochtonous coming from land is mainly deposited thus consuming oxygen near the bottom. As was observed on the residual flux velocities there are water exports from this area to the fjord waters which can be observed on Figure 2 as a plume with low oxygen concentrations. This process is permanent though during winter, levels the extension of this plume can be greater because of the increase of loads into the system, as the model results show (Figure 3).Oxygen concentrations are higher on the superficial fjords water than in the vicinity coastal waters due to the entry of more oxygenated fresh waters.

TABLE 1: *Cruises of the CIMAR program in the study area.*

Cruise	Initial Date	End Date	Season
CIMAR 1	08-10-1995	11-11-1995	Spring
CIMAR 4 E1	26-09-1998	09-10-1998	Spring
CIMAR 4 E2	25-02-1999	08-03-1999	Summer
CIMAR 7 E1	07-07-2001	21-07-2001	Winter
CIMAR 7 E2	12-11-2001	27-11-2001	Spring
CIMAR 8 E1	01-07-2002	26-07-2002	Winter
CIMAR 8 E2	15-11-2002	28-11-2002	Spring
CIMAR 9 E1	05-08-2003	25-08-2003	Winter
CIMAR 9 E2	03-11-2003	21-11-2003	Spring

TABLE 2: *Variables collected during each campaign.*

Campaign	Temp	Sal	Oxygen	Ammonia	Nitrite	Nitrate	Phosphate	Silicate	Chl *a*
CIMAR 1	X	X	X	X	X	X	X	X	X
CIMAR 4 E1	X	X	X			X	X	X	X
CIMAR 4 E2	X	X	X			X	X	X	X
CIMAR 7 E1	X	X	X	X		X	X	X	X
CIMAR 7 E2	X	X	X	X		X	X	X	X
CIMAR 8 E1	X	X	X			X	X	X	X
CIMAR 8 E2	X	X	X			X	X	X	X
CIMAR 9 E1	X	X	X			X	X	X	X
CIMAR 9 E2	X	X	X			X	X	X	X
SalmonChile	X	X	X						

TABLE 3: *Monthly values of nutrients and oxygen imposed on the Aysén River discharge.*

Month	Flow (m^3s^{-1})	Oxygen (mg l^{-1})	Nitrate (mg l^{-1})	Nitrite (mg l^{-1})	Ammonia (mg l^{-1})	Organic N (mg l^{-1})	Phosphate (mg l^{-1})	Organic P (mg l^{-1})
January	325.86	12.2	0.1262	0.0044	0.0113	0.0042	0.0076	0.0003
February	249.45	12.2	0.0808	0.0052	0.0128	0.0049	0.0080	0.0003
March	336.82	12.2	0.0885	0.0052	0.0142	0.0061	0.0073	0.0004
April	475.12	13.1	0.0856	0.0071	0.0230	0.0155	0.0096	0.0010
May	655.63	13.1	0.0916	0.0111	0.0437	0.0379	0.0152	0.0028
June	731.80	13.1	0.1089	0.0104	0.0409	0.0354	0.0150	0.0026
July	729.44	11.9	0.1385	0.0267	0.0972	0.0779	0.0347	0.0051
August	814.96	11.9	0.1136	0.0197	0.0700	0.0537	0.0268	0.0034
September	636.33	11.9	0.0877	0.0101	0.0345	0.0274	0.0144	0.0019
October	542.83	10.8	0.0752	0.0071	0.0227	0.0155	0.0104	0.0010
November	436.88	10.8	0.0623	0.0048	0.0135	0.0068	0.0075	0.0004
December	439.44	10.8	0.0578	0.0043	0.0114	0.0049	0.0066	0.0003

TABLE 4: *Vertical profiles of station 77 of CIMAR 1 used for characterising the Aysén fjord watercolumn.*

Depth (m)	Salinity (PSU)	Temperature (ºC)	Oxygen (mg l^{-1})	Nitrate (mg l^{-1})	Nitrite (mg l^{-1})	Ammonia (mg l^{-1})	Phosphate (mg l^{-1})	Silicate (mg l^{-1})
2	13.4558	10.8477	10.6957	0.0056	0.0018	0.0017	0.0016	2.4435
5	25.8857	10.1496	8.9678	0.1555	0.0028	0.0038	0.0375	0.5617
10	29.1555	9.5800	8.1110	0.1765	0.0045	0.0063	0.0443	0.4775
15	29.4397	9.4941	8.4395			0.0094		
20	29.5993	9.4922	8.2396			0.0074		
25	29.6338	9.5059	7.9682	0.2045	0.0014	0.0060	0.0480	0.4494
50	30.4125	9.8220	6.8544			0.0021		
75	30.7126	9.7861	7.0686	0.2395	0.0028	0.0021	0.0566	0.5536
100	30.9620	9.8179	7.5256	0.2283	0.0024	0.0021	0.0560	0.5617
150	31.0517	9.5745	7.9972	0.2255	0.0015	0.0021	0.0560	0.6741
200	31.0791	9.5326	7.6112	0.2003	0.0017	0.0021	0.0529	0.4775

On the other hand, nitrogen species on surface waters are nearly depleted for nitrate, nitrite and ammonia. In the case of nitrate, it appears to be nearly absent from the surface water of the fjord (Figure 4), mainly consumed by primary producers where light conditions favour their growth. Nitrate concentrations on the surface are higher during winter due to a higher river flow and slightly higher concentrations of this compound on river water, along with the adverse conditions for primary producers. Under the pycnocline, organic particulate matter tends to accumulate and to undergo remineralisation processes where nitrate is released and transformed into ammonia, nitrite being an intermediate in the process. The presence of nitrite indicates that transformation between nitrate and ammonia is taking place. In this sense, surface waters of the fjord present lower values of nitrate when compared with the waters of the Moraleda Channel nearby. Model results (Figure 5) reproduce the same pattern of concentrations and show that during summer the layer where nitrate concentrations are nearly depleted increases maybe due to better light conditions. Prado-Fielder (2000) considers that nitrite concentrations are, however, due to nitrification processes that would convert ammonia into nitrite.

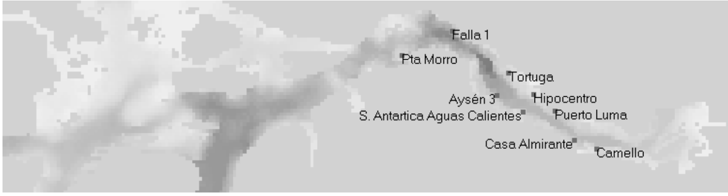

FIGURE 1: *Location of Intesal-SalmonChile vertical profiles.*

In the case of ammonia, only near the fjords head where several rivers discharge, this nutrient is present in the surface waters (Figure 6). Ammonia concentrations in the surface extend further due to higher river discharges during winter. Padro-Fielder (2000) also considered that differences in ammonia concentration between the continental fjords and the open channels correspond to a higher terrestrial input of organic nitrogen and the limited water exchange in these channels closed on one of its sides. Ammonia concentrations present a similar vertical distribution as the nitrate with subsurface maximum values due to remineralisation processes (Figure 7). However, the number of available ammonia observations is too small to reach a definitive conclusion.

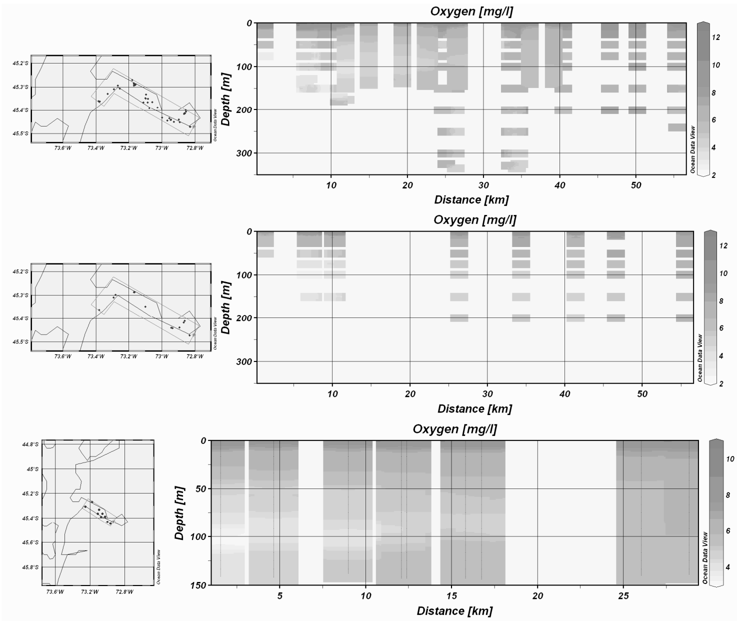

FIGURE 2: *Oxygen concentration profiles along the Aysén Fjord for the CIMAR spring-summer campaigns (Top), winter campaigns (Middle) and Intesal Campaign on February 2007 (Bottom). Distance is measured seawards from the innermost station.*

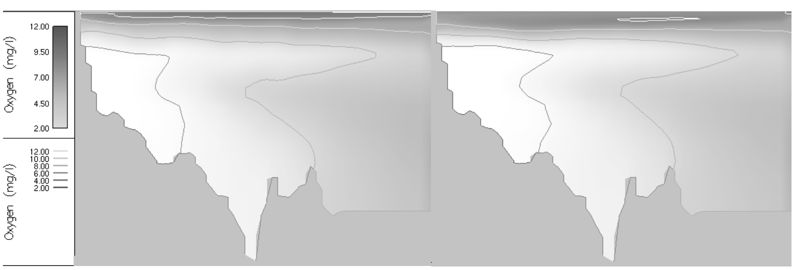

FIGURE 3: *Modelled oxygen concentrations for the months of July (Left) and January (Right).*

As in the case of nitrate, inorganic phosphorus concentrations are higher near the bottom of the fjord (Figure 8) due to remineralisation processes taking place on the sediment and to the low loads from the river. During spring/summer conditions phosphate is depleted in the fjords head, while during winter low levels are present in the top layer and concentrations are higher than in summer conditions. The fjord would act as a whole as a net exporter of phosphorus to the adjacent coastal waters. Model results (Figure 9) reproduce this same pattern of concentrations.

On the other hand, silica concentrations reach their maximum concentrations on the surface due to the river being the main source (Figure 10). Thus the maximum concentrations are found near the river mouth and decrease along the fjord and the value would be modulated by river flow. In the model a second maximum appears near the bottom due to remineralisation (Figure 11).

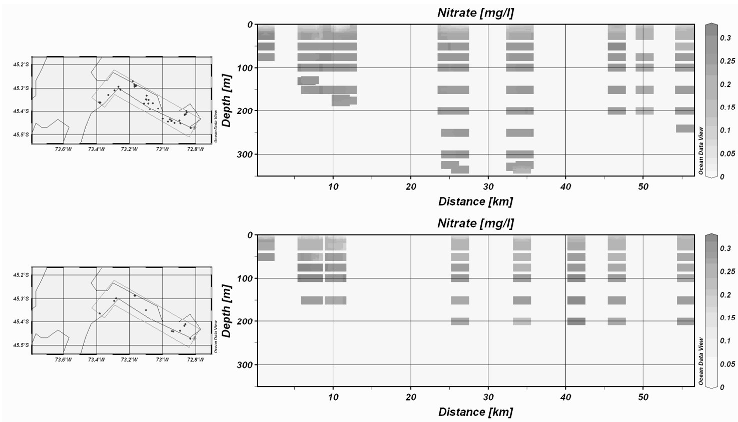

FIGURE 4: *Nitrate concentration profiles along the Aysén Fjord for the campaigns carried out during spring-summer (Top) and winter conditions (Bottom). Distance is measured seawards from the innermost station.*

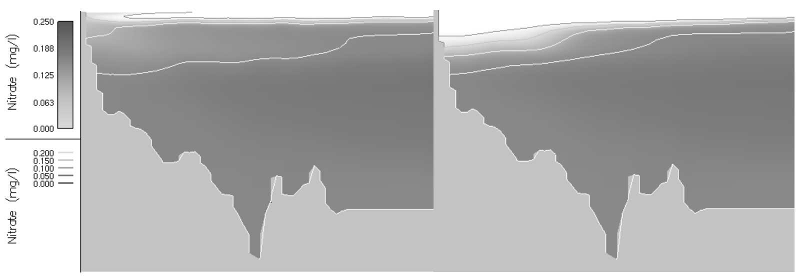

FIGURE 5: *Modelled nitrate concentrations for the months of July (Left) and January (Right).*

TABLE 5: *Monthly averaged atmospheric conditions from the Meteorological Annals of Chile (1968-1969).*

Month	Temperature (ºC)	Relative Humidity (%)	Cloud Cover (0-1)
January	14.15	88.17	0.792
February	12.30	89.33	0.879
March	11.50	88.33	0.692
April	9.35	92.33	0.810
May	7.65	93.00	0.888
June	4.75	92.17	0.790
July	4.55	93.00	0.858
August	5.60	91.00	0.794
September	7.00	85.83	0.848
October	8.00	88.17	0.796
November	11.30	86.17	0.842
December	12.85	85.00	0.775

3.2 Biotic variables

There is a gap in phytoplankton communities' publications in the southern Chile area (Cassis et al. 2002), with most of the studies focused on local phenomena and/or small areas. Diatoms are the dominant form of phytoplankton at the head of the Aysén fjord with 74 species out of the 106 recorded during the period 1993-1998, showing specific composition differences according to the origin of the waters (Cassis et al. 2002). Estuarine taxa stand out due to their abundance and constant presence and marine taxa because of their number of species. Estuarine species dominate the fjord with this dominance sometimes being interrupted by the entrance of continental and marine species (Cassis et al. 2002). Higher relative abundances of phytoplankton were found during summer. Biological changes in the surface layer at the head of the Aysén fjord are related to the fresh water inputs. Small *Skeletonema costatum* dominate the microphytoplankton level with very low diversity that only increases at the head of the fjord due to the presence of continental species (Avaria et al. 1997, Cassis et al. 2002).

Avaria et al (2007) during their sampling in the CIMAR 1 campaign found a good relationship between the algal biomass and the algal pigments especially in the Aysén fjord. Phytoplankton observations during that cruise in the Aysén fjord gave densities ranging between 60×10^3 and 743×10^3 cells l^{-1}, algal biomass concentrations between 0.1 and 0.8 mg l^{-1} and Chlorophyll a contents between 0.7×10^{-3} and 4×10^{-3} mg l^{-1} (Avaria et al. 1997). Using the maximum scales of the graphs that Avaria et al (1997) calibrated for showing the mentioned good relationship, a C:Chla factor of 225 has been obtained. Phytoplankton biomass concentrations are in agreement with the results obtained with the Mohid model water quality. They also found a negative relationship between phosphate and nitrate concentrations with the abundance of *Skelotonema costatum*. It was also observed the absence of dinoflagellates.

Euphotic depth on average was around 22 m though photosynthetically active radiation (PAR) was available down to 50 m depth. The mixing layer reaches at least 50 m depth and is larger than the euphotic area, thus, the mixing layer depth would compensate the lack of available light below 22 m (Pizarro et al 2005). The model results suggest that diatoms would bloom on the surface due to nutrients and light availability (Figure 12). The diatom blooming species would present maximum concentrations during summer enhanced by higher temperatures and existing light conditions. Below the surface a maximum of biogenic silica would appear due to diatoms respiration and death.

According to the model results, other phytoplankton groups such as flagellates, as occurs with diatoms, would bloom during summer as temperature and light conditions would enhance its growing. Maximum values for the bloom were found subsurface (Figure 13), due to the availability of nutrients and light, as they are depleted on the surface. According to the values from CIMAR campaigns (Figure 14), the phytoplankton bloom seems to occur during summer as chlorophyll a concentrations are higher than during winter conditions. This fact is more evident in the inner parts of the fjord where higher subsurface distribution variation occurs when compared with the Moraleda Channel stations.

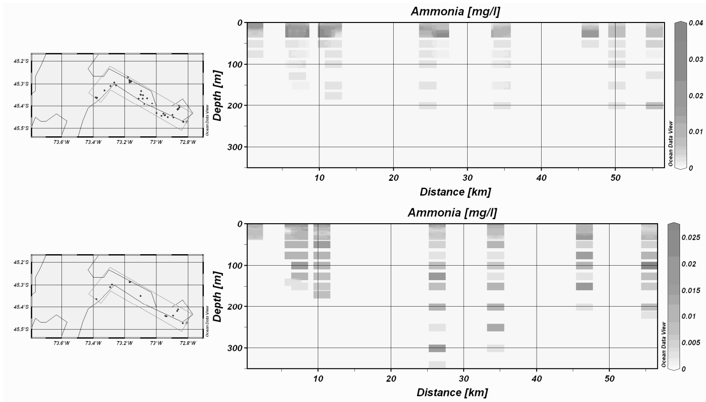

FIGURE 6: *Ammonia concentration profiles along the Aysén Fjord for the campaigns carried out during spring-summer (Top) and winter conditions (Bottom). Distance is measured seawards from the innermost station.*

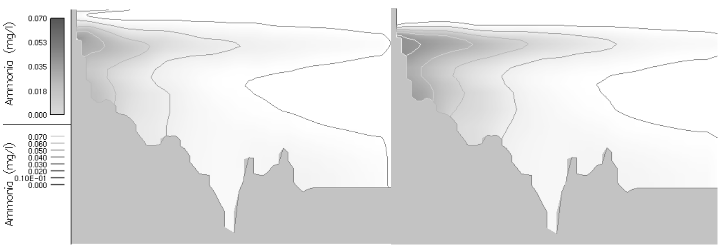

FIGURE 7: *Modelled ammonia concentrations for the months of July (Left) and January (Right).*

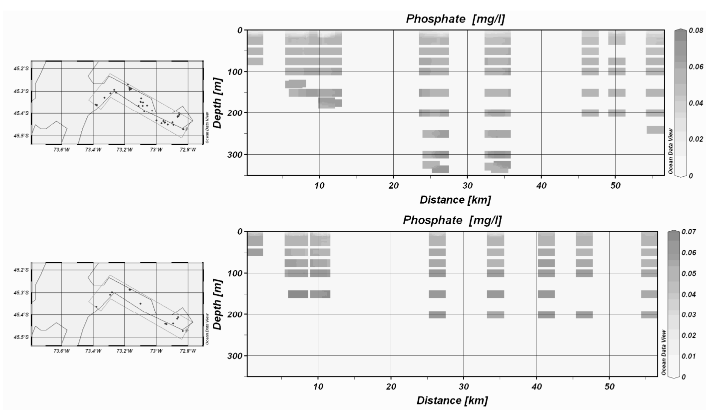

FIGURE 8: *Phosphate concentration profiles along the Aysén Fjord for the campaigns carried out during spring-summer (Top) and winter conditions (Bottom). Distance is measured seawards from the innermost station.*

FIGURE 9: *Modelled phosphate concentrations for the months of July (Left) and January (Right).*

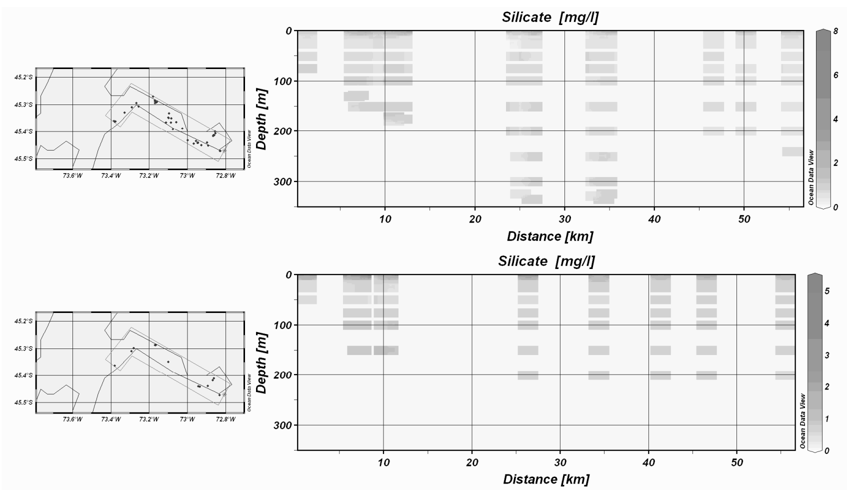

FIGURE 10: *Silicate concentration profiles along the Aysén Fjord for the campaigns carried out during spring-summer (Top) and winter conditions (Bottom). Distance is measured seawards from the innermost station.*

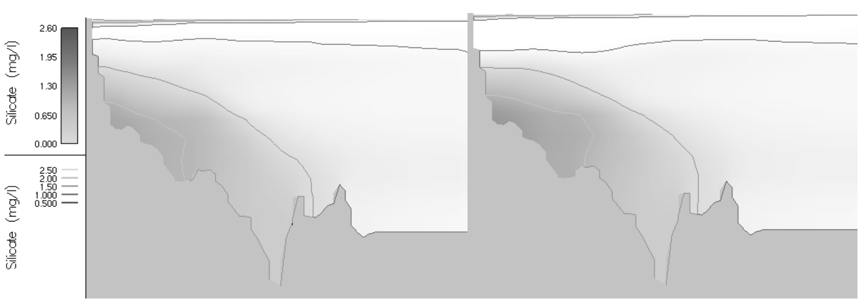

FIGURE 11: *Modelled inorganic silica concentrations for the months of July (Left) and January (Right).*

The Aysén fjord is located in one of the main areas of accumulation of zooplankton restricted to the Aysén fjord along with neighbouring Moraleda and Darwin channels (Palma and Silva 2004). However, this biomass abundance is not translated into species richness due to the spatial and temporal heterogeneity with marked gradients. Only few zooplanktonic species have been successfully adapted to the extreme oceanographic temperature, salinity and oxygen ranges (Palma and Silva 2004). During the CIMAR 8 campaign, zooplankton biomass mainly comprised copepods, chaetognaths and euphausids and highest biomass values in the system were found in the Guafo Mouth, the main connection between the Moraleda Channel and the open ocean, and the lowest values were observed in the Chile Inland Sea where the Aysén fjord is located (Palma and Apablaza 2005).

Though zooplankton is present in the model it is not controlling the phytoplankton biomass with the present parameterisation. Basically, its concentration is depending on the river discharge values that later is transported to the boundary where due to the boundary conditions they accumulate. As no data about zooplankton dynamics are available the function on regulating the primary producers' biomass is unknown.

4 DISCUSSION AND CONCLUSIONS

The model simulation reproduces the main physical-chemical features of the observed data. The sharp pycnocline acts as a barrier between the upper and bottom layers preventing mixing of water unless due to highly energetic processes. Surface waters are mainly dominated by atmospheric and river discharge processes while bottom water processes are related to sediment processes and coastal boundary processes. The main interaction between both layers consists of the particulate matter that the upper layer exports to deep waters. Surface waters are well oxygenated as the rivers discharging into the fjord present high concentrations of oxygen (Table 3). The nutrient loads from the Aysén River are of the same order of magnitude as the receiving environment, though slightly higher. In this sense, the Aysén River can be regarded as an oligotrophic river.

Perhaps, the most important effect would be the deposit of particulate matter in the adjacent fjord beds, also because the inner area would be the section of the fjord with the highest primary production. These deposits would consume oxygen through remineralisation, producing a plume of nearly anoxic waters and rich in inorganic nutrients that would extend trough the fjord transported by the deep water circulation. Oxygen is depleted near the bottom due to remineralisation of organic matter. In Figure 15 can be seen how nitrate concentrations increase landwards while oxygen concentrations decrease. Also in the inner stations can be appreciated higher concentrations of nitrite which is considered as an intermediate form of the whole process that goes from ammonia to nitrate; only the presence of nitrite concentrations indicates the active production of nitrate (Sverdrup et al. 1942).

According to the values from the CIMAR campaigns and model results the phytoplankton bloom seems to take place during summer with higher chlorophyll a concentrations than during winter, especially in the inner parts of the fjord. In this sense, also the production of the system would be conditioned by the river discharges, more due to the nutrient loads than to its flow and to light availability. Very little is known about zooplankton dynamics in the system so the control that can apply to the primary producers is practically unknown.

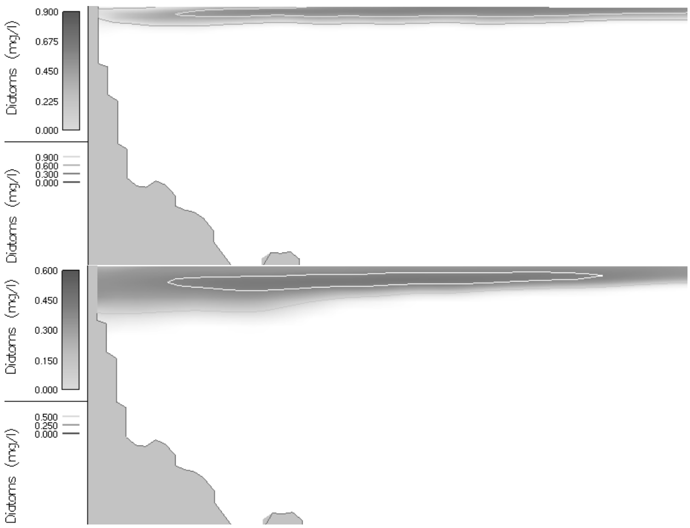

FIGURE 12: *Modelled diatoms concentrations for the months of July (Top) and January (Bottom).*

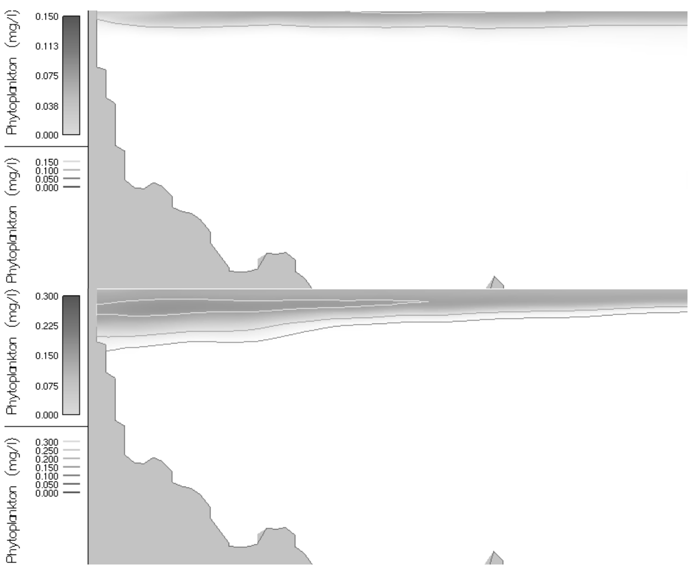

FIGURE 13: *Modelled phytoplankton concentrations for the months of July (Top) and January (Bottom).*

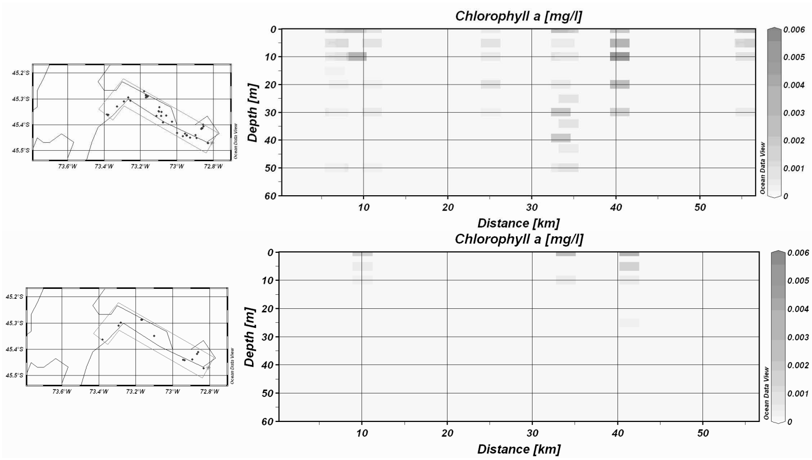

FIGURE 14: *Chlorophyll concentration profiles along the Aysén Fjord for the campaigns carried out during spring-summer (Top) and winter conditions (Bottom). Distance is measured seawards from the innermost station.*

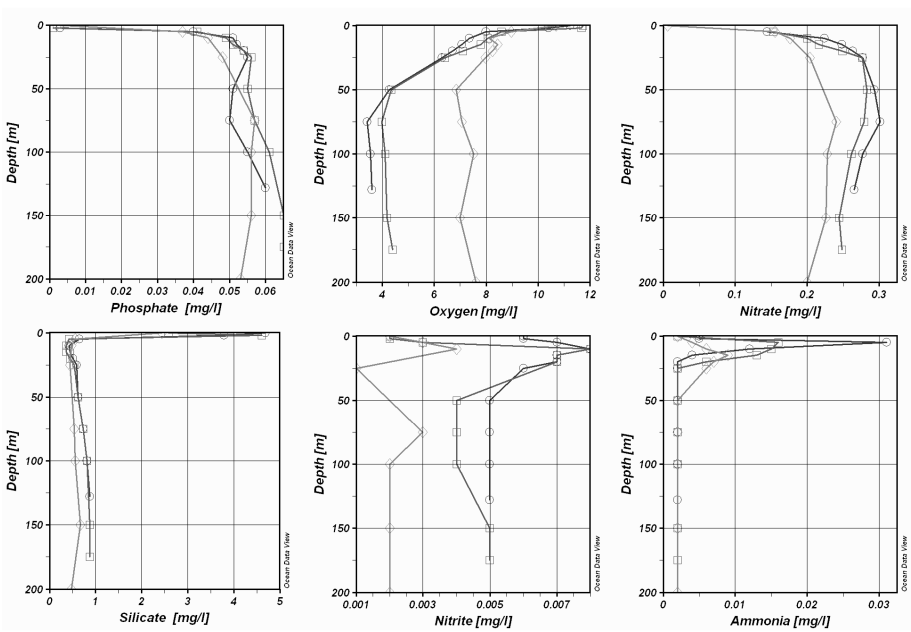

FIGURE 15: *Nutrient profiles obtained during the CIMAR1 campaign for the inner fjord (circles), middle fiord (squares) and near the Aysén fjord mouth (diamonds).*

In spite of the scarce data, knowledge of the system and model assumptions made it possible to establish a simple 2D vertical model that mainly reproduces the processes described by the bibliography and the general trends of the data. This model could serve as a first step to study the possible effects of modifying the entrance of organic matter as a product of a different management of the river catchment or other entrances of organic matter into the system.

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CONCEPTUAL, PHES-SYSTEM, MODELS OF THE AYSÉN FJORD: THE CASE OF SALMON FARMING

L.E. DELGADO, V.H. MARÍN, A. TIRONI AND P. BACHMANN

1 INTRODUCTION

In a previous chapter we have argued that the integrated management of coastal zones should be sensitive to the variety of perceptions that different social actors or stakeholders have on the ecological system being managed (Marín and Delgado, this volume). Our analysis of the economy and social conditions of Aysén suggested us that the main actors, in relation to the Aysén fjord, are: the local government and salmon farmers (Delgado and Bachmann, this volume; Bachmann et al. 2007). Consequently we studied the perception of these stakeholders by means of conceptual PHES-systems models of the Aysén fjord. Since they interact frequently on issues related to its management, we found that it was timely and indeed beneficial for them to know each other's perception of the ecosystem that generates the services they affect.

The basis and general description of the PHES-system approach can be found in the chapter by Marín and Delgado in this volume, on Marín et al. (2008) and Delgado and Marín (2005). The generation of conceptual models started by sending, two weeks in advance, a short list of questions that would be the basis of the brainstorming sessions. Several questions (e.g. the effects of fish hatcheries on the water quality of the fjord, the system components of salmon farming, and the effects of salmon on the fjord) were common to both groups for comparative purposes.

Conceptual models, coded using STELLA Research 9.0, were built after the brainstorming sessions. In the case of government experts, two models were built owing to the rather large number of people involved and then merged. In the case of salmon farmers one model was built. In all cases participants were encouraged to use the information generated during the previous session. Most results from the government stakeholder session have been reported by Marín et al. (2008) and the interested reader is encourage to read it. Here we analyze results only as they relate to the subsequent conceptual models of the Aysén fjord.

2 BRAINSTORMING SESSION RESULTS

The results of the brainstorming sessions showed that: while the local government perceives the fjord as an interactive ecological system, salmon farmers sees it as a market place where the "natural environment" only exists at the beginning of the market study. For example, when answering on the effect of hatcheries, government officers proposed two components (fish food and fecal material) and one process (excretion) as dominant. They further stated that although today there are only 14 hatcheries in the Aysén watershed, the salmon market has established that their number should grow up to 110 within the next 10 years. Furthermore, participants suggested that loosing fish food from hatcheries is not a problem (current losses are near 5-7% by mass), since this represents a cost that companies intend to decrease. They expressed concern over the fact that other components of the fish diet (e.g. zinc, antibiotics) are currently not regulated and that this may represent a problem in the near future. Salmon farmers, on the other hand, answered that the effect of hatcheries on environmental variables is unknown, that current legislation is unclear and that treating hatcheries as sources of pollutants is unjustified given that high value of the river flow/residues ratio. Finally they added that given these uncertainties, there is a clear need for more studies.

When confronted to the issue of salmon farming within the fjord, government officers identified several variables and processes that should be considered. Most government experts emphasized the potential negative effects of escaped salmons on the biological diversity of the fjord. Salmon farmers stated that since salmons have been in the area since 1969, when the Chilean government authorized its farming, they should be considered now (25 years later) as native fauna. Furthermore, they stated that most escaped salmons remain in the vicinity of the broken cages. They finally stated that the main causes for the escape of salmons are sea lions and robbery. The different perceptions of these two key stakeholders on the fjord were even clearer during the building of conceptual models.

3 CONCEPTUAL MODELS OF THE AYSÉN FJORD IN RELATION TO SALMON FARMING

Figures 1 and 2 show the conceptual models generated by government experts and by salmon farmers respectively. The first model divided the fjord in sections: head, center and mouth. Since salmon farming is developed mostly near the head, we have concentrated our modelling efforts in this part of the model. The government model (Fig. 1) shows the different processes related to the fate of nutrients in the system. From their point of view, salmon farming contribute with nutrients through salmon excretion and the waste of salmon food (see also Tironi et al., this volume). However, they also postulate that the government can influence these two processes by means of coastal zone management schemes. Thus, government proposes that they can have an impact on the two most important human nutrient sources: salmon farming and urban activity. Potential regulatory mechanisms are: secondary water quality norms and waste disposal legislation.

The environment, for salmon farmers, which represents a large portion of the government's model, is confined to one single converter inside one process: market study (Fig. 2). It also appears influencing the production process through the environmental norms set by the government[1]. Once the market study is over, and they have decided the location for the farming of salmons, the environment "disappears" with the exception of governmental water

[1] An example is the Supreme Decree 90 which regulates water quality for industrial residues (`http://www.ist.cl/archivos/ds90.pdf`).

quality rules and regulations. The rest of the processes and variables in the salmon farmer's model are related to two main processes: production and exports which in turn affect and will be affected by two main state variables: salmon biomass and markets (mostly international).

It is worth noting that the urban environment appears twice in the model. On one hand, the production process is affected by the urban environment in the sense of providing skillful manpower and their necessary education (Urban environment II, Fig. 2). On the other, the high water quality requirements of salmon farming makes the urban environment an unwanted variable (urban environment I, fig. 2), which then affects the production process both through the logistics necessary to stay "neither too close, nor too near" and, in turn, affecting the spatial distribution of the farming centers. Sea lions and salmon robbery appears in their model as affecting the export process.

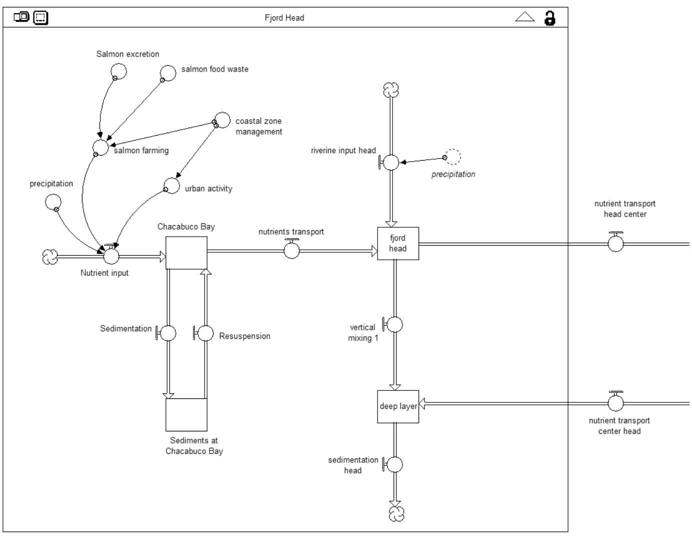

FIGURE 1: *Stella coded conceptual model (government PHES-system) of the Aysén fjord in relation to salmon farming. See text for details.*

4 LESSONS FROM AYSÉN FJORD PHES-SYSTEM MODELLING

We have shown, both through the development of brainstorming sessions as through the subsequent conceptual models, that the perceptions of the Aysén fjord ecosystem by two of the key stakeholder's related to salmon farming are very different. In one case (local government) the fjord is viewed as an interactive system composed of biotic and abiotic components and fluxes, one of them being salmon farming. In the other case (salmon farmers) the system is closer to a "market place" where most variables are related to the export process. Is it possible to link the two models? In this case the answer is positive, although we do not show the merged models for reasons of page space only. However, the interested reader can download it from the internet (`http://ecosistemas.uchile.cl`). In essence both views of the fjord ecosystem (or PHES-systems) are complementary. The government model lacks all ”market variables” and the salmon farmers model lack all ”ecosystem variables”. Thus merging in this case is simple. However, in order to use the merged product for integrated management both key stakeholders will have to learn from each other perspectives. This is indeed one of the most important findings from other groups generating frameworks for interdisciplinary modelling: the major problem when integrating and coordinating different stakeholders seems to be communication (e.g. Heemskerk et al. 2003). In this regard, the PHES-system approach can effectively be used as a communication tool among stakeholders for the benefit on integrated ecosystem management.

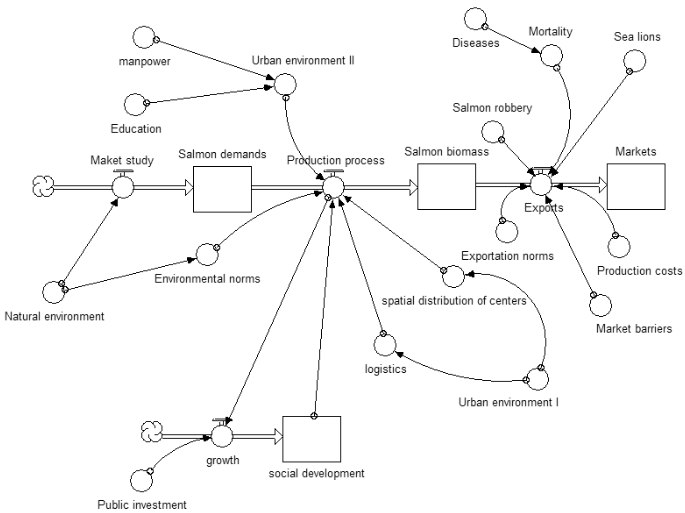

FIGURE 2: *Stella coded conceptual model (salmon farmers PHES-system) of the Aysén fjord in relation to salmon farming. See text for details.*

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A MANAGEMENT TOOL FOR SALMON AQUACULTURE: INTEGRATING MOHID AND GIS APPLICATIONS FOR LOCAL WASTE MANAGEMENT

A. TIRONI, V.H. MARIN AND F. CAMPUZANO

1 INTRODUCTION

Contemporary human pressures over world marine ecosystems could be unsustainable. Over 75% of the world natural fish stocks are considered fully exploited or over-exploited (FAO 2007). Direct impacts over biodiversity, habitat destruction, waste disposal and climate change indirect effects -and a possible synergy between these factors- (Harley and Hughes 2006) allow predicting the collapse of all harvested taxa by the year 2048, if we sustain today's trend of use and extraction (Worm et al. 2006). However, fish consumption has duplicated since 1960 and now it's the fastest growing food industry worldwide (FAO 2007).

In this context, fish farming -responsible of more than 70% of this growth- appears as a proper way to reduce human pressure over world natural fisheries. But, is this true for all types of aquaculture? Unfortunately no, farming of carnivorous species requires large food inputs and produces a series of local impacts on marine ecosystems, converting them in a mixed blessing for world fisheries (Naylor et al. 2000). Fish farming reduces the pressure over natural fisheries used for human consumption, but at the same time it raises the demand for some pelagic species (e.g Chilean mackerel) used for fish oil and flour, used in the elaboration of fish pellets. One of these forms of aquaculture is salmon farming (Naylor et al. 2000, 2003, 2005).

In general terms, the marine stage of salmon farming productive cycle consist of an accelerated and controlled growth of juvenile individuals, until they obtain an appropriated weight for their processing and commercialization. Fish are maintained in floating cages, settled in protected areas (like fjords, channels and inner seas), and fed with pellets made of a variable fraction of marine fish flour and oil. Aside from the pressure over pelagic fisheries, salmon farming produces a series of environmental impacts over marine ecosystems where the cages are installed. They are, in a short summary: (1) waste disposal over the water column and bottom, like food pellets, fish feces and antibiotics, (2) invasion by exotic fish species and pathogens every time fish escape from cages and, as a result of these processes, (3) habitat destruction (Naylor et al. 2003, 2005, Miranda and Zemelman 2002, Cabello 2006). These impacts could obscure salmon farming's contribution to reduce pressures on natural fisheries resources. In this context, it could be argued that salmon farming is now in a crossroad between been against or in favor of sustainable development of world fisheries. The path it shall take will depend on their fish flour and oil use policies (Naylor et al. 2000), and how it will manage local impacts related to their activities. It is toward the later that we have focused this work.

2 LAGRANGIAN MODELS AND THEIR USE IN SALMON FARMING MANAGEMENT

A common cause of most salmon farming local environmental impacts is particulate waste disposal in the form of food pellet and feces- to the water column and bottom. These particles usually deposit in the vicinity of cages, distressing the system in various ways: (a) artificial inputs of organic carbon, nitrogen and phosphorous, (b) increase in primary productivity, modifying bottom's community structure, (c) negative impacts over benthos biodiversity and (d) the formation of an anoxic layer in the sediments under the cages (Corner et al. 2006, Cromey et al. 2002, Findlay et al. 1995, Soto and Norambuena 2004)

A widely adopted management tool for this kind of particulate wastes is their dynamic simulation by means of lagrangian particle-tracking models (Cromey et al. 2002, Cromey and Black 2005). These models simulate the dispersion and sedimentation of particles in ocean's bottom. Coupling these with other numerical models, involving additional physical, biogeochemical and ecological process, could contribute to a more integral assessment of salmon farming local impacts (Corner et al. 2006, Cromey et al. 2002, Cromey and Black 2005, Panchang et al. 1997, Perez et al. 2002) . We implemented, as one of ECOMANAGE activities, a three-level nested, coupled circulation-lagrangian model to assess the fate of particles generated at salmon farming sites, one of the main economic drivers of the Aysén fjord region. We have applied this tool to the Chacabuco bay area inside the fjord.

3 METHODS

3.1 Hydrodynamic models

Hydrodynamic models utilized were developed with the open-source software MOHID Water Modeling System (Leitão et al., this volume). A 3-level nested modeling structure was chosen in order to simulate the complex tide signal of the fjord system, reduce numerical errors and to enhance model stability (Fig. 1). The first level, Fjords, is a barotropic, single-layer sigma model covering the northern part of Chilean fjords, between 41°S and 46°S, with a definition of 2.2 km. The main purpose of this level was to generate the tidal components, obtained from the FES2004 model (Lyard et al. 2006), for the lower level models. The second level in the nested structure (Aysen) is a model covering the area of the Aysén Fjord. It is also barotropic, but aside from incorporating tides from the Fjords model, it includes the three most important fresh water discharges (rivers) of the fjord. These two models have been described in detail by Marin and Campuzano (in press).

The third level, Chacabuco, covered the inner area of the Aysén Fjord. It is a baroclinic model, with a cartesian geometry of 11 vertical layers. Its resolution is approximately of 100 m, and it was over this level that the lagrangian particle-tracking sub-model was implemented. The numerical grid was generated using bathymetric data from Armada de Chile (Chilean Navy), using geo-statistic methods in order to generate a smooth bathymetry in areas where the point coverage was not accurate enough.

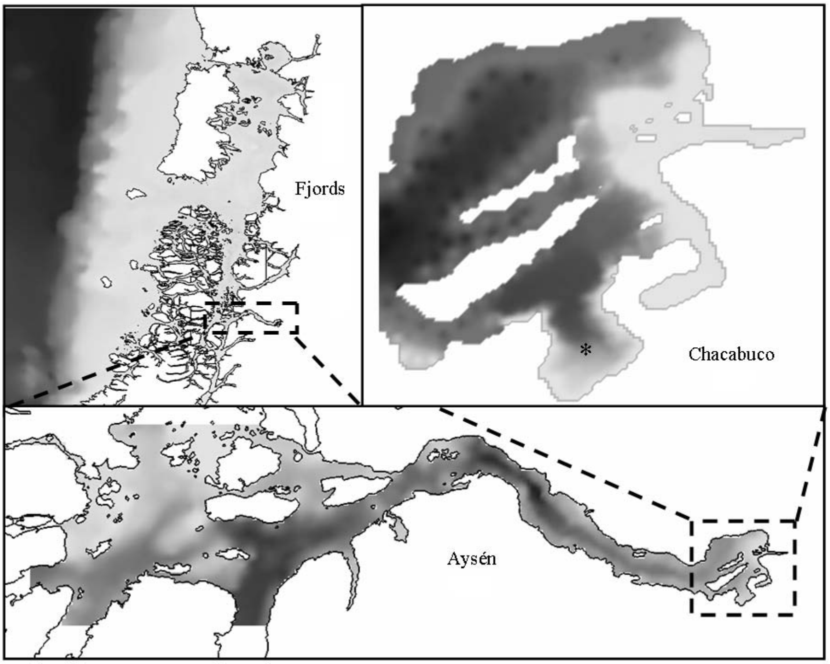

FIGURE 1: *Geographic area covered by the 3 level nested modeling structure in southern Chile. Grayscale areas represent each of the model grids. Chacabuco bay, where the particle-tracking model was mounted, is marked with an asterisk.*

The three models were initialized for 10 days with the purpose of stabilizing Aysen's Fjord water level, followed by a 25 days run to initialize water properties (temperature and salinity). Finally, with tide signal, temperature and salinity stabilized, a third 16-day run was implemented to simulate pellets dispersion in the bay area. Table 1 shows the main characteristics of each of the three models.

3.2 Lagrangian Particle Tracking Module

In what follows we describe the parameterization and assumptions used in relation to salmon farming loads and their dispersal in the water column.

Discharge volume: A fish density of 10 kg m^{-3}, for every 6000 m^{-3} cage, was considered for pellets output with a food/biomass ratio of 1.2. Only 5% of this load was considered to pass cage's depth and fall through the water column. We have further assumed that half of these wasted pellets were eaten by local marine fauna (Cromey et al. 2002). This pellet mass (in Kg) was then converted to particles assuming a caliber 2500 pellet (0.9 g per pellet, taken from Ewos Chile website, `http://www.ewos.com/cl/`, visited on 07/23/2007). The final number of particles was calculated as 450 particles per cage every two hours.

Sedimentation Velocity: Previous work has shown that sedimentation velocity depends on particle size (Perez et al. 2002) and that pellets suffer changes in their size as they descend through the water column (Chen et al. 1999). Here we have simplified this process applying a constant sedimentation velocity. The main purpose of this simplification was reducing calculation time and hardware requirements, given the rather large number of particles tracked in our simulation ($> 10^9$).

Salmon Production Cycle: Salmon farming involves different fish densities for each stage of the production cycle. For the purposes of this model we chose a fixed density (10 kg m^{-3}) corresponding to a high production stage. We subsequently tracked their discharges every 2 hours.

Consolidation: The model does not consider pellets consolidating in the sediments. There is no enough available information about sediments in the study area. A no-resuspension scenario was added to the sensitivity analysis to test for this assumption.

3.3 Sensitivity Analysis

Sensitivity analysis was done defining default values for every parameter tested (default run), and then adjusting them within the range found in literature to create sensitivity scenarios (Cromey et al. 2002, Panchang et al. 1997, Perez et al. 2002, Chen et al. 1999, Wiberg 2004). Seven scenarios were defined as shown in Table 2. The variable over which we perform the sensitivity test was the number of particles on day sixteen of simulation (N_{16}) in a specified area. The area was chosen after sampling the model for number of particles in a series of boxes over the whole bay. We chose the one showing greater variation between sensitivity scenarios. Model sensitivity (Sx) was evaluated as the change in "N_{16}" relative to changes in a model parameter "P" (Huntley et al. 1987), using equation (1).

$$Sx = \frac{(N_{16s,x} - N_{16def})/N_{16def}}{(P_{s,x} - P_{def})/P_{def}} \tag{1}$$

Where $N_{16s,x}$ is the value for N_{16} for the *x* scenario in the sensitivity analysis, and N_{16def} is the value for N_{16} on the default run. $P_{s,x}$ correspond to the value of parameter P for a given sensitivity scenario *x*, while N_{def} is the default value of the parameter.

3.4 Management tool

The management tool generated is a modified ArcView® 3.3 (ESRI Inc.) interface (programmed using AVENUE scripts) that shows, in a simple and user-friendly way, the combined results of the hydrodynamic and particle-tracking model in a Geographic Information System (GIS) environment. Custom buttons were added to show pellets dispersal, relevant hydrodynamic data and GIS coverage. Help text and button information were translated to local language (spanish) to improve user experience. During the development of the tool, local manager's capabilities and requirements were checked to improve its usage and to fulfill their information requirements.

4 RESULTS

The Chacabuco hydrodynamic model was validated with respect to salinity and water level. The model is capable of reproducing the typical halocline of the Aysen fjord, with the river flowing seaward through the upper level of the water column (Fig. 2). When comparing water levels given by the model with real values taken from mareographic station located inside the model, the results shows a good fit, with an r^2 of 0.94 (Marin and Campuzano, in press). Thus, the model is able to simulate the main characteristics of the estuarine system in the Chacabuco bay area. Therefore, the validated model was used to study the spatial dispersion of particulate wastes coming from salmon farming activities in the bay, using MOHID lagrangian module.

The results of the lagrangian particle tracking module (pellets dispersal after 16 days of simulation of five cages throwing 450 particles every 2 hour) is shown in Figure 3. The majority of the origins showed pellets dispersing mostly beneath the cages or in distances between 100 m and 500 m from the cage's center. The greater dispersal in our simulation reached 1100 m, not including isolated particles. In total, pellets covered between 17% and 27% of the Chacabuco bay area. Results for the sensitivity analysis, grouped by scenarios, are shown in Table 3. The most sensitive variables were those defining resuspension; specifically erosion and deposition shear stress. The management tool was delivered to local decision makers. Additionally, there is an online version of the tool (mapserver format: `http://ecosistemas.uchile.cl/ecomanage/resultados`). The tool has already been used in the process of generating new environmental regulations for the Aysen region (DGA, CONAMA, personal communication).

5 COMPARISON WITH OTHER MODELS

We have shown the development of a lagrangian particle tracking model that can be used by salmon farmers and local decision makers. When comparing model's output with previous work with lagrangian approaches (Cromey et al. 2002, Cromey and Black 2005), the results are qualitatively similar. DEPOMOD is the only model with its dispersion module validated, showing between 13% and 22% of difference between modeled and observed values (Cromey et al. 2002). Thus, we have used it as a reference in this comparison.

There are some important differences between MOHID and DEPOMOD that should be taken into account before comparing them. First, MOHID sedimentation velocity was fixed during the whole simulation ($1.28 \text{x} 10^{-1}$ m s^{-1}), while DEPOMOD uses random generated rates taken from a given range of values. The amount of particles simulated with MOHID ($4 \text{x} 10^9$) is five orders of magnitude above DEPOMOD ($7 \text{x} 10^4$). In fact, our volumes were close to real production values for particulate wastes discharges from Chilean salmon cages (EIA reports from salmon farmers POCH, 2004). Another important difference is the spatial and time scale of both models. While Cromey et al. (2002) used a grid definition of 10x10 m, covering an area of 0.25 km^2, here we used a grid of 100x100 m, covering an area of 147 km^2.

TABLE 1: *Main properties of the 3 nested model system.*

Model	Nesting Level	Grid Definition (km)	Geometry	Type	Discharges	Tides
Fjord	1	2.2	Sigma, 1 layer	Barotropic	No	FES2004
Aysén	2	0.5	Cartesian, 1 layer	Barotropic	Yes	Fjord Model
Chacabuco	3	0.1	Cartesian, 11 layers	Baroclinic	Yes	Aysén Model

TABLE 2: *Default values and sensitivity analysis scenarios. For every scenario, defaults parameters were modified to fit a range of values founded in literature.*

Parameters	Default values	Resuspension			Sedimentation		Random movement	
		E1	**E2**	**E3**	**E4**	**E5**	**E6**	**E7**
VARVELHX*	**0.02**	-	-		-	-	0.04	0.01
VARVELH*	**0.02**	-	-		-	-	0.04	0.01
Critical Shear Stress of Erosion	**0.02**	0.01	0.04	1×10^{-5}	-	-	-	-
Critical Shear Stress of Deposition	**0.004**	0.002	0.008	1×10^{-5}	-	-	-	-
Erosion Rate	**0.005**	0.0025	0.01	1×10^{-10}	-	-	-	-
Sedimentation velocity	**0.128**	-	-		0.096	0.064	-	-

*varvelhx y varvelh are both model parameters determining particle random movement

TABLE 3: *Results from the sensitivity analysis. Each scenario is represented by the relative amount of variation of a given parameter from their default values. Sensitivity was calculated based on Huntley (1986).*

Sensitivity Scenario	Sensitivity (x 10^3) as particle number			
	200%	**-75%**	**-50%**	**-200000%**
Resuspension (E 1-3)	-664.3		335.5	982.8
Sedimentation Velocity (E 4-5)		-1.8	107.1	
Random Movement (E 6-7)	90.5		128.6	

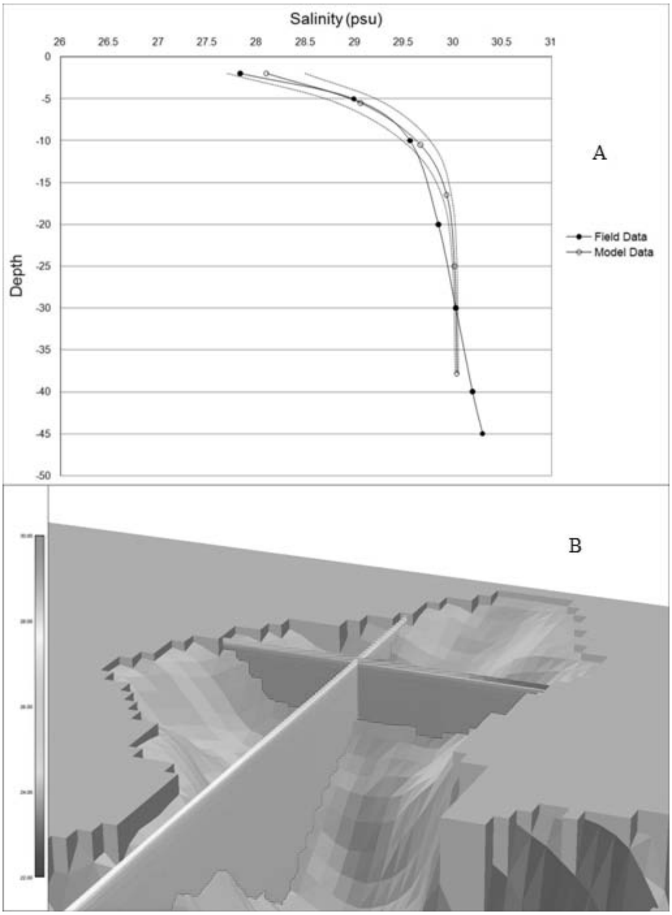

FIGURE 2: *Salinity values for model results (○) and available field data (●). The dotted lines represent standard deviation of model results.*

The time span of both simulations is also very different. Our model ran for sixteen days, with particle discharges every two hours for five origins, while Cromey et al. (2002) shows a 24 hour simulation with a single discharge. Despite the differences just mentioned, DEPOMOD and MOHID show similar results in their pellets' dispersal simulation. Cromey et al. (2002) showed that pellets dispersal occurred mainly beneath the cages (0-100 m), reaching a maximum distance for pellets dispersal of 200-300 m from the cage center. Our results, for comparable depths, show the same pattern, but with maximum dispersal distances of 300-400 m. Thus, the simulation of salmon farming particulate waste dispersal using MOHID lagrangian module shows congruent results when compared with previous experiences with similar approaches (Cromey et al. 2002) and in general, with previous simulations of particulate waste dispersal (Corner et al. 2006, Panchang et al. 1997, Perez et al. 2002).

FIGURE 3: *Final spatial distribution of our 16-days simulation. Darker particles represent the results from the default simulation, while grayscale particles symbolize particle dispersion from the sensitivity scenarios. White circles shows each cage location.*

6 A BROADER CONTEXT

Models, like the one we have developed here, do not gain enough relevance until they are consider as part of a wider strategy, thus allowing the application of their results outside academic circles. A management framework we think is able to provide this wider context is Integrated Coastal Zone Management (ICZM; Turner 1999, 2000) is one strategy where models can be used. This management framework understand the relationship between nature and society as a process where perturbations generated from the former alter some ecosystem functions, eventually affecting the flux of ecosystem services to society, generating negative impacts over it, in a cycle called Driver-Pressure-State-Impact-Response. ICZM also provides a good support for stakeholder involvement in any of the multiple stages of project development (Christie et al. 2005, de Araujo et al. 1999).

In this work we understand stakeholders involvement as a continuum from partial involvement, or cooperative research (e.g. information exchange agreements), to collaborative research, where scientist and stakeholders jointly develop research projects. In any point of this spectrum, recent studies show that participatory research is able to contribute to scientists and stakeholders more informed about each other and better engaged in projects' objectives and outcomes (Hartley et al. 2006). Furthermore, stakeholder involvement has the potential to increase their self-reliance and awareness of the issues being investigated, facilitating more equitable trade-offs between stakeholders with competing interests (de Araujo et al. 1999). From this point of view, our experience during the development of this work showed that stakeholders' involvement in the research process brings a series of benefits: (1) Considering the large field data required to set model's initial/boundaries conditions and for the validation process, government stakeholders participation -and their data bases- were a key element in the data gathering period, (2) the working relationship generated with local stakeholders allowed us to identify each of their administrative, technical and institutional capabilities, key information when generating public research reports and decision support systems to final users (Pedersen et al. 2005) and (3) the creation and definition of modeling scenarios was facilitated during this interaction, linking research objectives with stakeholders' specific interests (Hanson et al. 2006).

However, there's a huge distance between the theory and practice of ICZM, especially in the context of developing countries. ICZM requires an integrated approach to natural resource management, and strong institutions able to guarantee an equitable distribution of projects benefits among stakeholders, with the ability to resolve conflicts between partners, supervising the realization of conflict-solving agreements and assuring stakeholders participation (Oracion et al. 2005, Pollnac and Pomeroy 2005). None of these conditions are satisfied in Chile. First of all, Garces (2005) proposes that Integrated Management (IM) would not be possible given the reductionist approach taken by Chilean environmental institutions and legal framework. Furthermore, Chilean economic orthodoxy, based on a neoliberal economy with small-state, free-market priorities, sustained on raw material exports, privileges economic growth -measured as GDP increase- over other aspects of economic welfare, even showing a hostile attitude over more sustainable initiatives (Carruthers 2001).

On a potential scenario of ICZM, this historical predilection towards the generators of this economic growth on a national level (private companies, multinationals, sometimes the government itself) (Saez and Cerda 2007) over other more sustainable aspects has produced uneven relationships between stakeholders on the local (regional) and central (national) level, limiting government institution's capacity to solve potential conflicts. Finally, participatory processes are seriously constrained, mostly explained by; (1) stakeholder's apathy towards participatory measures and proceeding given recent historical and political developments (Carruthers 2001) and (2) a fragmentary, merely informative "top-down" approach by government institutions towards stakeholders participation (Bachmann 2007, Fraser et al. 2006). It's under this socio-political context that this work was meant to improve local decision-making processes involving salmon farming particulate waste management, strengthening regional environmental institutions. In the context of a developing country like Chile, we expect this to be one step towards ICZM.

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THE AYSÉN FJORD TSUNAMI OF APRIL 2007: UNEXPECTED USES OF CIRCULATION MODELS

V.H. MARÍN, L.E. DELGADO AND A. TIRONI

1 INTRODUCTION

One characteristic of the Chilean territory is its high frequency of earthquakes. The responsibility for the coordination of responses and management in case of natural disasters in Chile rests with the National Emergency Office (ONEMI, `http://www.onemi.cl`). When dealing with specific disasters, ONEMI seeks advice from experts and scientists through Scientific Technical Committees (STC). On January 22nd, 2007 a seismic crisis started in the Aysén fjord area. A series of earthquakes, sometimes referred as earthquake's swarms, occurred during a period of approximately four months. Given the steepness of the fjord's topography and the possibility of landslides that could in turn generate tidal waves within the fjord, ECOMANAGE scientists were invited to join ONEMI's STC for Aysén. On April 21st, 2007 a major earthquake (6.2 on the Richter scale) affected the Aysén region. This earthquake triggered a 7×10^6 m^3 landslide, mostly rocks, which, in turn, generated a 14 m tidal wave (SHOA 2007). Although the tsunami did not reach populated areas, it resulted in the dead of people which, at the moment of its triggering, were onboard small vessels within the fjord. Both national and local governments started a desperate search for the victims, in order to relief the pain of the affected families. At that point, ONEMI turned to ECOMANAGE posing a specific question: Was it possible to use the results of Aysén's circulation model as an aid in searching for the tsunami's victims? The remainder of this chapter is a summary our response.

2 MODELLING THE THREE-DIMENSIONAL CIRCULATION OF THE AYSÉN FJORD

The circulation of deep estuaries like fjords is highly three-dimensional (see Marín et al., this volume). In the specific case of the Aysén fjord, both data (Cáceres et al. 2002) and models show that there may be more than two layers which are influenced by the wind and the tidal cycle. Beyond this local complexity, the tidal signal that arrives in the fjord has to cross a singular seascape full of islands and small channels (Marín and Campuzano, in press). In order to model this complex signal a 3-level nested circulation model was implemented. Most details about the first level (FJORD), have been explained by Tironi et al. (this volume) and by Marín and Campuzano (in press). Since our main objective in ECOMANAGE was to understand and model the inner Chacabuco bay, we developed a second level nesting model which we termed AYSEN (Tironi et al., this volume). This second level would normally be 2D barotropic, since its only purpose would be to bring the tidal signal from the previous level (FJORD) into the target model (CHACABUCO). However, when ONEMI's request arrived it was clear that a three-dimensional model of the whole Aysén fjord was necessary. The reason for this was the processes that undergo a dead human body when falls in the water (explained to us by physicians from the Chilean Medical Legal Service), which makes them first sink all the way to the bottom, to re-surface again after a period of time that will depend on temperature to finally sink again permanently.

The modified AYSEN model was implemented with eleven Cartesian layers, seven of them in the first 50 m in order to accurately solve the circulation near the surface. Its grid size was 57x215, with a resolution close to 555 m. Its spatial extent and bathymetry are shown in Figure 1. Forcing for the AYSEN model corresponded mainly to the oceanic semidiurnal tidal signal and river discharges (see Tironi et al., this volume). The FJORD model (upper nesting level) was initialized for 15 days in order to stabilize the tidal signal. Then the AYSEN model was initialized for a month for the same reason. The potential advective patterns of the tsunami victims were modeled using MOHID's lagrangian module. The module was initialized for the three locations were victims fell into the fjord's waters (Fig. 1). For each location lagrangian drifters were continuously deploy for five minutes starting on April 21st at 14:00 hrs. Drifters were followed for 15 days, with hourly outputs.

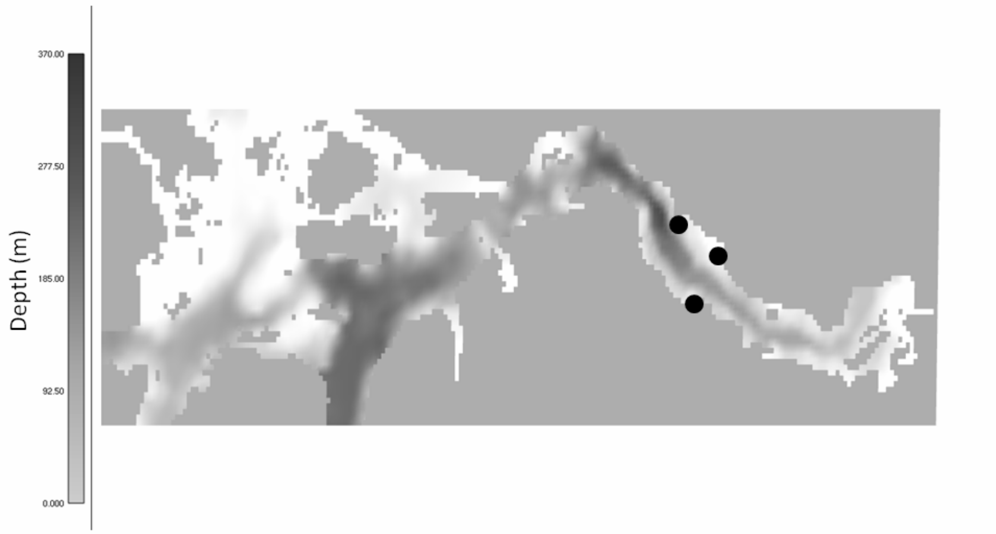

FIGURE 1: *Spatial extent and bathymetry of the AYSEN model. Please refer to the work by Tironi et al (this volume) for the geographic location and description of the study area. Black circles mark the approximate positions were victims fell into the water during the tsunami on April 21st, 2007.*

The average circulation of the fjord, for the standard no-wind run used in this experiment, is shown in Figure 2. The model effectively resolves a two layer estuarine circulation, with maximum surface velocities on the order of 0.2 $m\ s^{-1}$. This velocity is within the same order of magnitude of that reported by Cáceres et al. (2002).

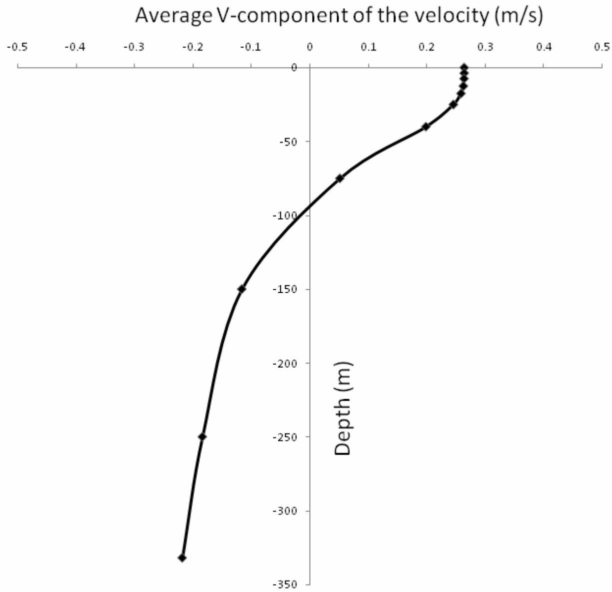

FIGURE 2: *Average along-fjord component of the velocity for the no-wind standard run used during the lagrangian experiments.*

3 LAGRANGIAN EXPERIMENTS

Considering the dynamics of bodies in the water (see 2. Above), we generated two experiments, both with the same initial conditions respect to the location of the lagrangian drifters: for the first experiments, objects stayed on the surface without sinking. This scenario would simulate human bodies that would get trapped on drifting objects such as tree trunks. For the second experiment, objects were allowed to sink at velocities close to that of a human body ($\approx$0.25 m/s). The results of both scenarios after 36 hours of simulation are shown on Figure 3. When objects stayed in the surface (Fig. 3 A) most of them had left the fjord after 36 hours or they anchored in the south shore. Conversely, if objects are allowed to sink, they would remain inside the fjord, in deep waters in the middle of the channel (Fig. 3B). In the last case, even if objects would re-surface again they would leave the system thorough its surface flow to disappear after 72 hours.

4 LANDSLIDE RISKS AND CIRCULATION MODELS

Contrary to what it is expect from tsunamis generated on open coastal oceans, landslide tsunamis occur so fast that there is hardly time for reaction. In the case of the Aysén tsunami

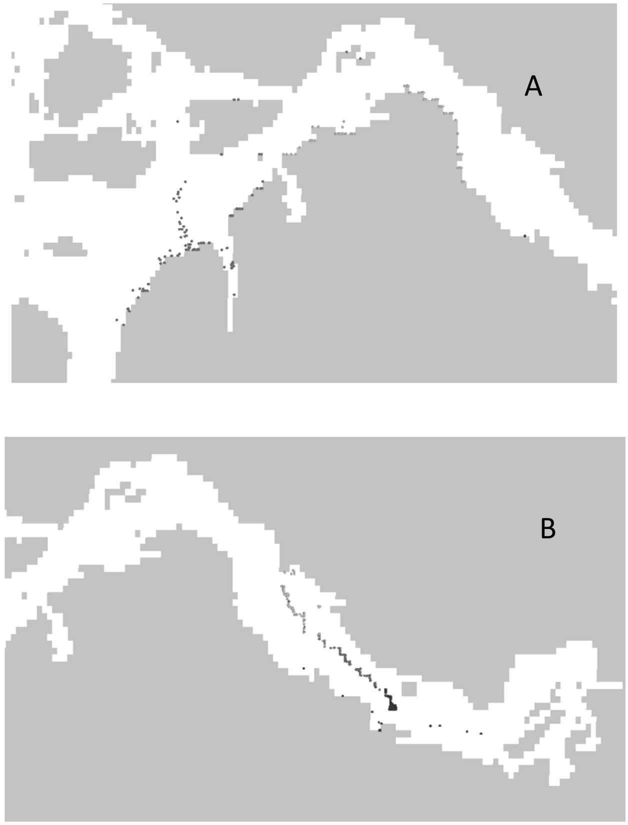

FIGURE 3: *Spatial distribution of lagrangian drifters for the first experiment, non-sinking drifters (A) and the second experiment (sinking drifters) after 36 hours of simulation.*

it took only 6 minutes for the waves to arrive at populated areas (SHOA 2007). Given the location of the landslide and the distribution of populated areas, victims were reduced only to those standing on boats during the event. Although the Chilean government deployed a rather costly operation in search of victims, none were found. Our lagrangian model results suggest that the main reason for this is that just 36 hours after falling I the fjord's waters, victims were already far away from the main event sector.

Risk prevention seems to be the only solution in the face of landslide tsunamis, given its fast dynamics. After the April event, the Chilean government organized a group of experts on numerical modelling, including ECOMANAGE scientists. The main task of this group was to generate a prediction model to assess risk potential and likely responses in other sectors of the Aysén fjord. The results of that work were delivered to the government on August 2007 (SHOA 2007). As a result of the studies conducted by ECOMANAGE and by other members of ONEMI's STC for Aysén, most parts of the inner fjord were declared as dangerous for aquaculture. However, although human lives and whole farming facilities were lost, salmon farmers soon started a lobby to reverse the decision of fjord closure for salmon farming.[1]

Thus, the main lesson from this experience is that numerical models, such as the ones developed by ECOMANAGE on the Aysén fjord, may be important tools for risk assessment and to suggest strategies for local governments. Specially when confronted to decisions that may affect human lives. However, in the end these tools will be only one component in a complex set of components related to the development and security of coastal areas. Final decisions will always be political!

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FINAL REMARKS

M. MATEUS, J.W. BARETTA AND R. NEVES

This volume has drawn together contributions from a range of disciplines to address the challenges and explore the opportunities for managing three very different coastal systems in South America: the Santos Estuary (Brazil), the Bahía Blanca (Argentina) and the Aysén Fjord (Chile). We have looked at a wide range of coastal management scenarios in the various geographical settings. But for all the differences, there are similar developments and a common recognition of the conflicting uses of the coastal area. This was the main theme at all sites, and hence, the principal focus of this volume.

A range of views emerges from this volume on managing coastal zones. Coastal managers generally agree that a broad and integrated view of coastal systems is necessary to capture their complexity. However this perspective sometimes is in conflict with a more reductionist approach to coastal sciences, which tends to concentrate on individual system components as such. While this approach can be very useful, if the component(s) studied are critical to system function, integration most of the time is the key to understand the link between system components and overall system function. Both approaches have been followed in the work developed during the ECOMANAGE project. Individual compartments of each system were studied and the knowledge gained was then used to gain a deeper understanding of the overall picture, where the complex feedbacks between man (socioeconomic features) and nature (ecological features) manifested themselves. The DPSIR framework has paved the way for this integrative approach.

A simple, yet fundamental lesson emerges from this book, connecting the local interest of development and economic growth with adequate management of the coastal resources. Social wellbeing means achieving the equilibrium between physical as well as socioeconomic stability, which makes the management task all the more difficult. Attaining such demanding goal implies minimizing the impacts of human occupation because they diminish or even destroy productive uses of coastal areas, undermining their ability to provide human populations with a healthy natural environment. As we have shown, the functioning of ecosystems may be disturbed by a wide range of impacts and these, in turn, have direct and indirect repercussions on the socioeconomic structure of the area. Through a series of responses and/or adaptive measures, which may range from changing social habits, laws and policies to technological solutions, humans attempt to prevent or lessen potentially destructive processes.

Despite the intrinsic complexity of these efforts, it is obvious that an assessment of the impacts of human activities on the system and the feedback mechanisms to the socioeconomy remains central to planning the use and development of coastal zones. And this is why the plans and politics for land use and for the management of the water is a shared responsibility between scientists, civil servants, politicians, stakeholders and NGO's. Coastal management is an integrating concept. The diverse group of people involved in this project has embraced the concept of a multidisciplinary approach, its value to society, and especially as a way of making their knowledge more relevant to coastal managers and stakeholders.

Hard decisions need to be made in both short- and long-term management of the three systems addressed in this volume. The complex interactions between the various subsystems in each case make such decisions particularly difficult. In face of this complexity, those involved in the management of these areas need to promote a continuing vision of the coast against which to plan both the immediate and the gradual development of the system. Certainly this will lead to difficult but fundamental choices and decisions in order to achieve the sustainable physical, social and economic development of the region.

This volume suggests that a critical lesson to be drawn for ICZM is that management decisions must be based on the understanding of the coastal system in a dynamic and holistic framework. It has been important to us to ensure that scientific approaches and models contribute to understand the integrated nature of coastal areas at multiple scales, and are available to support all levels of decision making for integrated management. This is the rationale behind the major effort devoted to the modelling of the three systems.

Fundamental understanding of the behavior of many components of each study site has been put forward in this volume by both physical and social scientists. However, integrating this disparate knowledge into understanding of the dynamics of each system and to apply it to achieve tangible outcomes for coastal management are still major challenges. Assuming that good decisions in coastal zone management emerge from detailed knowledge on the functioning of those areas, our goal has been encapsulating this knowledge in an integrative and comprehensive way. Achieving proper management policies is a moving target because the drivers change and, consequently, the pressures, state and impacts. As such, a continuous adaptation of management strategies is a necessity in order to optimize responses to these changes. While this also requires a continuous research effort and long-term study programs, we hope to have kick-started this on-going process with the work developed during the 3-year ECOMANAGE project.