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Technical Note / *Nota Editorial*

ANTHROPOGENIC INFLUENCES ON INTEGRATED COASTAL ZONE MANAGEMENT

INFLUÊNCIAS ANTROPOGÉNICAS NA GESTÃO INTEGRADA DA ZONA COSTEIRA

Francisco Taveira-Pinto[1], Paulo Rosa-Santos[1], Tiago Fazeres-Ferradosa[1]

One of the most challenging aspects of the integrated coastal zone management is the variety of interconnected topics that fall within the practical scope of a meaningful development of coastal regions. Such development is characterized by complex interactions between the inherent natural habitats, fauna and flora and their relationship with human occupied areas. These relationships include coastal settlements, fisheries centers, recreational areas, harbours and ports, among several other assets.

The Journal of Integrated Coastal Zone Management has a long history of published research on such fields. Some of the recently published works present useful information related to anthropogenic coastal interactions, environmental and coastal pollution and maritime works influencing integrated coastal management, e.g. Lisboa and Fernandes (2015), Almeida and Jardim (2019), Coutinho *et al*. (2019) and Taveira-Pinto *et al*. (2020a, 2020b). In this issue, the last one of 2020, four interesting and relevant works are compiled with a significant background on anthropogenic and engineering aspects of coastal management.

The work of Lima *et al*. (2020) provides an assessment of the soils and groundwater quality and the effects caused by anthropogenic influence. In this work the coastal plain of Paraíba do Sul river delta, in Brazil, is addressed as a case study, for which concentrations of trace metals are analysed in detail. The results revealed the presence of contaminants with natural geogenic origin. In addition, the concentrations of trace metals were compared to different locations around the world, being concluded that the values were significantly lower than in most of the cases reported in the literature.

The second research compiled in this issue is also related to pollution matters in coastal areas. Salgado *et al*. (2020) presents a chemical pollution study concerning the sediments present at the estuarine complex of Iguape-Cananéia, located in South-East of Brazil. The pollution metals analysed relate to the ancient mining activities at the region and ongoing anthropogenic activities. In this paper, the concentrations of cadmium, plumbum and zinc were analysed for 10 sampling locations spread across the estuary region, revealing a moderate contamination of the soils. This research highlights the importance of continuous monitoring of metals' concentration in coastal and estuarine soils.

Delgado and Riera (2020) provide a review of the anthropogenic disturbances and coastal conservation activities at the paradigmatic oceanic archipelago of the Canary Islands. This research provides additional insights on future scenarios concerning the threatened habitats and the taxonomic groups at the location, considering the intense human activities and main disturbances in the coastal ecosystems of the Archipelago. In addition, the results of this work outline the coastal regions which are more prone to be pressured due to anthropogenic activities currently existing. The analysis concludes that coastal protection actions are urgent to ensure that the archipelago develops in a sustainable path, which is crucial to avoid tropicalization, fisheries collapse and coastal degradation.

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Finally, Ezzeldin *et al*. (2020) address the interesting case of the Damietta harbour and its deep navigation channel, in Egypt, and its influence on nearby shorelines. This work stands as a practical case of anthropogenic influence caused by human infrastructure's development in coastal regions. The associated problems of sedimentation, which affect the North-East coast of Nilo's Delta are also presented. Remote monitoring techniques are used to assess the harbour effects on local sedimentation over the last 45 years. The shoreline past changes are analysed and predictions of future evolution are made until 2060. Significant retreat levels of the coastline are predicted, close to 280 m on average for 2060, highlighting the need for a stable solution to be implemented in the nearby Delta areas.

From pollution and contamination analysis to practical engineering case studies, this issue of the Journal of the Integrated Coastal Zone Management provides a broad set of papers, which are useful for any professional or researcher willing to dig into the wonders of applied research about anthropogenic influences on coastal zones.

Um dos aspetos mais desafiantes da gestão costeira integrada é a grande diversidade de temas interrelacionados com influência direta e relevante no desenvolvimento das regiões costeiras. Tal desenvolvimento é caracterizado por interações complexas entre os habitats naturais associados, a fauna, a flora e a sua relação com as áreas povoadas pelo Homem. Essas relações incluem cidades costeiras, vilas, centros pesqueiros, estruturas portuárias e recreativas, entre vários outros focos de interação intensa.

A Revista de Gestão Costeira Integrada tem uma longa história de trabalhos publicadas nessas áreas. De referir, por exemplo, que alguns dos artigos publicados mais recentemente abordam as interações antropogénicas com a zona costeiras, poluição ambiental e costeira e obras marítimas com influência direta na gestão costeira integrada (e.g., Lisboa e Fernandes, 2015; Almeida e Jardim, 2019; Coutinho et al., 2019; Taveira-Pinto et al., 2020a, 2020b).

Nesta edição, a última de 2020, são compilados quatro trabalhos interessantes e relevantes com uma forte componente relativa às influências antropogénicas e a aspetos de engenharia relacionados com a gestão costeira.

Lima et al. (2020) apresenta uma avaliação dos solos e da qualidade da água subterrânea, bem como os efeitos causados pela influência antropogénica. Neste trabalho, a planície costeira do Delta do rio Paraíba do Sul, no Brasil, é usada como caso de estudo, tendo as concentrações de metais-traço sido analisadas em pormenor. Os resultados revelaram a presença de contaminantes de origem geogénica natural. As concentrações locais de metais foram ainda comparadas com as existentes noutros locais do mundo, tendo-se concluído que os valores encontrados são significativamente menores do que na maioria dos casos graves relatados na literatura.

O segundo trabalho incluído na presente edição também está relacionado com questões de poluição em áreas costeiras. Salgado et al. (2020) analisam a poluição química presente nos sedimentos do complexo estuarino de Iguape-Cananéia, localizado no sudeste do Brasil. Os metais poluentes que foram analisados estão relacionados com as antigas atividades de mineração existentes na região e com as atividades antrópicas atualmente presentes. Neste trabalho, as concentrações de cádmio, chumbo e zinco foram analisadas em 10 locais de amostragem espalhados pelo estuário. Os resultados obtidos revelaram uma contaminação moderada dos solos. Este trabalho destaca ainda a importância da monitorização contínua da concentração de metais em solos costeiros e estuarinos.

Delgado e Riera (2020) apresentam uma revisão das perturbações antropogénicas e das atividades de conservação costeira no paradigmático arquipélago das Ilhas Canárias. Esta artigo fornece perspetivas adicionais sobre cenários futuros relativos aos habitats ameaçados e aos grupos taxonómicos no local, considerando as intensas atividades humanas e os principais distúrbios nos ecossistemas costeiros do Arquipélago. Para além disso, os resultados deste trabalho apontam quais as regiões costeiras mais pressionadas pelas atividades antropogénicas atualmente existentes. O estudo conclui que as ações de proteção costeira são urgentes para garantir que o desenvolvimento do arquipélago segue uma trajetória sustentável, crucial para evitar a tropicalização, o colapso da pesca e a degradação costeira.

Por fim, Ezzeldin et al. (2020) abordam o interessante caso do porto de Damietta e do seu canal de navegação profundo, no Egito, bem como a sua influência na zona costeira adjacente. Este trabalho é um bom exemplo prático da influência antrópica em regiões costeiras, causada pelo desenvolvimento de infraestruturas portuárias. Os autores apresentam este caso de estudo e *os problemas associados, designadamente no que concerne à sedimentação, que afetam a costa nordeste do Delta do Nilo. São utilizadas técnicas de monitorização remota para avaliar os efeitos do porto na sedimentação local, ao longo dos últimos 45 anos. As mudanças da linha da costa no passado são analisadas e as previsões da sua evolução futura são feitas até 2060. Este estudo prevê recuos significativos da linha da costa (cerca de 280 m, em média, em 2060), destacando também a necessidade de uma solução estável a ser implementada nas áreas do Delta mais próximas do Porto.*

Das análises de poluição e contaminação, aos estudos de casos práticos de engenharia, esta edição da Revista de Gestão Costeira Integrada fornece um amplo conjunto de documentos, que são úteis para qualquer profissional ou investigador disposto a mergulhar nas maravilhas da pesquisa aplicada sobre as influências antrópicas nas zonas costeiras.

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ASSESSMENT OF POLLUTANTS IN SOILS AND GROUNDWATER FROM THE COASTAL PLAIN OF PARAÍBA DO SUL RIVER DELTA, RJ, BRAZIL

L.S. Lima[1], U.C. Oliveira[2], C.M.A. Rangel[3], C.F. Barreto[1], V. M. C. Aguiar[1], J. A. Baptista Neto[1], E.M. Fonseca[1 @]

ABSTRACT:

In developing countries, the quality of the available freshwater is deteriorating mainly due to pollution. Underground water bodies are important resources, taking into account that they are usually more isolated and also protected from anthropogenic influence. Within this context, the present study aimed to assess trace metals, nitrate and organic contaminants in groundwater and soils from 10 bore wells located at the coastal plain of the Paraíba do Sul river delta. Groundwater of the study area revealed elevated concentrations of arsenic, iron, manganese and barium, surpassing the limits established by Brazilian legislation as well as the limits established by other regulatory agencies. As, Fe and Ba reached concentrations as high as 242.70, 31 919 and 4.041 µg.L^{-1} in groundwater. Despite the elevated values, results suggested a reducing environment and the contaminants appear to have a natural geogenic origin, which is corroborated by past studies in the same area. Soils of the aquifer presented low levels of trace metals, corroborating the hypothesis of geogenic contamination in groundwater. No significant differences among the bore wells were observed regarding trace metals in soils, and concentrations found in the present study were much lower than the ones found in impacted aquifers around the world. No PCB's were detected in groundwater or soils. Nitrate concentrations in groundwater were within the limits recommended by Brazilian legislation.

Keywords: Trace metals; PCB's; Groundwater, Soils, Paraíba do Sul, River Delta.

RESUMO:

Em países em desenvolvimento a qualidade da água doce está-se a deteriorar principalmente devido à poluição. Corpos de água subterrâneos são recursos importantes, se se condiderar que são mais isolados e protegidos da influência de atividades antropogênicas. Dentro deste contexto, o presente estudo teve por objetivo avaliar metais-traço, nitrato e contaminantes orgânicos em águas subterrâneas e solos em 10 poços localizados na planície costeira do delta do rio Paraíba do Sul. A água subterrânea da área de estudo revelou concentrações elevadas de arsênio, ferro, manganês e bário, ultrapassando os limites estabelecidos pela legislação brasileira assim como os limites estabelecidos por outras agências reguladoras. As, Fe e Ba atingiram concentrações máximas tão elevadas quanto 242.70; 31,919 e 4.041 µg.L^{-1} na água subterrânea. Apesar dos valores elevados os resultados sugerem um ambiente redutor e os contaminantes aparentam ter origem geogênica, o que é corroborado por estudos anteriores na mesma área. Os solos do aquífero apresentaram baixos valores de metais-traço, corroborando a hipótese de contaminação geogênica da água subterrânea. Não se observaram diferenças significativas para os teores de metais em solos entre os poços, e as concentrações encontradas no presente estudo foram muito menores que as encontradas em aquíferos impactados ao redor do mundo. Não foram detectados PCB's nas águas subterrâneas ou solos. As concentrações de nitrato nas águas subterrâneas mantiveram-se dentro dos limites recomendados pela legislação brasileira.

Palavras-chave: Metais-traço, PCBs; Aquíferos Subterrâneos, Solos; Paraíba do Sul, Delta Fluvial.

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1. INTRODUCTION

Of the total amount of water on Earth, freshwater accounts for only 3%. Among the freshwater resources, 30.1% is underground water, which makes it around 0.76% of the whole world's water volume. In the last decades, ground water reservoirs have been the target of substantial legislative, regulatory and scientific efforts (Samantara *et al.*, 2017; Vetrimurugan *et al.*, 2017; Burri *et al.*, 2019). Important data has been acquired about their geochemical characteristics to ensure their quality for human use since they represent an important water resource for mankind comprising different uses such as drinking, agriculture, industrial use, among others. Unfortunately, underground water may become a hazard if it becomes contaminated with toxic compounds (Amin *et al.*, 2011).

In areas dominated by agricultural activity, the contamination of water compartments, with nutrients, trace metals and organic compounds is a consequence of cultivation practices using large amounts of fertilizers and pesticides (Almasri and Kaluarachchi, 2007). Many studies reported the environmental increase of pollutant concentrations in groundwaters as a result of undue fertilization and pest control in farming sites (Larson *et al.*, 1997; Tang *et al.*, 2010; Wu *et al.*, 2015). Persistent pollutants, such as polychlorinated biphenyls (PCBs) and trace metals are commonly found in many parts of the world (Fu and Wu, 2006; Wu *et al.*, 2015), and represent a risk to human health, even at small levels (Brito *et al.*, 2005). PCB's use has been banned in many countries, however, these artificial chemicals still persist worldwide (Katsoyiannis, 2006). Neverthless, some of these chemicals are still used in tropical and subtropical countries and due to their extreme chemical stability PCB's levels will not be reduced substantially for many years (Rajendran *et al.*, 2005).

Trace metals are among the most concerning contaminants of drinking water, offering serious threats to human health, and being considered a major environmental concern. Toxicity of trace metals is associated with a continuous exposition to low levels of these elements, what can lead to concerning health issues (Momodu and Anyakora, 2010). The environmental balance of trace metals can be deregulated by anthropogenic activities (Smecka-Cymerman and Kempers, 2001). Urbanization of coastal areas, including industrial activities usually have the potential to contaminate its surroundings, a problem well reported all around the world (Fortunato *et al.*, 2012; Rasool *et al.*, 2016; Samantara *et al.*, 2017). The evaluation of contamination of underground water by trace metals has fundamental importance in order to prevent problems with potable water demand (Kumar *et al.*, 2017).

Contamination of underground waters by arsenic (As) is an environmental issue all around the world. Arsenic (As) is a metalloid, that dissolves in water, and its origins can be anthropogenic, through industrial discharges or mining operations, or natural through geological sources. Minerals such as Fe-Mn hydroxides can retain As in their structure, however, the reduction of these compounds by bacterial action can occur in anoxic environments, releasing the metalloid to the water. Arsenic bonded to sulfides in organic matter can also be released during the remineralization process (O'Day *et al.*, 2004). Arsenic is hardly detectable without analytical determination. This element cannot be tasted or smelled in potable water, however the ingestion of contaminated water with As can cause severe health issues such as cancer, skin problems, vascular diseases and damage to the nervous system, among many others (Chatterjee and Mukherjee, 1999; Roy, 2008; Chakraborti *et al.*, 2016). Arsenic is commonly detectable in shallow aquifers, around 30-70 m depth, rather than in deeper ones. Arsenic in underground water has been detected in 105 countries and it is estimated that over 200 million people have been exposed to As concentrations above the acceptable value established by World Health Organization (WHO) of $10 \mu g.L^{-1}$ (Chakraborti *et al.*, 2016). Several cases of contamination by arsenic through consumption underground water have been widely reported around the world (Chen *et al.* 1994; Hoppenhayn-Rich *et al.*, 1996; Borba, 2003; Yokota *et al.*, 2001; Erickson and Barnes, 2005, Rahman *et al.*, 2015).

The objective of the present study is to assess concentrations of trace metals, arsenic and PCB's in groundwater and soils from the aquifer locates at the coastal plain of Paraíba do Sul river delta, and evaluate the level of impact in the area. The study aims to establish a baseline for trace metals, arsenic, nitrate and organic contaminants on the study area.

2. MATERIALS AND METHODS

2.1 Study Area

The Paraíba do Sul drainage basin covers an area of approximately 57,000 km^2 including some of the most industrialized states in Brazil, such as Rio de Janeiro, São Paulo and Minas Gerais. In the state of Rio de Janeiro the basin covers around 20,600 km^2 (Azevedo *et al.*, 2018). This river basin is responsible for the water supply of the region of Paraíba Valley and also for approximately 75% of water consumption for the metropolitan area of the state of Rio de Janeiro (Paiva *et al.*, 2020). The river has a total extension of 1150 km and the area with watershed is highly urbanized. From the original vegetation of Atlantic Forest only 3% remains, whereas 70% of present vegetation is used as pasture for livestock and 27% for agriculture and reforestation (Ovalle *et al.*, 2013). The coastal plain of Paraíba do Sul is located in the lower river basin in the northern part of the state of Rio de Janeiro. The coastal plain itself has an area of 3000 km^2 with a very flat surface and 120 km length from north to south, an altitude of ~20 m and the lithology is composed of Tertiary and Quaternary terrains. The coastal plain is filled with lakes, ponds and swamps and has been extensively occupied by man, with construction of several artificial channels that reduced the volume of ground water and drained many pond and lakes (Ovalle *et al.*, 2013; Mirlean *et al.*, 2014). The city of Campos, located near the Paraíba do Sul river delta region, presents high water resource availability (Souza *et al.*, 2004). Caetano (2000) identified at this location, large underground water reserves, and one of them comprehends a reservoir of 11 billion m^3. According to Mirlean *et al.* (2014) the shallow groundwater in the delta region is found in a depth from one to several meters and is currently used by the local population. Environmental problems have already been identified as a result of the use and occupation of the area. The urbanization of the area with no sewage infrastructure, no treatment of industrial residues, the absence of domestic trash dumping sites and soil salinization problems were diagnosed (Marchioro *et al.*, 2011). This area is also home for Açu Harbor, operating since 2014 and one of the main poles in the sector of oil and gas operations in Brazil.

2.2 Sampling and analysis

Ten bore wells were randomly distributed in the study area (Figure 1). Sediments were sampled using a stainless steel tube corer and samples were stored in plastic bags properly identified and stored in ice until arrival the laboratory where they were frozen at -20ºC. The water samples for trace metals and arsenic determination were collected through pumping and immediately acidified with HNO_3. Water samples for determination of nitrate were collected and stored in 250 ml polyethylene bottles and kept in ice until arrival at the laboratory. Water samples were then filtered at the laboratory through cellulose acetate membranes of 0.45µm and frozen until the moment of analysis.

Filtered groundwater samples were analysed for determination of nitrate content through ionic chromatography. The determination of trace metals in groundwater were assessed using atomic absorption spectrophotometer (AAnalyst 800-Perkin Elmer®), after samples were filtered through 0.45 µm acetate membranes. In the laboratory sediment samples were freeze-dried and aliquots were separated for the determination of grain size analysis, nitrate, trace metals and arsenic. Grain size analyses were conducted using a Malvern 2600LC® laser analyzer after elimination the organic matter with hydrogen peroxide 10% (v/v). Determination of nitrate in soil samples was performed through ionic chromatography, after extraction of the nutrient from sediments with Milli-Q water and mechanical agitation for 24 hours, followed by centrifugation at 3500 rpm. PCB's were soxhlet extracted and then subjected to an alumina clean-up, silica fractionation and determination with GC–ECD and a capillary column (Smedes and Boer, 1997).

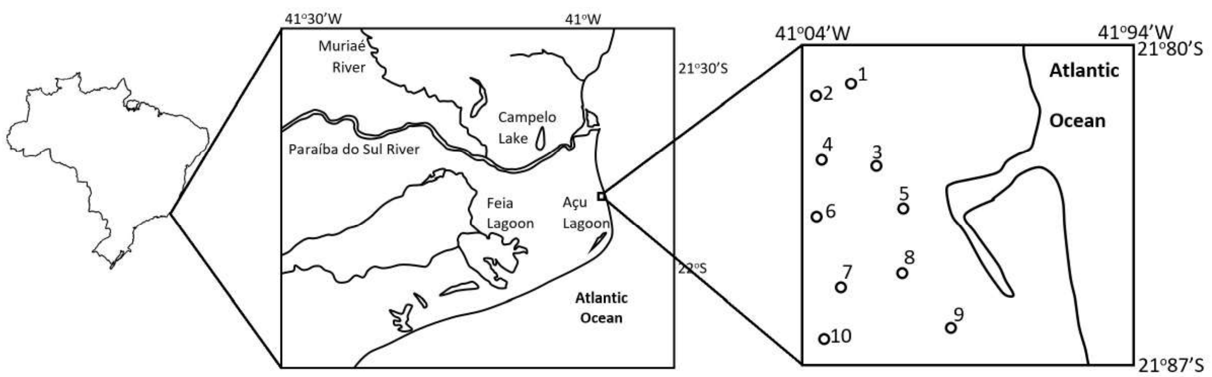

Figure 1. Sampling points for underground water and sediments at the coastal plain of Paraíba do Sul river delta.

The procedure for the determination of the trace metals (Al, Ba, Cd, Pb, Co, Cu, Cr, Fe, Mn, Ni, Ag, V, Zn) and As in the soils followed the method recommended by the EPA method 3050b (USEPA, 1996). Soil samples were grounded with mortar and pestle and 2 g of each sample was placed in digestion tubes with 5 ml of HNO_3 7 M and heated in a digestion block at 95ºC for 15 minutes and allowed to cool. Next to this, 2.5 ml of HNO_3 14 M was added to the digestion tubes and samples were heated again at 95oC for 30 minutes. The process of adding 2.5 ml of HNO_3 14 M was repeated, and after this, tubes caps were removed and samples were heated for another 2h at 95oC. After cooling, the tubes received 3 ml of H2O2 30% (v/v) and 1 ml of Milli-Q water and the mixture was heated with uncapped tubes at 95oC for 2h. After that, tubes were removed from the digestion block and capped and the mixture was left to rest for a period of 16 h. In the last step, the tubes received 2.5 ml of HCL 12 M and the samples were heated again at 95ºC for 30 minutes and allowed to cool. Samples were then filtrated through 0.45 µm acetate membranes and the filtrated was completed to 25 ml with Milli-Q water. Samples were determinate through flame atomic absorption spectrometry in a AAnalyst 800-Pekin Elmer®. Arsenic in ground water and sediments was determined through electrothermal atomic absorption spectrometry using a. Mercury analysis was performed according to USEPA method 7471B (USEPA, 2007).

The software Statistica 7.0® was used to perform Spearman analysis ($p<0.05$), to test significance of correlations among the variables, and Kruskal-Wallis test ($p<0.05$) in order to evaluate significant differences among the bore wells.

3. RESULTS AND DISCUSSION

3.1 Ground water

Among the established trace metals for determination in ground water only Ba, Fe, Mn and Zn were detected. Apart from trace metals, As levels were also found in ground waters from the study area (Figure 2). Table 1 shows current potable water quality guidelines around the world for trace metals and arsenic. In Brazil, the control of trace metals and arsenic in ground waters and sediments is established by CONAMA 420 (2009). The current results showed that the levels of trace elements were mostly far below the maximum allowed concentration, however, arsenic, barium, manganese and iron surpassed the levels established by CONAMA 420 (2009), as well as the levels proposed by other regulatory agencies around the world (Table 1).

Arsenic concentrations in ground waters were very concerning, varying from 1.11 to 242.7 $\mu g.L^{-1}$, surpassing CONAMA 420 (2009) guideline in all the bore wells, except 2 and 8 (Figure 2). Over the last decades, occurrence of high concentrations of arsenic in groundwater became a major environmental problem and started being recognized as a major public-health concern in several parts of the world (Shankar *et al.*, 2014). In Brazil, consternations arose about reports of human exposure to arsenic in drinking-water as an impact of gold-extraction in the region of Minas Gerais, located in southeastern Brazil (Mukherjee *et al.*, 2006). Costa *et al.* (2015) mapped the arsenic concentrations in water and stream sediments using data of 512 sampling points distributed over its 7,000km^2, located to the north from the present studied area. Although data have shown that arsenic occurs naturally in the Iron Quadrangle region, the authors concurred with the possibility that human action has increased these concentrations. In the same way, Borba *et al.* (2000), repeats the suggestion that in addition to the natural loadings of As from the rocks as a result of the weathering processes, the high concentrations of As currently registered in the soils of the Iron Quadrangle can also be faced as an influence of the residues discharged into the watershed during the 300 years of mining along the river margins. Still according to the same authors, since no arsenic minerals were identified in the environment, the arsenic concentrations registered in the river beds could be adsorbed on goethite, kaolinite and illite, minerals that are good “traps” of dissolved arsenic anions. On the other hand, Sakuma *et al.* (2010) suggested that human contamination already seems to be a problem in the area.

According to the same authors, the natural presence of arsenic in the southeastern region of Brazil, combined with the anthropogenic contamination resulted from mining activity may be the reason for the higher urinary arsenic levels among the local children. In 1998, urine exams were done for arsenic measuring in 126 school children. Results showed a mean level of 25.7µg/L. Further environmental evaluations in the surrounding areas found that the mean concentration of arsenic in surface water was 30.5µg/L; levels of arsenic in soil ranged from 200 to 860mg/kg, and sediments had a mean concentration of 350mg/Kg (Matschullat *et al.*, 2000). The study area of the present study, however is far from the problematic region of the Iron Quadrilateral and its upper geological section is mainly composed of quaternary alluvial sediments (Rocha *et al.*, 2013).

A study conducted by Mirlean *et al.* (2014) in the groundwater of the Paraíba do Sul river delta revealed that the aquifer is rich in arsenic fixed by authigenic sulfides. The referred authors have

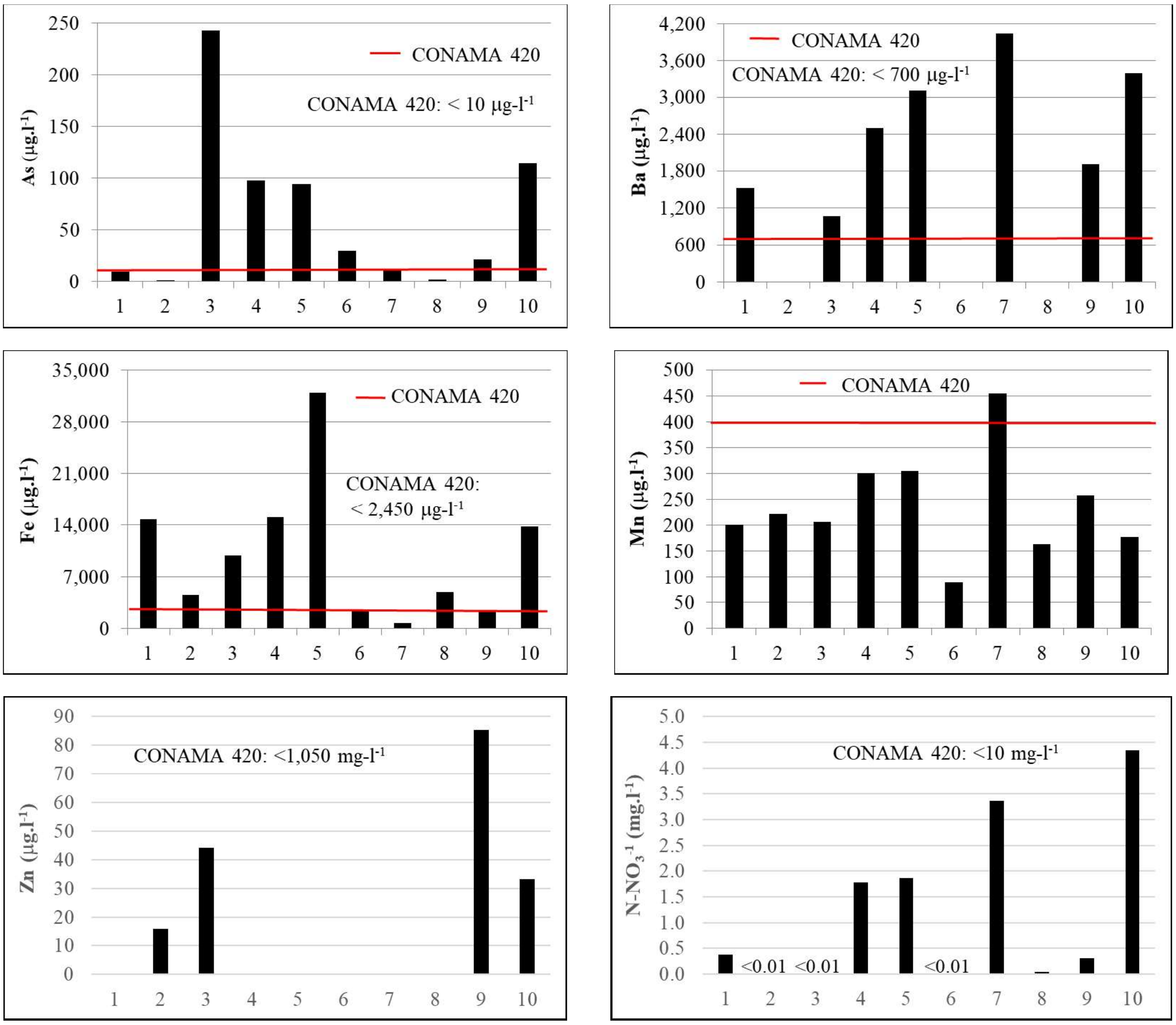

Figure 2. Results of arsenic (As), barium (Ba), iron (Fe) and manganese (Mn), zinc(Zn) and nitrate (N-NO3-) in ground water for the 10 bore wells in the coastal plain of Paraíba do Sul river delta.

described concentrations of As from 5 to 600 µg.L^{-1} in local tube wells and state that the arsenic in groundwater, corroborating the results of the present study, with values surpassing guidelines established by different health and environmental organizations all around the world (Table 1). Mirlean *et al.* (2014) conducted a very detailed study in the area of Paraíba do Sul river delta through a 20 m bore well core and also sampled the area of coastal inter-dune lakes/swamps 3.5km away from the coastline in the inner part of the beach ridge complex. The study of the bore well core included not only some trace metals, but also contents of organic carbon and palynomorphs. Combined results of the study made the refereed authors revealed that As in the ground water of the study area comes from the lower part of deltaic sediments, and that the sediment column has been formed during the coastal lagoon development through inter-dune swales to aquifer buried under eolian sands. The study of Martin *et al.* (1993) suggests that part of the geological profile of the study area may represent sediments of ancient freshwater shallow lakes/swamps or lagoons that existed during the first steps of the Paraíba do Sul delta formed within the 7060±260-5140±200BP period. The refereed authors demonstrated that this lake or swamp was later overlaid by sands, and that this process was long enough to form a 5-6 m layer enriched in organic matter. Aquifers associated with As are usually related to the input of organic matter, which can facilitate the release of this metalloid to ground waters (Herath *et al.*, 2016). According

Table 1. Current drinking water quality guidelines (µg L^{-1}) for trace metals from different regulatory agencies around the world and values found in the present study for ground waters.

Element	aWHO	bUSEPA	cECE	dFTP-CDW	ePCRWR	fADWG	gNOM-127	Present study
Sb	20	6	5	6	5	3	---	0
As	10	10	10	10	50	10	25	1.11-242.70
Cd	3	5	5	5	10	2	5	0
Ba	700	2000	1000	1000	1000	---	---	0-4,401
Cr	50	100	50	50	50	50	50	0
Cu	2000	1300	2000	1000	2000	2000	2000	0
Fe	---	300	200	300	---	300	300	749-31,919
Pb	10	15	10	10	50	10	10	0
Mn	100	50	50	50	500	500	150	89.70-454
Hg	6	2	1	1	1	1	1	0
Ni	70	---	20	---	20	20	---	0
Ag	---	100	---	---	---	100	---	0
Zn	---	500	---	5000	5000	3000	5000	0

a-World Health Organization (WHO 2011); b- United Stated Environmental Protection Agency (USEPA, 2011); c-European Commission Environment (ECE, 1998); d-Federal-Provincial-Territorial Committee on Drinking Water (CDW), Health Canada (FTP-CDW, 2010); e-Pakistan Council of Research in Water (PCRWR, 2008); f- Australian Drinking Water Guidelines (DDWG, 2011); g- Norma Oficial Mexicana NOM-127-SSA1-1994 (DOF, 1994).

to Mirlean *et al.* (2014), the As contents in the core was inherited by the metalloid earlier accumulated in organic sediments from an ancient inter-dune lake or lagoon. The release of As from microbial reduction of Fe (III) hydroxides and its subsequent migration to upper layers in the sediments to be fixed again in the oxic zone is well established (Campbell *et al.*, 2006), however, the diffusion downwards can also occur favoring the bonding of the metalloid with particulate sulfides, which is described to happen in the study of Mirlean *et al.* (2014) that proved the occurrence of sediments remarkably enriched in sulfide and organic carbon, especially at the bottom of the bore well core in the study area.

Concentrations of Ba in the bore wells surpassed the value established by CONAMA 420 (2009) up to 5 times, reaching the maximum concentration of 4,41.02 µg.L^{-1} in bore well 7 (Figure 2). Barium was not detected at bore wells 2, 6 and 8. Elevated concentrations of Ba in ground waters used as drinking water poses risks to human health since this element can cause hypokalemia, eletrocardiographic changes, increases blood pressure and affects the nervous system (Luu and Sthiannopkao, 2009; Bondu *et al.*, 2020). There are a number of sources that can account for Ba in groundwater. Aquifer solids play a similar role to estuarine particles, adsorbing Ba from groundwater which can be released in the event of exposure of the aquifer to salt water intrusion. Diagenetic controls of barium include the dissolution of Ba bonded to metal oxides phases or to authigenic barite ($BaSO_4$). Barite has low solubility and is very unstable in reducing environments due to the reduction of sulfate. High concentrations of sulfate would limit the concentration of Ba in ground waters due to saturation of barite (Giménez-Forcada and Vega-Alegre, 2015), and opposed to this, low concentrations of the anion (<5mg.l^{-1}) would increase Ba contents (Bondu *et al.*, 2020). The afore mentioned sources would be transients and maintained by the flow path of the low salinity desorption. The decomposition of organic material containing barium could release this element to ground water and this would suggest the existence of a large buried reservoir of organic matter enriched in Ba (Shaw *et al.*, 1998). The concentrations of Ba in the present study could be originated from sediment enriched with organic matter and controlled by sulfate reduction, since results found by Mirlean *et al.* (2014) confirm the existence of high concentrations of organic carbon and sulfides in the study area.

In the same way as the other metals, iron and manganese occur naturally in the global geology. In the underground reservoir, water accumulation shave contact with these solid materials dissolving and incorporating them, including Fe and Mn. Manganese has been known as a neurotoxin for at least 150 years, although it is still not clear whether eating or drinking foods and liquids with excessive levels of manganese can cause symptoms of manganism (ATSDR 2000). In the present study Mn varied between 89.7 and 454 µg.L^{-1}, surpassing CONAMA 420 (2009) guidelines only in the groundwater from bore well

Table 2. Concentrations of As, Ba, Fe, Mn, Zn (µg.L^{-1}) and nitrate (mg.l^{-1}) in different aquifers around the world.

Aquifer	Reference	As	Ba	Mn	Fe	Zn	NO_3-
*Espadan-Calderona Triassic Domain (Spain)	(Giménez-Forcada and Vega-Alegre, 2015)	2.31	90.5	4.25	92.2	--	46.4
Costal Aquifer Kalpakkam, Tamil Nadu (India)	(Samantara *et al.*, 2017)	--	--	n.d.-587.3	n.d.-89.9	1.8-220	n.d.-263.5
Calvery river basin, Tamil Nadu (India)	(Vetrimurugan *et al.*, 2017)	--	--	10-7000	50-550	--	--
Utar Pradesh (India)	(Kumar *et al.*, 2017)	0.07-237	--	0.9-301	40-12,700	4.8-1,500	--
*Pleistocene Aquifer Cambodia (Vietnam)	(Gillispie *et al.*, 2019)	146.9	--	1.8	2,900	--	143.2
Southern Quebec (Canada)	(Bondu *et al.*, 2020)	n.d.-280	n.d.-90,000	n.d.-7,500	n.d.-38,380	--	--
Aquifer System of Santa Catarina (Brazil)	(Carasek *et al.*, 2020)	--	--	n.d.-497	n.d.-2,685	n.d.-0.77	0.003-8.20

*mean values; n. d.- not detected

7 (Figure 2). Manganese concentrations in the groundwater, however, greatly overcame the limits established by other regulatory agencies around the world indicating threat to human health with regards to the use of groundwater by local population (Table 1). Iron varied between 749 to 31,919 µg.L^{-1}, and surpassed the brazilian guideline in all bore wells, except for 7 (Figure 2). Iron concentrations also surpassed by far the guidelines established by other regulatory agencies (Table 1). The dissolution of water in groundwater is usually in the form of Fe (II), since Fe (III) forms insoluble hydroxides in water. The release of Fe then could be originated by microbial reduction of Fe oxyhydroxides, an import process in the geochemistry of iron in anaerobic soils and sediments (Roden and Zachara, 1996).

A high concentration of Hg had been detected in only one sampling station bore well 2 (12 µg.L^{-1}). Zinc was also detected in the groundwater of some bore wells (Figure 2), but values were within the recommended limits by the brazilian legislation.

Overall, some of the results of groundwater from the present study are comparable to the ones found in aquifers impacted by anthropogenic activities (Table 3), as is the case of arsenic, which presented levels similar to the one found by (Samantara *et al.*, 2017; Gillispie *et al.*, 2019; Bondu *et al.*, 2020), however the previous studies in the study area suggest a natural source of the metalloid in the aquifer of the study area. The elevated levels of barium found in the present study, despite surpassing limits of current brazilian legislation, were much lower than the ones found by Bondu *et al.* (2020) in the aquifer of Quebec (Table 3).

The maximum acceptable concentration of nitrate in groundwater recommended by the brazilian legislation (CONAMA 420, 2009) is 10 mg.l^{-1}, and all the samples for underground water presented concentrations under this value (Figure 2), between 0.04 and 4.34 mg.l^{-1}. PBC's, were not detected in the groundwater of the ten bore wells in the present study.

3.2. Soils

Grain size influences the soil porosity, transport and deposition, and for this reason provides fundamental indications to the sediment depositional history and past environmental conditions (Pye and Blott, 2004). Results from showed predominance of gravel and sand at bottom sediments from the study sites, surpassing 80% of grain size for most samples. Fine sediments, composed of silt and clay varied between 7.8 and 31.9%, with highest values (>20%) at sites 1, 2 and 10 (Figure 2). Grain size analysis suggested the predominance of soils with a low retention capacity for trace metals and other pollutants, due to the low concentration of fine particles.

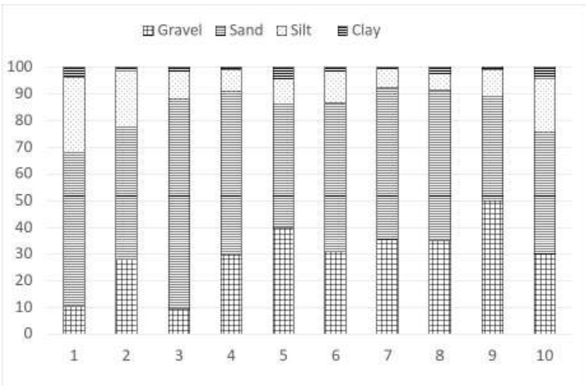

Figure 3. Grain size results for soils in bore well sites at the coastal plain of Paraíba do Sul river delta.

The trace metals environmental contamination can result from excessive fertilization, pesticide usage, irrigation, atmospheric deposition, and waste materials pollution (Aydinalp and Marinova, 2003). Unlike organic contaminants, trace metals do not experience microbial or chemical depuration (Kirpichtchikova, *et al.*, 2006), and their levels in soils are maintained for a long time after their input (Adriano, 2003). Studies assessing the use of sewage mud as fertilizer in sugarcane cultures suggested that the doses applied increased heavy metal concentration in the soil. The metals are also components of many pesticides, mainly Cu, Zn and Pb, which cause high soil contamination by these elements (Núñez *et al.*, 2006). The trace metal concentrations in soils obtained in the present study are presented in Table 3. Differences in metal content among the bore wells were not considered significant, except for copper (Table 4).

Table 3: Basic statistics for the trace metals concentrations detected in the bore wells from the study area. Mean, standard error, maximum and minimum values ($mg.kg^{-1}$).

Element	Mean±SE	Max	Min
Al	434.61±95.17	1615.94	84.49
As	1.25±0.47	7.03	n.d
Ba	261.81±51.85	774.73	61.81
Cu	0.99±0.24	3.13	n.d.
Cr	8.26±4.92	101.00	0.68
Fe	1554.98±429.99	5804.17	n.d
Mn	11.06±3.33	45.24	n.d
Hg	0.04±0.01	0.27	n.d
Mo	8.20±5.03	101.00	n.d
Zn	3.86±0.94	13.69	n.d

The registered concentrations in the present study can be considered low if compared with other studies of trace metals in aquifer soils. (Gillispie *et al.*, 2019) studied sediments from an aquifer located in a deltaic floodplain between Mekong and Bassac rivers in Cambodia (Vietnam) and found As, Mn and Fe with maximum concentrations of 24.9; 2,100 and 35,200 $mg.kg^{-1}$, respectively. (Kashouty and Sabbagh, 2011) found concentrations of metals much higher than the ones in the present study in sediments from an aquifer in Wadi El Natrum (Egypt), reaching values as high as 509; 1,849; 2,752; 243,000; 106 and 45,840 $mg.kg^{-1}$, for Ba, Cu, Cr, Fe, Mn and Zn, respectively. Mirlean *et al.* (2014) found a maximum concentration of As of 30.2 $mg.kg^{-1}$ in a bore well core in the coastal plain or the Paraíba do Sul river delta, a value much higher than the maximum found in the present study, 7.03 $mg.kg^{-1}$ (Table3). The maximum value of As found by Mirlean *et al.* (2014) was justified by the position of the sample in the core, around 15 m, coinciding with the end of the swampy period of profile development and beginning of the sediment overlap by eolian sands in the study area. These lower concentrations of metals compared to other study areas, could be explained by the sandy nature of the samples collected, with very low contents of fine sediments, which lowers the capacity of bonding metals. The Spearman analysis proved that there was no correlation between fine sediments and the metallic elements determined in this study (Table 5). The elements Fe, Mn, Zn, and Cr presented significant and direct correlation among them, suggesting a common origin (Table 5). The exception to low metal concentrations was Ba, that reached values over 700$mg.kg^{-1}$, surpassing values found in other aquifers by Kashouty and Sabbagh (2011) and Gillispie *et al.* (2019). Pérez *et al.* (1997), conducting a study covering 15 soil types distributed over 5 regions in Brazil registered concentrations of Ba varying between 0.09 and 201.4 $mg.kg^{-1}$, much lower than the concentrations obtained in the present study. Mercury values reached the highest concentration in bore well 8, 0.27 $mg.kg^{-1}$. No hg was detected in bore wells 1, 2 3 and 4. According to some authors, although not as expressive as the use of Hg pesticides, the usage of Hg in gold exploration activities in Rio Paraiba do Sul watershed between 1986 and 1987 has probably resulted in negative impacts like raising Hg sediments and biota levels (Primo, 2000). Souza el al. (2004) studied Hg levels in sediment cores distributed in some lagoons located in the same area of the present study. According to their study, in the Açú lagoon the levels ranged from 40.3 $\mu g.g^{-1}$ to 50.7 $\mu g.kg^{-1}$. PCB's were not detected in the soils of the present study.

Table 4: Kruskal-Wallis test for trace metal content in soils among the bore wells from the study area ($p<0.05$).

Element	*p*
Al	0.9596
As	0.0659
Ba	0.8677
Cu	0.0487
Cr	0.8722
Fe	0.9571
Mn	0.9325
Hg	0.2581
Mo	0.1029
Zn	0.0640

Table 5: Spearman correlation ($p<0.05$) for the soil variables in the study area (significant correlations in bold).

	Al	As	Ba	Cu	Cr	Fe	Mn	Hg	Mo	Zn	Gravel	Sand	Silt	Clay
Al	1.00	0.28	**0.71**	-0.14	**0.77**	**0.94**	**0.86**	-0.37	-0.18	**0.54**	0.14	0.01	-0.12	0.18
As		1.00	0.05	**-0.62**	0.34	0.36	0.37	**0.52**	**-0.49**	**0.70**	**0.49**	-0.39	-0.22	-0.14
Ba			1.00	0.23	**0.53**	**0.73**	**0.71**	**-0.56**	0.15	0.22	-0.13	0.22	-0.10	0.01
Cu				1.00	-0.07	-0.16	-0.14	**-0.67**	0.39	**-0.77**	**-0.72**	**0.49**	0.38	-0.14
Cr					1.00	**0.64**	**0.59**	-0.43	0.18	0.35	-0.08	-0.05	0.08	0.24
Fe						1.00	**0.92**	-0.26	-0.31	**0.56**	0.22	0.04	-0.25	0.04
Mn							1.00	-0.25	-0.31	**0.55**	0.25	0.06	-0.30	-0.03
Hg								1.00	**-0.50**	0.34	**0.65**	**-0.55**	-0.27	-0.08
Mo									1.00	**-0.48**	**-0.54**	0.15	0.32	0.00
Zn										1.00	**0.71**	-0.40	-0.38	0.13
Gravel											1.00	**-0.65**	**-0.56**	-0.15
Sand												1.00	-0.14	-0.17
Silt													1.00	0.30
Clay														1.00

4. CONCLUSIONS

The study of the ground water in the aquifer from the coastal plain of Paraíba do Sul river delta, detected levels of arsenic, barium, iron, manganese, zinc and nitrate. Results showed high concentrations of iron, barium and arsenic in the goundwater, which in most bore wells extrapolated the limits of the brazilian legislation, as well as the limits from some other regulatory agencies around the world. Arsenic levels in groundwater were comparable to the levels of the metalloid found in other aquifers impacted by anthropogenic activities. Despite that, there are no activities in the surroundings of the study area that could justify these concentrations. Indeed, the origin of arsenic in groundwaters of the Paraíba do Sul river delta was already proven to be from lake/swamp origin resulting from sulfide complex deposition through oxidation, which can occur during the pumping to collect groundwater. Barium and iron concentrations suggest an anoxic environment in groundwater, since only the reduced form of Fe is soluble, and large concentrations of Ba are usually controlled by reduction of sulfate. The reducing environment could also contribute to the concentrations of arsenic in groundwater through the reduction of As rich minerals such as Fe-Mn hydroxides. Concentrations of metals in soils did not corroborate the findings in the groundwater with regards to pollution, reinforcing the hypothesis that levels of As, Fe and Ba can be of natural origin, and no significant differences were observed among the bore wells. Results suggested a common and natural origin for iron, manganese, zinc and chromium in soils. The low concentrations of silt and clay on soils did not seem to play an important role in retention of the metals in the study area. Apart from small concentrations of mercury in soils, results suggest that most of the elements determined in the present study have geogenic origin. The present study presented very useful information to be used by public policies in the management of ground water resources that serve the local population. Geogenic contamination of ground waters used for human consumption is a common issue around the world, and mitigation measures must be taken by public authorities in order to prevent population illness, like prohibiting the installation of supply and consumption wells, in addition to the provision of alternative water sources. Parallelly, health and nutrition of the local community programs may be stimulated in order to render people more resilient and to lower the incidence or seriousness of the health impact.

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CONCENTRATIONS OF PB, CD AND ZN IN SEDIMENTS OF AN ESTUARINE COMPLEX AFFECTED BY ANCIENT MINING ACTIVITIES IN SOUTHEAST BRAZIL

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ABSTRACT: An artificial canal, built in 1852, diverted 60% of the flow of the Ribeira de Iguape River (RIR) into the northern region of the Estuarine-Lagoon Complex of Iguape-Cananéia (ELCIC), Southeast Brazil. Since then the river has become the main contributor of fresh water and suspended matter into the system. Additionally, the RIR was contaminated by mining activities, especially of Pb, which disposed their wastes directly into this river from their beginning until its cease in 1996. Thus, unknown quantities of metals began to be continuously introduced into the ELCIC. This study evaluated lead, cadmium and zinc concentrations in the surficial sediment at 10 points sampled along the ELCIC, 15 years after the closure of the mining. Zinc concentrations were in accordance with legislation and similar across the ELCIC regions, ranging from 10.4 to 22.8 $\mu g.g^{-1}$. Lead concentrations ranged from 19.3 to 67.9 $\mu g.g^{-1}$, with higher concentrations in the northern ELCIC region, especially in areas near to the canal and the mouth of the RIR. These values were above the Level I limits established by the Brazilian Environmental Agency, suggesting the possibility of a sediment toxicity. Cadmium was only detected (up to 3.6 $\mu g.g^{-1}$) in the northern region, exceeding Level I limits too. However, Pb and Cd values were not above the Level II limits. Results revealed areas of moderate lead and cadmium contamination in the northern ELCIC region, still indicating a strong contribution from the ancient mining activities and from other current anthropogenic activities such as agriculture or an incorrect disposal of sewage and residues. Therefore, a continuous assessment of metal concentrations in the sedimentary record is important to monitor the ELCIC environmental quality.

Keywords: contamination, estuary, metals, surficial sediments, Ribeira de Iguape River, Brazil.

RESUMO: Um canal artificial, construído em 1852, desviou 60% do fluxo do rio Ribeira de Iguape (RIR) para a região norte do Complexo Estuarino-Lagunar de Iguape-Cananéia (CELIC), sudeste do Brasil. Desde então, o rio tornou-se o principal contribuinte de água doce e material em suspensão no sistema. Adicionalmente, o RIR foi contaminado por minerações, especialmente de Pb, que descartavam os resíduos diretamente no rio desde o início das atividades até 1996. Assim, quantidades desconhecidas de metais começaram a ser continuamente introduzidas no CELIC. Este estudo avaliou as concentrações de chumbo, cádmio e zinco no sedimento superficial em 10 pontos amostrais ao longo do CELIC, 15 anos após o encerramento das minerações. As concentrações de zinco estavam de acordo com a legislação e foram similares em todo o CELIC, variando de 10,4 a 22,8 $\mu g.g^{-1}$. As concentrações de chumbo variaram de 19,3 a 67,9 $\mu g.g^{-1}$, com maiores concentrações no norte do CELIC, especialmente nas áreas próximas ao canal e a foz do RIR. Esses valores estiveram acima dos limites do Nível I estabelecidos pela Agência Ambiental Brasileira, sugerindo a possibilidade de toxicidade dos sedimentos. O cádmio (até 3,6 $\mu g.g^{-1}$) foi detectado apenas na região norte, excedendo também os limites do nível I. No entanto, os valores de Pb e Cd não estavam acima dos limites do nível II. Os resultados revelaram áreas de contaminação moderada por chumbo e cádmio no norte do CELIC, indicando ainda uma forte contribuição das antigas minerações e de outras atividades antrópicas atuais, como agricultura e disposição incorreta de esgoto e resíduos. Assim, a avaliação contínua das concentrações de metais é importante para se monitorar a qualidade ambiental do CELIC.

Palavras-chave: contaminação, estuário, metais, sedimento superficial, Rio Ribeira de Iguape, Brasil.

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1. INTRODUCTION

Metals are naturally occurring elements in the environment and many are essential for metabolism and life, but when in excess, they can become toxic (De Groot 2018). The presence of metals in water, soil or sediment is often associated with a geographical location (Fadigas *et al.*, 2006). Nevertheless, anthropic activities alter the distribution of these elements in the planet, with the industrial development being one of the main factors responsible for environmental contamination by metals (Duruibe *et al.*, 2007, Alvarez-Iglesias and Rubio 2012, Machado *et al.*, 2016, Lü *et al.*, 2018, Vardhan *et al.*, 2019).

The introduction of materials, including metals, into an aquatic system constitutes a disturbance that can start a complicated series of chemical and biological reactions (Hadzi *et al.*, 2018). Aquatic systems are very sensitive to pollutants as they have longer trophic chains that favor the distribution and accumulation of metals and other contaminants into the biota (Fernandez *et al.*, 2014, Yilmar *et al.*, 2017). Certain organisms accumulate metals, and bioaccumulation and/or biomagnification processes are observed. Therefore, metal contamination can be considered a direct threat due to habitat destruction, such as in the case of particle inputs from mining activities, and indirect due to the transfer of contaminants through the food chain (Machado *et al.*, 2016, Yilmar *et al.*, 2017, Cabrini *et al.*, 2018).

Contamination by metals in aquatic systems can be detected from the analysis of water, sediments and/or organisms (Ferreira *et al.*, 2010, Yilmar *et al.*, 2017, Cabrini *et al.*, 2018, Hadzi *et al.*, 2018). Generally, dissolved metals are first absorbed by organic or inorganic particles, and then incorporated into the sediment by particle settling (La Colla *et al.*, 2015, Machado *et al.*, 2016). Bays and estuaries act as geochemical barriers, intensifying the fixation and accumulation of metals (by processes such as flocculation) into sediments. It especially occurs when these regions are associated with mangrove areas. Such areas present sediments with abundance of organic matter and fine-grained particles, such as clay and silt (Semensatto-Jr *et al.*, 2007, Amorim *et al.*, 2008, Álvarez-Iglesias and Rubio 2012, Cruz *et al.*, 2019). However, several biotic processes (*e.g.* human interference – dredging; decomposition; some metabolic reactions) and abiotic processes (*e.g.* changes in environmental conditions – pH, temperature, salinity, redox potential, organic and inorganic complexing agents) can remobilize metals from sediments (La Colla *et al.*, 2015, De Groot 2018, Vardhan *et al.*, 2019).

Sediment analysis becomes an indispensable tool for assessing environmental quality of aquatic environments (De Groot 2018). Current values can be assessed by analyzing surface sediments, which are in direct contact with the water column (Semensatto-Jr *et al.*, 2007, Salgado *et al.*, 2018a, Azevedo and Salgado 2019). Ancient concentrations can be determined from deeper layers of the sedimentary record (Mahiques *et al.*, 2009, Cruz *et al.*, 2019). These assessments may also allow the identification of the main sources of pollution within a given aquatic system (Mahiques *et al.*, 2009, Salgado *et al.*, 2018a).

Metal pollutants cause great concern as coastal environments receive large amounts of discharges from human activities that contain these contaminants (Rubio *et al.*, 2010, La Colla *et al.*, 2015, Machado *et al.*, 2016, Yilmar *et al.*, 2017). Diverse Brazilian estuarine regions have registered metal contamination problems (*e.g.* Choueri *et al.*, 2009, Ferreira *et al.*, 2010, Garcia *et al.*, 2018, Cruz *et al.*, 2019, Vezzone *et al.*, 2019). Thus, the study of metal concentrations and their monitoring in coastal waters is necessary to provide tools for decision-makers so that they can promote protective actions and a sustainable use of marine resources, as well as the protection of the human health.

Many authors have studied contamination by metals in surface sediments from sheltered environments, such as basins, lagoons and estuaries (*e.g.* Semensatto-Jr *et al.*, 2007, Amorim *et al.*, 2008, Salgado *et al.*, 2018a, Cruz *et al.*, 2019). The Estuarine-Lagoon Complex of Iguape-Cananéia (ELCIC), Southeastern Brazil, is among the most environmentally relevant areas in the South Atlantic and is recognized by UNESCO as part of the Biosphere Reserve of the Atlantic Rainforest (Morais and Abessa 2014, Salgado *et al.*, 2018a). However, over the years, this region went through several changes that have affected its environmental quality. The most important was the opening of an artificial canal which increased the fresh water input, allowing the Ribeira de Iguape River (RIR) to enter the estuarine complex. Metal mining and refining affected this river for decades. These activities ceased in 1996. Among the consequences, the sediments of the RIR and of some points of the ELCIC contain significant levels of metals, such as Zn, Cr, Pb and Cu, which may be of concern (Vukan *et al.*, 2012, Abessa *et al.*, 2012, Mahiques *et al.*, 2009, 2013, Morais and Abessa 2014, Tramonte *et al.*, 2016, 2018, Salgado *et al.*, 2018a, Cruz *et al.*, 2019, Azevedo and Salgado 2019).

Therefore, the objective of this study was to investigate the concentrations and distribution of Pb, Cd and Zn in surficial sediments from the Estuarine-Lagoon Complex of Iguape-

Cananéia, 15 years after the closure of the mining activities in the region, as well as to compare them with the present Brazilian legislation, generating new data on the local environmental health. This research addresses not only metal inputs from the local watershed due to natural processes, but also possible inputs from different anthropogenic activities.

2. MATERIALS AND METHODS

2.1 Study Area

The Estuarine-Lagoon Complex of Iguape-Cananéia (Figure 1) has a total area of 2,500 km^2 and is located in the São Paulo State, Southeastern Brazil, between latitudes 24º 50' to 25º 10'S and longitudes 47º 25' to 48º 00'W. It's a complex system, between four large islands (Cardoso, Cananéia, Comprida and Iguape), with long narrow channels that extend approximately parallel to the coast. These are called Mar de (sea of) Cubatão, Mar de Cananéia and Mar Pequeno with average depths of 6, 10 and 6 m, respectively. The Comprida Island separates the estuarine system from the Atlantic Ocean, with the southern and northern limits of the Cananéia and Icapara mouths, respectively (Tessler and Souza 1998, Tramonte *et al.*, 2016, 2018).

The region is influenced by the Brazil Current, which carries Tropical Water, South Atlantic Central Water, Antarctic Intermediate Water and Upper Circumpolar Deep Water thought the southeastern coast of Brazil (Bilo *et al.*, 2014). The hydrodynamic pattern inside the estuarine complex is influenced by tidal currents and freshwater inputs wich causes periodic changes. These tidal currents determine the variation in salinity, oxygen content and other water conditions in the region, also influencing the plankton distribution.

In the south, the tidal stream splits into two branches after passing through the Cananéia mouth, one though the Pequeno channel and the other though the Cubatão channel. In the north, the tidal stream penetrates the mouth of the Icapara and flows through the Pequeno channel until the tidal currents

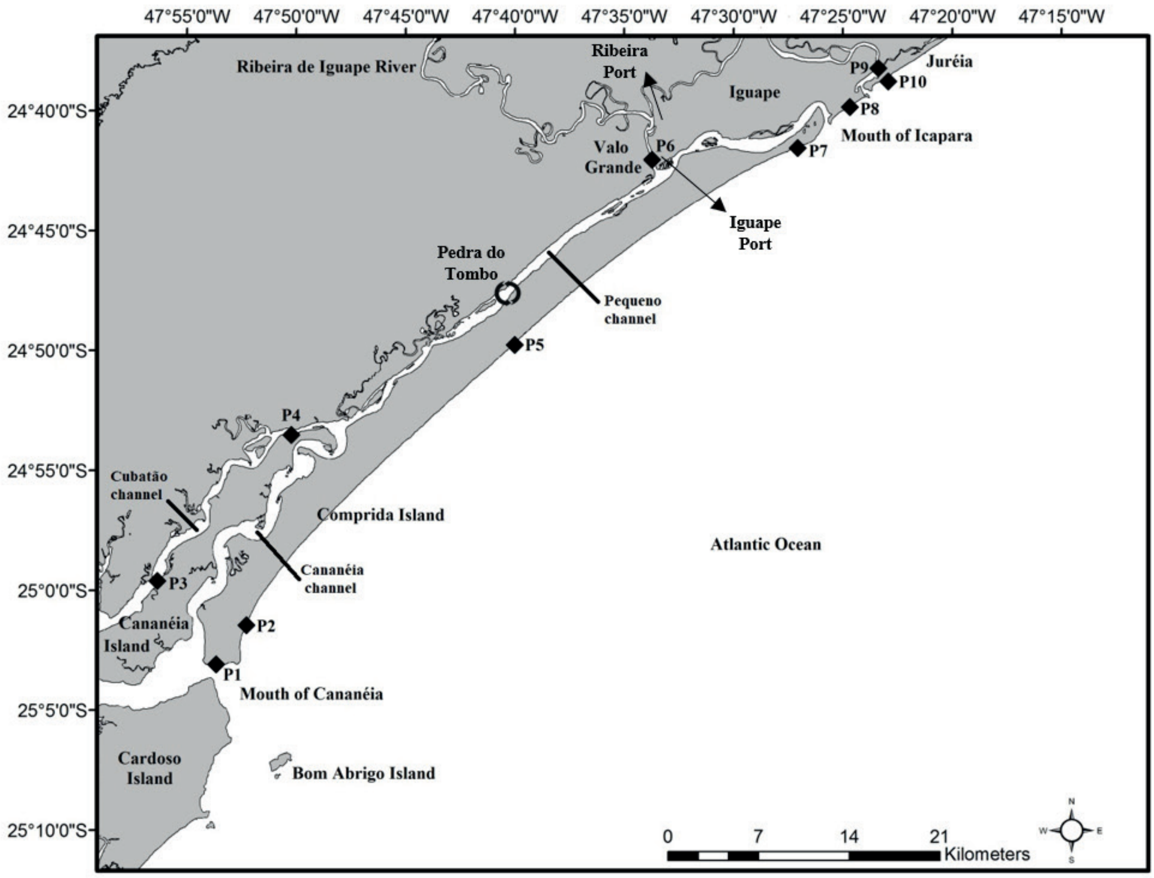

Figure 1. Map of the Estuarine-Lagoon Complex of Iguape-Cananéia, with emphasis on the sampling points of surficial sediments.

meet in the Pedra do tombo region (Figure 1), where the estuary waters are reversed. During ebb tide, the waters flow in the opposite direction (Tessler and Souza 1998, Salgado *et al.*, 2018a). An intense erosion occurs on the concave margins and sediment depositions takes place on the convex margins of the channels (Tessler and Mahiques 1998, Tramonte *et al.*, 2016). Sedimentation rates differ among locations and range from 5 to 10 mm yr^{-1} (Saito *et al.*, 2001a, 2001b, Mahiques *et al.*, 2013).

The climate of the region is divided in two well-defined seasons with dry winters and rainy summers (Cunha-Lignon *et al.*, 2009). The area is characterized by a low population density, a lack of large-scale economic activities and the presence of several terrestrial and aquatic protected areas, due to the large Atlantic Forest Reserve and extensive mangrove areas (Cunha-Lignon *et al.*, 2009, Morais and Abessa 2014, Tramonte *et al.*, 2016, 2018). However, the ELCIC suffered a major anthropogenic interference from mining activities and the opening of an artificial channel which caused many environmental changes in the region-, before the creation of environmental protection areas (Mahiques *et al.*, 2013, Morais and Abessa 2014, Cruz *et al.*, 2018).

The mouth of the Ribeira de Iguape River (RIR) is located near the city of Juréia (Atlantic Ocean). However, to shorten the path between the two old ports of the region (Iguape Port - located in the estuary and the Ribeira Port - located in the river) and maximize the transport and export of local agricultural products, an artificial canal called Valo Grande was built in 1852 (Figure 1). This construction caused, among many problems, the detour of the 60% of the RIR waters into the estuarine complex altering the salinity pattern (that decreased drastically), and the modification of the sedimentation patterns (generating new erosion and deposition areas), and the input of contaminants in the ELCIC (Saito *et al.*, 2001a, 2001b, Mahiques *et al.*, 2009, 2013; Tramonte *et al.*, 2018, Salgado *et al.*, 2018a).

Thus, nowadays the ELCIC presents quite distinct environmental characteristics between its southern and northern regions (Mishima *et al.*, 1985, Bonetti-Filho and Miranda 1997). Near the Cananéia Island (South) there is a strong marine influence. In this place the fluvial discharge (on average 50 $m^3.s^{-1}$) comes from the rivers Taquari, Mandira, das Minas, Itapitangui and several streams that drain an area of about 1,339 km^2 (Mishima *et al.*, 1985, Bonetti-Filho and Miranda 1997, Mahiques *et al.*, 2009). Near Iguape city (North) the characteristics are typically fluvial due to the freshwater inputs (on average 443 $m^3.s^{-1}$ - measured at Iguape) made by the RIR and its tributaries, which drain a basin of 23 350 km^2.

Nonetheless, the RIR basin has several metal deposits, which were exploited since the 17th century, most intensively between 1945 and 1995 (Moraes *et al.*, 2004, Mahiques *et al.*, 2009, 2013; Tramonte *et al.*, 2018). Lead, Zn, Au, Ag, As and Cu were extracted, and mines were operated for years discarding the refuses and slags from the smelting furnace indiscriminately into the river (Guimarães and Sígolo 2008, Mahiques *et al.*, 2013). It was estimated that during the mining period the river received about 5.5 tons per year of metal-rich wastes (Guimarães and Sígolo 2008).

This occasioned the contamination of the water and the sediments along the course of the RIR, especially by Pb (Moraes *et al.*, 2004, Cunha *et al.*, 2005, Guimarães and Sígolo 2008, Abessa *et al.*, 2012, Mahiques *et al.*, 2013). Lead contamination in the RIR and its effluents has been reported since the 70`s (Morgental *et al.*, 1975, Tessler *et al.*, 1987, Eysink *et al.*, 1988, Corsi and Landrim 2003, Moraes *et al.*, 2004, Cunha *et al.*, 2005, Cotta *et al.*, 2006, Alba *et al.*, 2008, Abessa *et al.*, 2012), including points were Pb concentrations exceeded up to 100 times the limit established by Prates and Anderson (1977) for non-contaminated sediments of 40 $\mu g.g^{-1}$ (Eysink *et al.*, 1988).

Mining activities ceased in 1996 due to a decrease in profitability and environmental problems (Tramonte *et al.*, 2016, 2018). After the closure of the mines, residues were deposited on the banks of the river in the form of piles of wastes that were exposed to the weather and consequently leached. Some of these piles remain in the area nowadays (Cruz *et al.*, 2019). Even after the 2000s, studies continue to show contamination points into the RIR area (Cotta *et al.*, 2006, Alba *et al.*, 2008, Abessa *et al.*, 2012, 2014).

Released metals can absorb to suspended matter and underwent mobility through the RIR and, as consequence, reach the ELCIC (Mahiques *et al.*, 2009, 2013, Tramonte *et al.*, 2016, 2018, Cruz *et al.*, 2019). Previous publications indicated that Pb, Cu and Zn were the main elements of concern in the region (Mahiques *et al.*, 2009, 2013, Tramonte *et al.*, 2018, Cruz *et al.*, 2019, Azevedo and Salgado 2019).

Mahiques *et al.*, (2009) showed that this estuarine environment was significantly polluted by lead, and that even after mine closure pollution continued, indicating contributions from the tailings piles still existing in the upper RIR. In addition, they indicated that Pb values in those samples recovered from the upper layers of the sedimentary column affected bymining activities were twice as high as those found in the contaminated sediments of the Santos estuary, one of the most industrialized areas off the Brazilian coast.

The analysis of sediment cores from the estuarine complex by Mahiques *et al.*, (2013) revealed an increase in Pb inputs between the 1930s and the 1990s. These authors also traced the anthropogenic influence at locations within 20 km of the Valo Grande Canal, and indicated that the suspended load caused by the artificial canal may not be the only factor controlling the distribution of metals from anthropogenic sources.

Nowadays the area is still under an increasing anthropogenic pressure and faces numerous environmental management problems. In addition to natural inputs and mining, the disposal of garbage and domestic effluents, the existence of agricultural activities in the nearby areas and the presence of vessels are among another possible metal sources to this ecosystem (Alba *et al.*, 2008, Amorim *et al.*, 2008, Morais and Abessa 2014, Tramonte *et al.*, 2016, 2018, Salgado *et al.*, 2018a, Cruz *et al.*, 2019).

2.2 Sampling procedures

Samplings were carried out on April 2011, the 9th and the 11th. Surficial sediments were sampled in 10 points under different anthropogenic pressures along the ELCIC (Figs. 1 and 2). Four of them were located in Comprida Island facing the open sea (P1, P2, P5 and P7), two of them in Cananéia Island facing the Cubatão Sea (P3 and P4), one in the Valo Grande Canal in Iguape (P6), one in the Juréia beach facing the open sea (P10), another at the mouth of the Ribeira de Iguape River in Juréia (P9), and the last one near the mouth of Icapara (P8). All sampling points corresponds to areas of briny-salty waters (salinity in the range 0.5-30 ‰) in the ELCIC, except that located at the Valo Grande canal (P6) which corresponds to fresh waters (salinity <0.5‰) according to salinity values determined in the ELCIC in previous studies (Bonetti-Filho and Miranda 1997, Maluf 2009) and considering the waters classification established in the Resolution No. 357 of March 15, 2005, of the Brazilian National Environment Council (CONAMA, which is the consultative and deliberative Agency that disposes about the National Environmental Policy).

The surficial sediments were collected manually with a plastic corer (15 cm length), at a distance of approximately 2 meters from the coast. Due the different sedimentation rates in the ELCIC (Saito *et al.*, 2001a, 2001b, Mahiques *et al.*, 2013), only the first 2 cm of sediment samples were considered for metal analysis. Samples were stored in plastic bags with no fixing reagent and conditioned in thermal containers at 4˚C during transportation. Afterwards they were kept in a freezer at -20˚C until analysis according to the Guide of sampling and preservation of samples of the Environmental Company of the State of São Paulo (CETESB 2011).

2.3 Determination of metal contents

Each sediment sample was homogenized by the quartering method and dried in an oven at 60°C for 12 hours. Then 50 g of the sediments were sieved at 32 tyler/mesh to remove the coarser fractions such as shells, animals and roots. The sieved fraction of each sediment sample (lower than 32 µm) corresponded to 93% to 97% of the bulk sample. These sieved fractions were divided into triplicates of 2g and then used for metals' determination according to the 3050B method proposed by the U.S. Environmental Protection Agency (U.S. EPA 1996). The procedure implies an acid digestion with HNO3, H2O2 and HCl. This pseudo-total digestion allows obtaining the environmental available metal fraction. All the reagents used were of analytical grade and the materials used were previously decontaminated by washing with neutral detergent and bathing in a nitric acid solution (10%) for 24h. Lead, Cd and Zn contents were determined by Flame Atomic Absorption Spectrometry (FAAS) in a Shimadzu AA6800 equipment, respecting the limits of quantification for each of the metals (Table 1). Standard solutions were prepared by successive dilutions using stock solutions of each metal of analytical grade (1.000mg L^{-1}), using 13% v/v HNO3. Analytical blanks were used and determinations were performed in triplicate.

The exactitude of the method was verified with CRM used to check the consistence of the internal standards in the laboratory. The RTC-CRM031-040 (sewage sludge) from Sigma-Aldrich was used with an analytical result of: 217.39 $µg.g^{-1}$ for Cd; 882.22 $µg.g^{-1}$ for Zn; and 126.48 $µg.g^{-1}$ for Pb; which were in accordance with their certified values of: 212 $µg.g^{-1}$ for Cd; 908 $µg.g^{-1}$ for Zn; 121 $µg.g^{-1}$ for Pb. The coefficient of variation (CV) was: of 1.78% for Cd; 2.03% for Zn and 3.13% for Pb. Metal concentrations were expressed in $µg.g^{-1}$ dry weigh.

2.4 Data treatment

The mean concentrations found in the sediment at each sampling point were compared to the limits determined by the CONAMA Resolution No. 454 of November 1, 2012, which establishes general guidelines and minimum procedures for assessing the material to be dredged from the Brazilian jurisdiction waters, and also gives other steps, once there is no

Figure 2. Sampling points of surficial sediments. P1: River at Ponta da Trincheira beach, extreme south of Comprida Island. P2: River on Boqueirão Sul beach, Comprida Island. P3: Cananéia-Continent ferry, Cananéia Island. P4: Cananéia-Continent Bridge, Cananéia Island. P5: River on Vilarégio beach, Comprida Island. P6: Valo Grande Canal, Iguape. P7: Beach at the extreme north of Comprida Island. P8: Mouth of Icapara. P9: Mouth of the Ribeira de Iguape River, Juréia. P10: Juréia beach.

Table 1. Intervals of linearity (IL) of the calibration curve, method linearity (r^2) and limits of quantification (LQ) for the analysed metals.

Metal	IL ($mg.L^{-1}$)	r^2	LQ ($mg.kg^{-1}$) Sediment
Pb	0.25 – 1.25	0.9984	12.5
Cd	0.01 – 0.4	0.9998	0.5
Zn	0.2 – 1.2	1.0000	10.0

specific federal legislation that deals with the maximum limits allowed for the concentration of metals in estuarine sediments. This resolution establishes two quality criteria, where values below the quality criterion I (Level I) indicate a non-expected toxicity of the sediments to the aquatic life. Values above the quality criterion II (Level II) indicate a probable toxicity of the sediments. Intermediate values (between Levels I and II) indicate a possible sediment toxicity. These values established by the CONAMA national legislation are present in Table 2. Values < LQ were excluded from groups of samples to generate the averages of the different regions.

Table 2. Limits of lead, cadmium and zinc established by the Resolution No. 454 of the CONAMA for surficial sediment (in $\mu g.g^{-1}$).

CONAMA	Pb		Cd		Zn	
	Level I	Level II	Level I	Level II	Level I	Level II
Salt water sediments	46.7	218	1.2	7.2	150	410
Fresh water sediments	35	91	0.6	3.5	123	315

For statistical analyses, sampling points were grouped according to their proximity: Southern region (P1, P2 and P3), central region (P4, P5 and P6) and northern region (P7, P8, P9 and P10). The STATISTICA 7.0 program was used. A single-factor Analysis of Variance (ANOVA) statistical test was applied to compare results between regions, followed by a Least Square Difference (LSD) test, considering 0.05 as the level of significance. To compare dissimilarities between regions, an Agglomerative Hierarchical Cluster Analysis (AHCA) method was applied using the Bray Curtis distance proximity type with the Ward agglomeration method. The product of this analysis was a dendrogram by using the statistical program XLSTAT.

3. RESULTS

Average concentrations of Pb, Cd and Zn for the studied samples are presented in Table 3.

Table 3. Mean (n=3) and standard deviation of the concentrations of lead, cadmium and zinc in the surficial sediments of the Estuarine-Lagoon Complex of Iguape-Cananéia. Those samples exceeding the Level I limits established by the Resolution No. 454 of the CONAMA are marked by an asterisk (in µg.g-1).

Points	Pb	Cd	Zn
P1	44.9 ± 10.0	<LQ	<LQ
P2	33.7 ± 3.5	<LQ	<LQ
P3	19.3 ± 4.1	<LQ	14.4 ± 1.0
P4	40.8 ± 6.1	<LQ	22.8 ± 6.3
P5	64.3* ± 10.0	<LQ	<LQ
P6	36.5* ± 5.6	<LQ	12.5 ± 2.2
P7	52.4* ± 15.2	<LQ	<LQ
P8	67.9* ± 10.9	3.6* ± 0.4	10.4 ± 1.3
P9	48.1* ± 4.1	2.6* ± 0.3	20.3 ± 1.4
P10	46.9* ± 5.3	2.4* ± 0.9	<LQ

< LQ - Values detected below the limits of quantification.
* Values above present Brazilian legislation.

3.1 Lead

Lead contents varied from 19.3 to 67.9 $\mu g.g^{-1}$ in the sampled points. The points P5, P6, P7, P8, P9 and P10 had concentrations above the Level I limit determined by the Brazilian legislation for fresh and salt water sediments (Table 3). When comparing values of lead in the three considered groups, an increase in concentrations to the north was evidenced: $F(2, 27)=6.9290$, $p=0.00373$ (Figure 3). The southern region (Mean=32.65 $\mu g.g^{-1}$ ± 12.54 $\mu g.g^{-1}$) had lower concentrations than the other two regions ($p<0.05$). However in the central (47.20 ± 14.50 $\mu g.g^{-1}$) and northern regions (53.85 ± 12.19 $\mu g.g^{-1}$) Pb contents were similar ($p>0.05$).

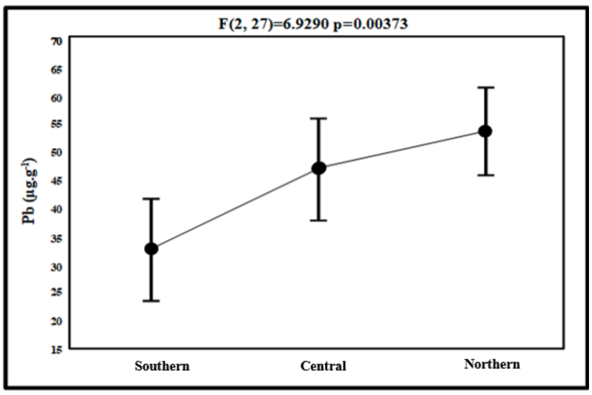

Figure 3. Mean + 0.95*. Reliability of intervals of lead concentrations in the southern, central and northern regions of the Estuarine-Lagoon Complex of Iguape-Cananéia.

3.2 Cadmium

Regarding the cadmium, concentrations were below its LQ in samples from the central and southern regions and in sample P7. However, values ranging from 2.4 to 3.6 µg.g^{-1} were observed at points P8, P9 and P10, located at the northern region. These values were higher than the Level I limit defined by the Brazilian legislation for salt water sediments (Table 3). Comparing the cadmium concentrations in the three regions, the northern region had higher cadmium concentrations than the other two regions of the estuarine system.

3.3 Zinc

Zinc concentrations of five of the analyzed points (P1, P2, P5, P7 and P10), all facing the open sea, did not exceed its LQ. However, Zn was detected in the other samples (P8 and those located in the inner part of the estuarine system-P3, P4, P6 and P9) ranging from 10.4 to 22.8 µg.g^{-1}. These concentrations were below the Level I limits for fresh and salt water established by the CONAMA legislation (Table 3). When comparing Zn contents in the three considered regions differences are not evidenced: $F(2, 27)=2,8183$, $p=0.07734$, $p>0.05$ (Figure 4). The central region (14.25 ± 11.98 µg.g^{-1}) had higher concentrations of Zn than the southern region (7.68 ± 5.18 µg.g^{-1}; $p<0.05$). However, it was similar to the northern region (11.98 ± 5.20 µg.g^{-1}; $p>0.05$). Nevertheless, the North and South regions did not differ in the concentrations of Zn in the surface sediments ($p>0.05$).

3.4 Comparison of the metal concentrations in the surficial sediments of the three regions

Summarizing, this study found lower concentrations of Pb, Cu and Zn in the surficial sediments from the southern region and higher concentrations in those from the northern region of the estuarine system, near the Iguape City. It was observed that Pb and Zn contents were more similar between the central and northern regions. According to the AHCA method, these two regions presented a dissimilarity of 0.089, while the southern region presented a dissimilarity of 0.285 from the other two regions (Figure 5).

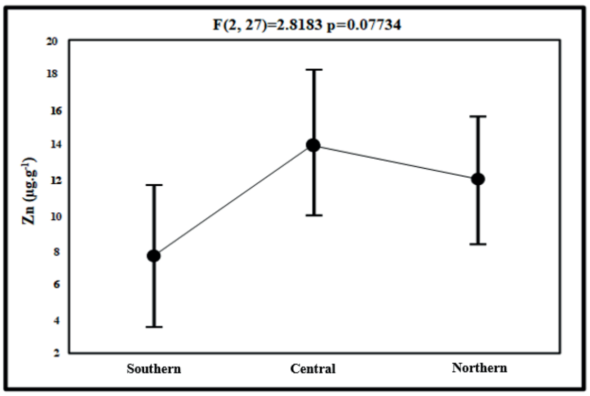

Figure 4. Mean + 0.95*. Reliability of intervals of zinc concentrations in the southern, central and northern regions of the Estuarine-Lagoon Complex of Iguape-Cananéia.

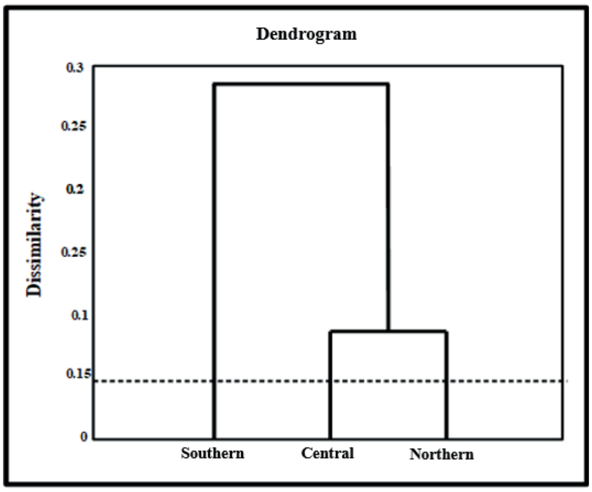

Figure 5. Dendrogram showing the dissimilarity in the Pb, Cd and Zn concentrations in surficial sediments from the three considered regions (southern, central and northern) of the Estuarine-Lagoon Complex of Iguape-Cananéia.

4. DISCUSSION

Lead concentrations in ELCIC surface sediments did not exceed the Level I limit established by the CONAMA legislation in four of ten analyzed points, in particular, in the southernmost points (P1, P2, P3 and P4). Then, sediments from the southern ELCIC region are not expected to cause toxicity. These results corroborated previous findings by Semensatto-Jr *et al.*, (2007), Mahiques *et al.*, (2009) and Salgado *et al.*, (2018 a, b) in the same region in sediments from near Cananéia Island.

However, Pb concentrations in the other six analyzed points exceeded the Level I limits of CONAMA legislation, indicating the possibility of a sediment toxicity to aquatic life (CONAMA, 2012). In particular, five of them (P5, P7, P8, P9, P10) presented values up to 45% higher than the limit value for briny-salty waters, while the sampling point located at the mouth of the Valo Grande Canal (P6), showed values exceeding in a 4% the fresh water sediment limit. All these points, excepting P5 are located in the northern region of the ELCIC, near the Valo Grande Canal (Iguape city) and the original Ribeira de Iguape River mouth (Juréia beach), thus indicating the contribution from this river to the Pb inputs in the area.

These results are in accordance with the previous literature of the area that indicates Pb contamination in the RIR basin happened since the 1970s (Morgental *et al.*, 1975, Tessler *et al.*, 1987, Eysink *et al.*, 1988, Corsi and Landrim 2003, Moraes *et al.*, 2004, Cunha *et al.*, 2005, Cotta *et al.*, 2006, Alba *et al.*, 2008, Abessa *et al.*, 2012, 2014). In addition to the natural contribution from the local river basin, all previous studies have highlighted metallic inputs to sediments caused by intensive mining activities, which thus constitute the main source of lead along the river area.

Although different authors have also observed a significant decrease in Pb concentrations in RIR waters and sediments following the mines closure in 1996, recent studies indicate that the local environment still remains affected by mineral waste inputs (Guimarães and Sígolo 2008, Rodrigues *et al.*, 2012, Abessa *et al.*, 2012, Tramonte *et al.*, 2018, Cruz *et al.*, 2019) including metal contributions from the tailings' piles still existing in the upper RIR (Mahiques *et al.*, 2009). These facts may explain the Pb contamination observed at those points near the mouth of the RIR (Iguape and Juréia city) in the present study 15 years after mines closure.

Nevertheless, a contribution of lead from another sources may also occur (Alvarez-Iglesias *et al.*, 2012). The RIR drains large areas of banana and tea cultivation along its course, and a contribution from the leaching of fertilized soils can be expected (Barcellos *et al.*, 2005). In addition, this river runs through the urban center of the city of Iguape and is influenced by various anthropogenic activities along its route, such as untreated effluents inputs.

Since the RIR waters reached the ELCIC through the Valo Grande Canal, the Pb contamination in the estuarine complex has also been evidenced in different studies over the years (Vukan *et al.*, 2012, Mahiques *et al.*, 2009, 2013, Morais and Abessa 2014, Tramonte *et al.*, 2016, 2018, Salgado *et al.*, 2018a, Cruz *et al.*, 2019, Azevedo and Salgado 2019). As observed in the RIR, the Pb values found in the present study in the northern ELCIC region (P5 to P10) were lower than those described years ago by other researches (Mahiques *et al.*, 2009, 2013, Tramonte *et al.*, 2016, 2018). This is probably because there is a natural tendency for these values to decrease over time after mining activities have ceased with the arrival of new and less contaminated sediments (Cunha *et al.*, 2005). However, the lead residence time in the soils is quite high, close to decades, or even higher than 100 years in the marine environment (Kabata-Pendias and Pendias 1985), explaining its persistence after over a decade.

Moreover, these high concentrations of lead in the estuarine area may still indicate the contribution of waste piles left on river banks that contaminate sediments entering the system through the Ribeira de Iguape River (Mahiques *et al.*, 2013) and the contribution of coastal cities and anthropogenic activities within the estuarine complex. These include the disposal of waste and effluents, agricultural and mariculture activities and the presence of boats. All ELCIC coastal cities have insufficient infrastructures for proper collection and treatment of their effluents and wastes, causing sewage to be discharged into streams and rivers without any prior treatment (Morais and Abessa 2014, Gusso-Choueri *et al.*, 2015, Salgado *et al.*, 2018a). Thus, the supply of Pb to the ELCIC may be influenced by these problems.

The anthropogenic influence on the concentrations of metals in the surficial sediments were observed by Mahiques *et al.*, (2013) at locations within 20 km of the Valo Grande Canal indicating that the suspended load made by the artificial canal was not be the only factor controlling the distribution of metals from anthropogenic sources in the ELCIC, corroborating the results found in the present research. Thereby, the high concentration of Pb observed at point P5, located in the central region of the system, can be explained by the anthropic presence in the area. This point is located at the beach of Vilarégio (Comprida Island), at the mouth of the river that crosses the neighborhood, and similarly to the other cities of the estuarine system, the villages of Comprida Island do not have sewage systems. Thus, this may be a possible source of lead contamination in this area.

As a consequence of the large quantities of Pb-enriched particles discharged to the RIR and the local atmosphere, which deposited afterwards on soils and sediments surfaces, Pb bioaccumulation was verified in RIR mollusks (Guimarães and Sígolo 2008, Rodrigues *et al.*, 2012), and in edible products (vegetables, eggs and milk) that grown in areas near the RIR (Cunha *et al.*, 2005, Lammoglia *et al.*, 2010). Furthermore, human lead contamination has been detected in some communities along the RIR, especially near the mining areas and near the ancient main refinery, although a non-alarming situation has been evidenced (Figueiredo *et al.*, 2004, Cunha *et al.*, 2005). However, the authors warn that these local populations live with environmental hindrances, are exposed to harmful substances and need to be assisted by local and state health and environmental authorities (Figueiredo *et al.*, 2004, Cunha *et al.*, 2005, Lammoglia *et al.*, 2010).

In the estuarine region, previous studies indicate the incorporation of Pb by aquatic animals, such as edible fishes (Duarte *et al.*, 2016, Gusso-Choueri *et al.*, 2015) and dolphins (Salgado *et al.*, 2018b). Lead values up to 14.64 $\mu g.g^{-1}$, were reported in muscle tissue from Cathorops spixii fish species from the ELCIC (Gusso-Choueri *et al.*, 2018) well above the 0.3 $\mu g.g^{-1}$ allowed by the Brazilian legislation (ANVISA, 1998). Then, an accentuated consumption of these fishes by the local population may pose risks to human health (Gusso-Choueri *et al.*, 2018). In Sotalia guianensis dolphins the lead mean concentration (3.17 $\mu g.g^{-1}$) were the highest described for this species, suggesting a trophic transfer, which may also affects the human population (Salgado *et al.*, 2018b). However, there are no studies involving metal contamination in inhabitants from the estuarine complex.

Regarding Pb toxicity in ELCIC, previous studies showed that this metal is among the major contaminants responsible for toxic effects on crustaceans (Duarte *et al.*, 2016), fishes (Gusso-Choueri *et al.*, 2015, 2018, Salgado *et al.*, 2018a) and possibly in aquatic mammals (Salgado *et al.*, 2018b). Such indications corroborate the findings of the present study, which indicate the possibility of sediment toxicity to aquatic life in the ELCIC (CONAMA, 2012).

Cadmium contamination may occur associated to lead mining (Cardoso and Chasin 2001). In the present study, most of the sampled points showed Cd contents below the LQ. Nevertheless, three points in the northern region (P8, P9 and P10), showed Cd values up to 3 times above the limits defined by the Brazilian legislation. These points were located at the original mouth of the Ribeira de Iguape river (Juréia), corroborating previous studies that found higher values of Cd in the northern region of the estuarine complex and values below the LQ in the southern ELCIC region (Maluf 2009, Salgado *et al.*, 2018a, 2019).In addition high concentrations of Cd in the waters of the estuarine complex were reported near the Iguape city (Maluf 2009) while low values of Cd were observed in the estuarine waters of the southern region (Souza *et al.*, 2012). Sediment analyses in one of the tributaries of the Ribeira de Iguape river held by Cotta *et al.*, (2006) found only one point with detectable Cd concentrations, and analyses on surficial sediments from the Comprida and Cananéia islands (Salgado *et al.*, 2018a, Azevedo and Salgado 2019) showed low Cd values too.

However, concentrations of Cd of 8,8 $\mu g.g^{-1}$ were found in the sediments from the Cardoso island by Semensatto-Jr *et al.*, (2007). These authors indicated that these concentrations were not related to an anthropic interference, since there is not known activity in the region that would release this metal. Thus, this high value was attributed to the predominance of mangroves in the area, which have a substantial quantity of clay, silt and organic matter that absorbs metals (Semensatto-Jr *et al.*, 2007). The Cd concentrations found by the present study were in the range of those presented in the bibliography, however the high concentrations found in the norther region can be attributed to anthropic interference (Semensatto-Jr *et al.*, 2007, Maluf 2009, Salgado *et al.*, 2018a, Azevedo and Salgado 2019).

Cadmium is a non-essential metal, highly toxic, considered dangerous to the health of organisms in general (Cardoso and Chasin 2001). However, studies on aquatic biota from Ribeira de Iguape River (Guimarães and Sígolo 2008) and the estuarine ELCIC region (Machado *et al.*, 2002) report that average Cd levels were below the limit value of 1.0 $\mu g.g^{-1}$ allowed for filter feeding organisms by the Brazilian National Health Surveillance Agency - ANVISA Ordinance 685/1998 and at low levels in the liver of the Guiana dolphins from the ELCIC (Salgado *et al.*, 2018b). In addition, Cd concentrations in blood samples from populations along the RIR did not arouse special concern (Figueiredo *et al.*, 2004).

Regarding Zinc, this metal is found associated to lead in the mineral deposits along the Ribeira de Iguape River basin and was also a target for mining activities (Cunha *et al.*, 2005, Cotta *et al.*, 2006). Thereby, high values of this metal have already been reported for sediments in the RIR basin (Corsi and Landrim 2003, Cunha *et al.*, 2005, Cotta *et al.*, 2006, Alba *et al.*, 2008). However, the present study found that Zn concentrations throughout the estuarine complex were up to 14 times lower than the limit set by CONAMA for briny salt water and 9 times lower than the limit set for freshwater. As previously described for Pb, there is a tendency for decreasing zinc concentrations in sediments from the ELCIC with time.

Previous studies have not revealed Zn contamination in the estuarine area (Semensatto-Jr *et al.*, 2007, Souza *et al.*, 2012, Salgado *et al* 2018, Azevedo and Salgado 2019). Semensatto-Jr *et al.*, (2007) indicated Zn concentrations ranging from 5 to 50.4 $\mu g.g^{-1}$ in sediments from the Cardoso Island, whereas Salgado (2009) obtained values averaging 11.01 $mg.kg^{-1}$ in sediment and 0.5 $mg.L^{-1}$ in surface water samples from the southern region of the estuarine complex; and Souza *et al.*, (2012) also found no high Zn (II) concentrations in the waters from the entire system.

However, zinc bioaccumulation has been reported in clams

(Corbicula fluminea) from the Ribeira de Iguape river (Guimarães and Sígolo 2008), in oysters (Crassostrea brasiliana) from the estuarine region of Cananéia Island (Machado *et al.*, 2002), and in the liver of estuarine fishes (Mugil curema) from the ELCIC (Fernandez *et al.*, 2014). In these three cases the Zn concentrations were above the 50.0 $\mu g.g^{-1}$ limit allowed in food by the National Health Surveillance Agency (ANVISA, 1998).

Regarding the possibility of metals causing adverse effects on local biota, Azevedo and Salgado (2019) analyzed the bioavailability of Fe, Zn, Mn, Co, Cu, Cr, Cd, Pb and Ni in the ELCIC sediments and observed that sediments presented toxicity at all points, being more toxic those from the northern region of the ELCIC due to the presence of the Valo Grande Canal. Thus, as observed in the present study, there is the potential of these sediments to cause adverse effects to aquatic biota. In addition, those authors highlight a possible metal contribution from other anthropogenic activities in ELCIC, such as urban and agricultural activities. As previously mentioned, incorporation of metals by the ELCIC aquatic biota occurs, including animals that are also intended for human consumption (Duarte *et al.*, 2016, Gusso-Choueri *et al.*, 2015).

In summary, in the present study zinc concentrations in the surficial sediments of the ELCIC were similar in the different system regions, while cadmium and lead concentrations showed a clear differentiation between the south (Cananéia Island) and north (Iguape) regions. In the southern region the concentrations of the three studied metals in the surficial sediment did not exceed the limits recommended by the Brazilian Environment Agency (CONAMA) for the preservation of aquatic life and for the sediment. Thus, the southern region can be characterized as slightly influenced by these metals. In the northern region, at those points mainly influenced by the Ribeira de Iguape River (Valo Grande Canal and the RIR mouth in Juréia), Pb and Cd concentrations exceeded the Level I limits established by the national legislation. This indicates a moderate contamination in the sediments of the area that have a potential toxicity to cause adverse effects to local biota.

In the aquatic environment, metals predominantly associate with suspended matter before depositing in the sediment where they are usually retained. However, under certain physicochemical conditions, such as the modification of pH, salinity, potential redox or low content of organic matter, solubilization of metals to the aqueous phase is possible (Álvarez-Iglesias and Rubio 2008, Souza *et al.*, 2012, La Colla *et al.*, 2015, De Groot 2018). Therefore, the residence time of metals in sediments depends on the changes of these variables in a given aquatic system and on the sediment composition (Cunha *et al.*, 2005).

According to previous studies, ELCIC sediments are predominantly sandy. Azevedo and Salgado (2019) observed 97% of sand, 2% of silt and 1% of clay in the grain size distribution of the analyzed points in the ELCIC. Tessler and Souza (1998) also observed a majority of sandy sediments (75% of the samples). This sediment distribution is influenced by local hydrodynamics: the ELCIC channels show low fine-grained particles contents due to the strong dynamics of the currents that occur near the bottom because of the proximity of the estuary mouths (Cananéia and Icapara), while those areas with lower hydrodynamics have higher silt and clay contents (Tessler and Souza 1998, Tessler and Mahiques 1998, Azevedo and Salgado, 2019). Due to the typical association of organic matter to these fine-grained particles (Fadigas *et al.*, 2006, Álvarez-Iglesias and Rubio 2008, Amorim *et al.*, 2008, De Groot 2018) higher levels of organic matter and nitrogen are detected in those low hydrodynamic areas of the ELCIC (Barcellos *et al.*, 2005, Azevedo and Salgado 2019).

For the studied region, many authors highlighted that climate, sedimentation processes and biogeochemical and hydrodynamic processes can cause differences in the concentrations of the elements at some points seasonally (Amorim *et al.*, 2008, Tramonte *et al.*, 2016, 2018, Azevedo and Salgado 2019, Cruz *et al.*, 2019). Thus, due to the set of variables that control this process, an extrapolation from the results obtained in the present study to another areas should be done with caution.

The obtained data corroborate that environmental conditions and anthropogenic impact levels are different between the southern (Cananéia) and northern (Iguape) regions of the Estuarine-Lagoon Complex of Iguape-Cananéia. These differences are related to typical estuarine characteristics in the Cananéia region and predominantly fluvial characteristics in the Iguape region, due to the impact of the Valo Grande Canal (Bonetti-Filho and Miranda 1997, Tramonte *et al.*, 2018). Moreover, in this last region, the anthropic presence is more evident and the waters register low salinity, high phosphate and silicate values (Maluf 2009), and as evidenced in this and previous studies, sediments are contaminated bylead and cadmium (Bonetti-Filho and Miranda 1997, Maluf 2009, Mahiques *et al.*, 2009, 2013, Morais and Abessa 2014, Tramonte *et al.*, 2016, 2018, Cruz *et al.*, 2019).

5. CONCLUSIONS

The present study evidenced that even fifteen years after the closure of the mining activities in the Ribeira de Iguape river, the lead (from 19.3 to 67.9 $\mu g.g^{-1}$) and cadmium (2.4 to 3.6 $\mu g.g^{-1}$) contents in those sediments from the Estuarine-Lagoon Complex of Iguape-Cananéia (ELCIC) sampled near the Iguape city and Juréia beach exceeded the limits of the Brazilian environmental legislation. This may be a matter of concern as sediment toxicity is expected. Although the measured values were lower than those found in previous studies, suggesting that restoration processes are underway, they still indicate the influence from the former mining activities and the environmental hindrances left behind, added to the persistent characteristics of metals in the environment. In addition, the contribution of another anthropogenic activities, such as agriculture and the contamination generated by sewage and wastes from local urban centers, may contribute to metal inputs into sediments. Thus, studies on metals' contents in the ELCIC continue to be of great importance. The authors encourage the continuous monitoring of the environment quality of the region and the evaluation of possible adverse effects in the local biota. This study generated new data on local environmental health, in order to contribute to the development of public policies for the preservation and/or improvement of the environmental, life and public health quality of the ELCIC region.

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ANTHROPOGENIC DISTURBANCES AND CONSERVATION OF COASTAL ENVIRONMENTS IN AN OCEANIC ARCHIPELAGO

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ABSTRACT: Oceanic islands are biotically fragile environments prone to suffer irreversible anthropogenic disturbances. The growth of the human population and the intensive occupation of the coastline are the cause of great ecological pressure on global insular coastal ecosystems. We review the current situation and future scenarios on a paradigmatic oceanic archipelago (Canary Islands, NE Atlantic Ocean), as a case study of the human footprint on marine coastal communities. The role of humans is pivotal, as we directly affect patterns of coastal occupation, pollution, invasive species or fishing. Here we synthesize the information that describes the current situation of the coastal ecosystems of the Canary Islands, indicating the main sources of environmental conflict and impacts. In addition, we review the state of the most relevant or threatened habitats and the taxonomic groups as actors of the main disturbances in the coastal ecosystems of the archipelago. We propose future general scenarios about expected changes, and foreseeable interactions that could occur to transform the coastal environments of the islands, in order to indicate areas susceptible to improvement for the conservation of these ecosystems. Integrative coastal actions are urgently needed for sustainable future scenarios to oppose deleterious trends such as tropicalization, fisheries collapse and extensive coastal degradation due to urbanization and infrastructure construction.

Keywords: Coastal development, overfishing, overpopulation, human-induced disturbance, introduced species, islands, oceanic archipelago.

RESUMO: As ilhas oceânicas são ambientes bioticamente frágeis e sujeitos a distúrbios antropogénicos irreversíveis. O crescimento da população humana e a ocupação intensiva do litoral são a causa de uma grande pressão ecológica sobre os ecossistemas costeiros insulares globais. Com o presente trabalho pretende-se rever a situação atual e os futuros cenários num arquipélago oceânico paradigmático (Ilhas Canárias, NE do Oceano Atlântico), como um caso de estudo da pegada humana nas comunidades costeiras marinhas. O papel dos humanos é fundamental, dado que afetam diretamente os padrões de ocupação costeira, a poluição, as espécies invasoras e/ou a pesca. Aqui sintetizamos a informação que descreve a situação atual dos ecossistemas costeiros das Ilhas Canárias, indicando as principais fontes de conflito ambiental e seus impactos. Adicionalmente, analisou-se o estado dos habitats mais relevantes ou ameaçados e os grupos taxonómicos como atores dos principais distúrbios nos ecossistemas costeiros do arquipélago. Propoem-se futuros cenários gerais sobre as mudanças esperadas e as interações previstas que podem . Alsotransformar os ambientes costeiros das ilhas, no intuito de indicar áreas suscetíveis de melhorias para a conservação desses ecossistemas. São necessárias ações costeiras integrativas urgentes para futuros cenários sustentáveis em oposicao a tendências deletérias, tais como tropicalização, colapso da pesca e extensa degradação costeira devido à urbanização e construção de infraestruturas.

Palavras-chave: Desenvolvimento costeiro, pesca excessiva, superpopulação, perturbação induzida pelo homem, espécies introduzidas, ilhas, arquipélago oceânico.

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1. INTRODUCTION

Oceanic islands are inherently fragmented environments that shelter relatively species-poor but endemism-rich biotas (Whitakker and Fernández-Palacios 2007). Species richness in oceanic islands, particularly in the tropics and subtropics, is very high in relation to its small surface area. Thus, many of them are threatened biodiversity hotspots and their contribution to global biodiversity is of paramount relevance (Myers *et al.*, 2000, Kueffer and Kinney 2017). Climatic and oceanographic variables, mainly currents and prevailing winds, along with distance to mainland (remoteness), location regarding frequented trade routes, which is a today's proxy for effective isolation (Helmus *et al.*, 2014, Delgado *et al.*, 2017), and island attributes such as island origin and type (Nunn *et al.*, 2016), topography, area and elevation (Brown and Lomolimo 1998; Whitakker and Fernández-Palacios 2007) are main determinants for the colonization of islands by species from the adjacent continents or other islands. The interaction of these features with geographical and anthropogenic factors also confers susceptibility for alien species invasions and human occupation (Russell *et al.*, 2017).

The major immediate environmental threat to island biodiversity is the presence of humans and their associated impacts (Kier *et al.*, 2009). Oceanic islands are usually considered more fragile than mainland areas due to combined effects of stochastic climatic events and anthropogenic disturbances that have driven extinction processes since the arrival of humans (Simberloff 1995; Graham *et al.*, 2017). The role of human activities and related shifts in disturbance regimes on island ecosystems are often studied from a short-term perspective (<50 years) (*e.g.* Helmus *et al.*, 2014). However, palaecological studies on islands worldwide have shown that human impacts are not new; in most cases, the effects of human settlement are noticeable even at early stages and for long periods (Connor *et al.*, 2012; Braje *et al.*, 2017).

Coastal deterioration on islands has experienced severe increases during the last five decades, through peaks in touristic, constructive and extractive industries (Brooks *et al.*, 2002; Arenas 2010; Halpern *et al.*, 2015; Semeoshenkova and Newton 2015). This has led to a necessity of new or enhanced Marine Protected Areas (MPA) as a mean of protecting habitats and species, and of recovering biotic resources such as fisheries (Roberts *et al.*, 2002; Edgar *et al.*, 2014). This growing concern has led to a reinforcement of legality to detect weak points in environmental protection. In this sense, progress has been made with an overall decrease in infractions associated with several types of impacts on the marine ecosystem (Figure 1). However, this does not guarantee the absence of multidimensional and diffuse coastal impacts that escape strict legal control.

Anthropogenic disturbances, especially on oceanic islands, include irreversible abiotic and biotic alterations, such as changes in community composition, species loss (and/or gain, as invasive taxa are concerned), and biodiversity disturbances of global resonance. Often intertwined and synergic, most of these disturbance factors are commonly treated with independence from one another, and concrete impacts in different areas are reported as scattered items both in the specialized literature and from the legal and management points of view.

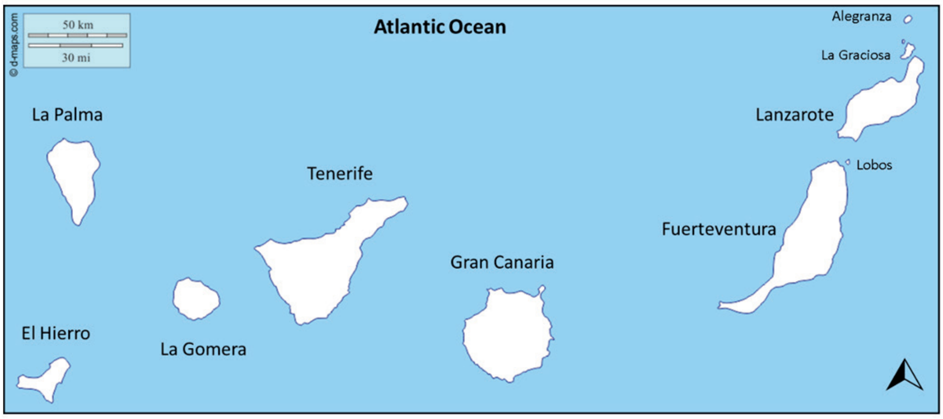

Figure 1. Map of the Canary Islands.

Here we review the status of conservation and anthropogenic disturbances of insular coastal ecosystems, taking an oceanic archipelago, the Canary Islands (E Atlantic), as reference case. An approach to this archipelago's concrete conservation issues seems opportune because it is a well-known oceanic territory located in a secularly disturbed region of great geostrategic, economic and ecological relevance. We aimed to characterize the human footprint on these islands, providing a synthetic frame of the main causes of ecosystem degradation at the archipelago scale. We concentrated in deterioration of the biotic values of coastal assemblages, *i.e.* those disturbances and deleterious forces affecting mostly habitats and species, rather than focusing in general eco-health issues of the marine realm.

We approached this multifaceted problem as follows:

1. We describe the current situation, reviewing the state of habitats and relevant taxonomic groups as descriptors of the coastal marine ecosystems.
2. We outline scenarios regarding expected changes, foreseeable interactions and synergies between different sources of disturbance, which could concur to transform the coastal island environments. This would aid to identify susceptible areas to introduce strategies leading to improved sustainability.

2. THE CURRENT SITUATION

The key threats to the coastal ecosystems in oceanic archipelagos are those globally affecting marine flora and fauna, including climate change, non-indigenous species, and human-induced disturbances, such as overfishing and coastal development (Riera *et al.*, 2014). There is an unprecedented rate and scale of development on oceanic islands. Such threats are accentuated in oceanic archipelagos due to massive growth of coastal settlements in the last decades, together with the presence of narrow island platforms where most of the coastal resources are confined.

2.1 Human-induced perturbations

A great variety of anthropogenic perturbations is currently occurring all over the planet, mostly due to the concentration of the population on coastal settlements and its subsequent pressure on the surrounding environments. Human activities present considerable threats to coastal ecosystems and resources in the following areas:

(i) structure and function of natural ecosystems;

(ii) shifts in natural resources, especially commercial fisheries;

(iii) extensive transformation of coastal natural landscapes.

2.1.1 Large scale and recreational overfishing

Overfishing is a major anthropogenic perturbation in coastal ecosystems. Coastal resources have been overexploited throughout the last decades in many oceanic islands (*e.g.* Falcón *et al.*, 1996; Tuya *et al.*, 2004) and currently, this situation is worsening because of the growing importance of shore recreational fishing, and other activities such as uncontrolled shellfishing or indiscriminate harvesting, as well as to a lesser extent illegal underwater fishing (Castro and Hernández-García 2012, Corral *et al.*, 2017). Canarian fishery catcheries have experienced a sharp decrease in the last four decades, from 2.2 kg trap^{-1} day^{-1} in the 1970's to 0.15-0.19 kg trap^{-1} day^{-1} (Castro and Hernández-García 2012; García-Mederos *et al.*, 2015). The development of an efficient management plan is a daunting task (*e.g.* Martínez-Saavedra 2011). Meanwhile, other factors need to be taken into account, *e.g.* oversizing of professional fleet and recreational fishing through the last decades (Zeller *et al.*, 2008; Sistiaga 2011). Other "biological" factors contribute to this situation, such as the high variety of commercial fish, approximately 100 species, and the necessity to update the minimum size of captures (García-Mederos *et al.*, 2015).

An important aspect is the increasing number of recreational anglers in the last decades. Current studies have shown that recreational fishing is responsible of >50% of the total catches landed (Jimenez-Alvarado 2010). This trend is more accentuated in islands with scarce presence of professional fishermen, where over 60% percentages of catches comes from recreational fishing. During the last decade, the impact of recreational fishing in reducing fish stocks and invertebrates has increased substantially (Martínez-Saavedra 2011).

2.1.2 Impacts from coastal development and infrastructures

2.1.2.1 Littoral development

The coastal perimeter of the Canary Islands is dominated by rocky cliff shores with interspersed beaches (mainly pebble and to a lesser degree sand beaches) (Table 1). This configuration creates limited space for coastal works, and hence lesser chance to correct biotic impacts from these constructions by selecting alternative suitable locations. The coastal development includes building for residential and tourist purposes; it can be associated with installation of land transport infrastructure for access to the

Table 1. Morphological characterization of the coastal perimeter of the Canary islands (length, in km). Data source: Instituto Canario de Estadística (ISTAC), Gobierno de Canarias.

Coast configuration	Canary archipelago	Lanzarote	Fuerteventura	Gran Canaria	Tenerife	La Gomera	La Palma	El Hierro
High cliffs	720.04	110.59	99.68	104.49	137.80	99.43	102.00	66.05
Low cliffs (up to 20 m height)	319.36	47.79	64.23	33.77	119.68	0.50	25.69	27.70
Low coast	170.22	2.20	82.18	17.38	47.96	1.50	11.90	7.10
Pebble beach	65.59	6.64	3.10	24.26	29.64	0.05	1.90	0.00
Pebble and sand beach	93.03	16.94	22.29	13.37	12.40	14.62	8.11	5.30
Fine and coarse sand beach	106.77	9.64	51.69	18.94	25.10	0.30	1.10	0.00
Artificial coastal works (ports)	78.88	19.46	2.74	24.43	25.60	1.25	5.05	0.35
Total length (km)	1553.89	213.26	325.91	236.64	398.18	117.65	155.75	106.50

sea (roads and promenades) and marine (pontoons, marinas, and larger ports with eminently touristic use but also with commercial or industrial purposes).

The economy of this archipelago, like that of other oceanic islands, is strongly dependent on sea transport (Tovar *et al.*, 2015). However, despite the increase in the extension and number of port infrastructures there has not been an increase in maritime traffic in the Canary Islands. Several of these large projects have indeed experienced abandonment without succeeding in stimulate local economies of that area. Some new port works have been approved with significant reductions of their original spatial and use scope, although they continue to cause significant alterations in the local coastal marine ecosystems.

Extensive projects have been developed, *i.e.* waterfront settlements, ports, marinas and piers, whereas in most cases lacking a sound environmental perspective, leading to severe impacts. For some projects, at least, mitigating measures have been taken, including: a) prohibition of anchoring vessels in the area of coastal influence near the port to avoid disturbances to the remaining *Cymodocea* beds; b) permanent sand transfer from N to S to reduce the impact of sediment loss south of port infrastructure, among others. Compensatory measures have also been addressed, mainly aimed at qualifying protected areas and restoring habitat (*i.e.* planting *Cymodocea* seagrass to rehabilitate seagrass meadows, and habitat restoration of nearby terrestrial areas).

Coastal development has significantly transformed shallow and productive coastal ecosystems into land for recreational and industrial purposes, introducing a variety of anthropogenic stressors to the coastal environment in many oceanic archipelagos. The steadily growing trend human population along and near the coast has been accompanied by new infrastructures occupying the coastal area, including the intertidal and supratidal environments (Bulleri and Chapman, 2010). Moreover, extensive coastal works have had both direct and indirect environmental effects on the ecosystems (Relini *et al.*, 2002; Sheehy and Vik, 2010). For example, dredge and fill procedures for construction of ports and marinas not only cause short-term problems of sedimentation and turbidity, but also postponed impacts due to re-suspension of fine sediments (Erftemeijer *et al.*, 2012).

Coastal Canarian ecosystems are heavily impacted by overpopulation (with an average of 254 ind./km^2, and ca 500 ind./km^2 on the islands of Tenerife and Gran Canaria; see Riera and Delgado 2019), an upward trend that still does not stop. To the pressure of the resident population must be added the “floating” or tourist population, which exceeded 6000 tourists per kilometer of coastline between 2005 and 2013 (MAGRAMA 2014; Riera and Delgado 2019). Almost 10% of the Canary Islands’ coastal perimeter is heavily transformed by the construction of breakwaters, groins, dykes and other rigid linear structures, as well as artificial beaches (*ca.* 80 km of diffrent structures along the shoreline), especially on the coasts of the southern slope of the capital islands Tenerife and Gran Canaria the use of artificial beaches (Riera *et al.*, 2014).

As remarked by Nunn *et al.*, (1999), smaller oceanic islands (the vast majority of them) are differentially vulnerable to coastal human impacts due to their greater amount of coastline compared to island area. It has also been argued that the distinctive nature of small islands (driven mostly by size and isolation) could determine levels of anthropogenic development in their ecologically limited environments (Fernandes and Pinho 2017, Delgado *et al.*, 2017). In this sense, an influential feature conferring some passive protection from development would be the ruggedness and relief, which would be a chief constraint to urban sprawl on smaller islands with smaller availability of buildable areas (Nunn *et al.*, 1999).

The fragmentation and individualization of coastal infrastructural projects may derive in a lack of anticipation of cumulative effects, of especial concern for small and isolated islands, exemplified by some archipelagos in the Indo-Pacific (Bass and Dalal-Clayton 1995). Solutions to such conflicts for small islands include multifunctional artificial reefs, as in Azores in the Atlantic (Ng *et al.*, 2013), and consideration of intrinsic protective qualities integrated with sound management of native coastal ecosystems (Gracia *et al.*, 2017).

2.1.2.2 Pipelines, desalination and aquaculture

Some activities on the coast cause a localized impact that, when repeated and accumulated along the coastal perimeter of the islands, causes synergistic and cumulative effects on a wider scale. Here we include buffer zones of pipelines, desalination plants, and aquaculture offshore cages. The environmental affection of these perturbations is limited to the adjacent area (10s to 100s meters). The addition of POM (particulate organic matter) and brine from sewage and desalination plant effluents, respectively, only affect the communities located on the proximity of the outfall (Riera *et al.*, 2011b, 2012). This pattern is also observed on the surroundings of power station outfalls (Riera *et al.*, 2011a), because of the presence of coastal currents through the whole water column and the rapid temperature dissipation as a consequence of temperature differences between the outfall and the receiving coastal water mass.

Offshore cages also originate a “footprint” of environmental perturbations limited to the aquaculture lease (Riera *et al.*, 2013). However, their development has so far limited to certain archipelagos such as Canary Islands, Madeira or Hawaii, with suitable conditions for culturing temperate-water species (*e.g.* Kam *et al.*, 2003). The most noticeable effects of this activity are concomitant factors, *i.e.* fish aggregates that are attracted by the excess organic input from uneaten pellets and fish droppings (Boyra *et al.*, 2004). One of the main issues about fish aggregates is their concentration in a limited area, on the surroundings of cages that make them vulnerable to fishermen. Fish aggregates are characterized to be diverse and formed by several species that dominate overwhelmingly these assemblages (Riera *et al.*, 2014). The increase of taxonomic and ecologic diversity of fish aggregates also occurs in functional terms (Riera *et al.*, 2017), with a functionally more diverse fish community compared to sites far from cages.

2.2 Coastal ecological imbalance

The high degree of restriction in biotic resources, generating increased vulnerability to human exploitation, and in part as a result of this, of coupling between the marine and terrestrial biotic environments in small oceanic islands conditions the nutrient fluxes between both compartments and thus indirectly the biodiversity present in both environments (Hunt 2007). Furthermore, immigration rates to the more isolated islands like Fiji, Samoa, Easter Island and Galápagos are commonly lower than for most continental grounds or even continental platform islands, and hence losses of biotic elements from islands are more difficult to replace (Whittaker and Fernández-Palacios 2007).

In addition, smaller oceanic islands have very limited capacity to compensate for both natural and anthropogenic impacts, as well as constrained capacity to regulate its local climate. Hence, for smaller islands, mesoclimate would be strongly governed by regional-scale climate, and ecosystems would suffer from overspecialization in ecologically impacting activities such as tourism (see Wong *et al.*, 2005, for the so-called Small Island Developing States). Coastal ecological imbalances may thus derive in a great deal from an interaction between these intrinsic vulnerabilities of oceanic islands.

Current imbalance of coastal ecosystems is perceived in several ways in oceanic archipelagos like the Canary Islands. There are consistent differences among islands within the same archipelago in human population size and density, and hence in the number and intensity of human-induced environmental disturbances. Overpopulated islands are currently suffering coastal ecological imbalance because of degradation of environmental processes, both direct and indirectly linked to human activities. Most human direct and indirect impacts occur within the first 50 m depth, where about half of the described species can be found (Martín 2010). These processes may be summarized in two categories: (i) outbreak of opportunistic and fast-growing species; and (ii) meteorological stochastic events.

2.2.1 Outbreaks or die-offs of opportunistic and fast-growing species

The proliferation of the sea urchin *D. africanum* has been responsible for an impoverishment of coastal rocky substrates in the Canarian archipelago, with the exception of El Hierro

Table 2. Casuistic scheme of coastal management and conservation problems on oceanic islands worldwide. The list does not pretend to be exhaustive, but to review the most prevalent causes of environmental degradation in insular coastal ecosystems, with proposals for improvement from the local to the global scale.

Example problems	Solutions/proposals of management, protection and mitigation	Site/Reference
• Coastal degradation (seagrass beds), • Resort construction at the shoreline • Harbours and marinas • Pollution control • Tourism impact • Beach nourishment	• Urgent review of EIA licensing and resolutions for coastal developments • Revising strategy for marine protection and management • Aquatic tourism impact control	Mauritius, Indian Ocean/Daby (2003) Madeira (Abrantes, 2017) Canary Islands (Riera *et al.* 2014)
• Local fisheries affected by climate change • Illegal fishing (including spearfishing and angling) • Interference of foreign fleets and loss of insular fishing sovereignty	• Consider needs and claims of insular artisanal fishermen communities (*i.e.* "include all stakeholders") • International protection of local traditional fisheries • Modelling approaches for temporal and spatial predictions • Creation of fishing reserves (Marine Protected Areas) and marine sanctuaries (no-take)	Indonesia (Sumatra, Sulawesi –Spermonde Islands)/Glaeser and Glaser (2010) Guadeloupe (Guyader *et al.* 2013)
• Overexploitation, coastal fisheries depletion • Illegal fishing (including spearfishing and angling) • Lack of marine reserves acting as reservoirs in the face of natural/human catastrophes	• Consider needs and claims of insular artisanal fishermen communities • Management takeover by island fisheries associations with sound scientific advice • Reconstruction of fisheries data for trends and predictions • Management of recreative fishermen • Creation and buffering of fishing preserves and marine sanctuaries • More restrictive fishing measures at the regional scale for reef fish communities	El Hierro and Tenerife, Canary Islands/ Manrique de Lara and Corral (2017); Castro et al (2019) Guadeloupe (Frotté *et al.* 2009) Galápagos (Eastern Island (Zylich et al 2014)
• Biodiversity loss, pollution, waste management and urban and industrial sewage in islands with reefs, lagoons and narrow coastal areas. • Growing population/tourism concentration along coastal strips. • Lack of trained personnel for conservation policy inforcement. • Monitoring difficulties due to natural isolation and insularity. • High economic cost of measures in impoverished island nations • Partial abandonment/substituion of traditional activities by less sustainable ones (forced by external market pressures)	• Coral reef protection and restoration • Education/formation • Enhancing environmental awareness • Mixing traditional and modern practices [enhance traditional cultural practices of exploitation, including tourism] • Preserve societal nets traditional to certain smaller island communities from external market interference • Reinforcement of environmental laws, particularly in edification and EIA [and public information process] • Long term planning of activities with environmental incidence (including tourism) • Ecological coastal zonation for allowing ample spawning and nursery areas for fish [umbrela effect for biodiversity] • Pollution and sewage monitoring and waste treatment [it should be approached on a per-island basis to increase efficiency]	Pacific Ocean Islands- Oceania; Indonesia; Philippines/ Thistlethwait and Votaw (1992); Nunn *et al.* (1999); Campbell (2000); Hay (2013); SPREP (2016) Canary Islands (Riera *et al.* 2014) Martinica (Bocquéne & Franco, 2005)
• Sea level rise; • Water temperature rise; • Ocean acidification; • Other climate change consequences (especially related to inherent vulnerability and low resources of isolated small island states) • Macro-scale oceanographic processes (El Niño or La Niña, among others)	• International compliance with climate agendas • Strengthen and adapt local socio-economic systems • Improve and adapt coastal protections with adequate engineering (may require external technological support) • Coral reef and mangrove protection and restoration will favor habitat integrity, sediment stability, fisheries nurturing, shoreline protection	Pacific Ocean Islands (including small island states); Indonesia/ Mimura and Nunn (1998); Nunn *et al.* (1999); Christie (2005); Gilman *et al.* (2006); Campbell (2009); Ellison (2010); Hay (2013) Galápagos (Eddy *et al.* 2019)
• Degradation by urbanization of intertidal rocky and sandy shores, halophytic vegetation and dune fields.	• Monitoring and protection of scarcer sedimentary coasts (*i.e.* sand dunes) • Coastal moratory	Canary Islands/ Bianchi (2004); Otto *et al.* (2007); Fernández-Cabrera *et al.* (2011); Delgado *et al.* (2017); Ferrer--Valero *et al.* (2017)
• Fisheries and shellfish over-exploitation in relation to poor self-management of local fishermen guilds ("cofradías"); commercialisation problems; por co-management strategies; illegal fishing; artisanal fishing decline	• Adapt regulations to the needs and integrate knowledge of the local fishermen's guilds; promote fishermen participation and collaborative strategies	Tenerife, Canary Islands/ Corral *et al.* (2017)

Table 2. Casuistic scheme of coastal management and conservation problems on oceanic islands worldwide. The list does not pretend to be exhaustive, but to review the most prevalent causes of environmental degradation in insular coastal ecosystems, with proposals for improvement from the local to the global scale.

Example problems	Solutions/proposals of management, protection and mitigation	Site/Reference
• Elevated population density • Tourism, beach disturbance • Agriculture (fertilizers, pesticides); eutrophication • Oversedimentation • Overfishing/aquaculture • Industrial pollution (heavy metals, hydrocarbons) • Silting and pollution of coastal lagoons and estuaries • Sewage discharges • Maritime traffic • Invasive alien species (*i.e.* algae) • Loss of seagrass meadows and associated biodiversity	• Strengthen legal protection in biodiversity hotspots • Measures to minimize eutrophication and reduce phytoplankton blooms • Functional monitoring in sensible sites • Protect sensible sites from further port and marina development; strengthen Environmental Impact Assessment	Mediterranean Sea/ European Communities (1999, 2000); Güreen *et al.* (2020); Waycott *et al.* (2009); Myers *et al.* (2000)
• Trade-off between coastal conservation and human behavior at a local scale	• Analyzing perceptual social data for improving resource management	Caribbean Sea coasts and islands, and other global coastal regions/ Aswani (2019)
• Macro- and microplastic pollution in coastal waters, beaches, and seafloor sediments, reaching even remote oceanic islands.	• Palliative cleanups • Preventive actions (prohibition/banning of packaging & other non-biodegradable plastic ítems) • Shift to biodegradable materials • Education to consumers • Control spilling from both land- (rivers, submarine sewage outfalls), and marine-based sources • Analysis of presence in trophic chains	Global islands and coasts, all oceans/ Monteiro *et al.* (2018); Ling *et al.* (2017)

where the fishing pressure has been lower and more strictly regulated in recent decades (Tuya *et al.*, 2004). Several environmental factors are likely to have played a role in the demographic explosion of *D. africanum* populations in the Canaries (Clemente *et al.*, 2009), being the overfishing of predators, *e.g.* hogfishes, snappers and groupers, the most important (Tuya *et al.*, 2004). Also, other anthropogenic factors are considered to be associated with increased abundances of *D. africanum*, such as the population density and the numbers of fishing boats (Hernández *et al.*, 2008). The outbreak of this voracious sea urchin results in commonly spreading barren grounds throughout the entire archipelago (Brito *et al.*, 2004), reaching up to 50 m depth and covering approximately 75% of rocky substrates of the Canary Islands (Barquín *et al.*, 2004). Even, small densities (< 5 ind. m^{-2}) can remove up most of the algal cover and thus reduce significantly macroalgal diversity, resulting in impoverished bottoms dominated by encrusting algae (Sangil *et al.*, 2012).

In comparison, in the Caribbean islands, the situation is radically different with major algal growths developed over reefs following the mass mortality throughout the 80s of the long-spined sea urchin *Diadema antillarum* caused by a toxic infection (Hughes *et al.*, 1985). The current population densities are *ca.* 12% of those before the die-off, and still the factors are unclear though recruitment limitation seems to be a pivotal factor (Lessios, 2016). This urchin is a keystone herbivore on Caribbean coral reefs, controlling benthic algae growth by grazing on macroalgae; this species is responsible for 40% of the grazing activity that occurs on a coral reef (Mumby *et al.*, 2006). Thus, the die-off of this sea urchin species underpinned a shifting baseline from coral-dominated to algae-dominated communities, since macroalgae compete and harm corals (Rivers and Edmunds, 2001).

Climatic variables are primary drivers of distributions and dynamics of marine communities in coastal ecosystems (Parmesan 2006). However, global climate change is affecting the marine realm in different ways, *e.g.* warming waters, acidification and anoxia, and thus, several species are favored by these changes compared to other. Moreover, coastal ecosystems are subjected to multiple drivers of human-induced environmental changes. One of the consequences of these changes is the increase of opportunistic fast-growth species, better adapted to the current fluctuating conditions. These outbreaks in turn may cause changes in the structure and function of the broader ecological community, with modifications in physical-chemical properties of the ecosystem (Valiela *et al.*, 1997). Most of the blooms are normally formed by opportunistic algae, which are a natural component of shallow subtidal marine communities, however, humans promote the magnitude, frequency and duration of their proliferation by increasing nutrient loads in coastal waters, *e.g.* pipelines, run-offs, etc. (Eriksson *et al.*, 2009).

In the Canary Islands, blooms of the blue-green algae *Lyngbya majuscule*, linked to declines in the abundance of the seagrass *Cymodocea nodosa*, have been observed in the eastern islands (Martín-García *et al.*, 2014). In addition, two green algae (*Caulerpa racemosa* and *C. prolifera*) have proliferated in cover and extension in *C. nodosa* meadows (Tuya *et al.*, 2014a). These algae are better adapted to eutrophic conditions, and even manipulative experiments have shown faster growth rates of *Caulerpa* occupying *C. nodosa* habitats (Tuya *et al.*, 2013). In addition, the growth of *Caulerpa* is favored by high sedimentation and resuspension rates (Williams 2007). Other factors that are steadily increasing in importance affecting seagrass ecosystems include pollution from several industrial and aquaculture activities (Riera *et al.*, 2013).

2.2.2 Non indigenous species (NIS)

Probably, a net gain of species has been shown in the last decades as consequence of the non-indigenous species reports, mostly associated to fouling (Godwin, 2003) and shipping traffic (Ware *et al.*, 2014). However, this species increase is not relative to the species loss that is positively correlated to the human pressure on the coastal environment, *e.g.* Canary Islands (Riera *et al.*, 2014).

Non-native, introduced marine organisms can act as competitors, predators and as ecological engineers in the recipient ecosystems (Wallentinus and Nyberg 2007). The threat to biodiversity posed by invasive taxa is increasing as the arrival of new species is facilitated by ocean warming and international maritime traffic (Occhipinti-Ambrogi and Savini 2003; Molnar *et al.*, 2008). For example, the species *Zoobotryon verticillatum*, originally from the Mediterranean Sea, have greatly expanded its biogeographic range by the proliferation of marinas and vessel traffic in the Atlantic Ocean (Minchin 2012).

Hull fouling, ballast waters and sediments from commercial ships and recreational yachts constitute important vectors for introducing NIS to oceanic archipelagos. Oil platforms have also recently been identified as a vector of exotic tropical fish from Brazil, Guinea, or the Caribbean (Falcón *et al.*, 2015). Sea warming is also an environmental factor creating favorable conditions for an increase in tropical and subtropical fauna and flora, partially due to latitudinal shifts or range expansions (*e.g.* Brito *et al.*, 2005). However, in several cases the arrival of exotic species cannot be plainly interpreted as a human-mediated invasion, but as a spontaneous process, although human activities can introduce favoring synergies (*i.e.* oil platforms and ships as vectors for algae, invertebrates and fishes, fish farming cages) in an already-occurring tropicalization scenario.

Ballast water transported by globally increased ship traffic, along with other interacting factors such as nutrient imbalances, salinity, acidity and temperature changes, and coastal degradation, may be enhancing the risk of toxic microalgal outbreaks (Ruiz, 2000). Shipping traffic has other inherent consequences concerning antifouling products, especially in the last decades after the ban-use of Tributyltin (TBT) due to its toxicity and persistence. Currently, other biocides for ship hulls of any length have been tested (Sánchez Rodríguez *et al.*, 2011); however, their toxicological effects are not well studied, especially for non-target species, *e.g.* gastropods and bivalves (Sánchez Rodríguez *et al.*, 2011).

3. FUTURE SCENARIOS

The current environmental situation in most of oceanic archipelagos is a consequence of several factors and will be accentuated in the next decades, with an increase of coastal degradation and the number of environmental perturbations that interact and establish ecological synergies deserving further attention.

3.1 Prospects of a gradual tropicalization of the Canary islands

Seawater warming has profound consequences on coastal populations, community composition and ecosystem functions (IPCC 2014). For example, the tropicalization of ecosystems drives changes in herbivory, with a deforestation of temperate algal forests and seagrass beds due to the expansion of tropical herbivores (Vergés *et al.*, 2014). In some coastal temperate areas, the increase of sea surface temperature and a rise of tropical fish abundance have coincided with a dramatic decline in macroalgal beds (Nagai *et al.*, 2011). In the last decades, sea warming has been starting to affect the structure of coastal ecosystems in the Canaries with the appearance of species with tropical affinities, from Cape Verde, the West African coast and the Caribbean as main biogeographic origins (Riera *et al.*, 2014).

The growing prevalence of tropicalized climate, *i.e.* sea warming temperature (Parrilla *et al.*, 1994; Santos *et al.*, 2012), including warming of the Canary Current upwelling, (Arístegui *et al.*, 2009), and other influential phenomena such as increases in algal fertilization from Saharo-Sahelian dust plumes (Alonso-Pérez *et al.*, 2007), may establish favorable conditions for phytoplankton blooming events. Many new species are being described with the mentioned potential for ecological change of coastal ecosystems and with repercussions in other environments, as occurred to the recently discovered dinoflagellates *Gambierdiscus excentricus* (Fraga *et al.*, 2011) and *G. silvae* (Fraga and Rodríguez 2014).

An obvious tropicalization of littoral fish fauna has been experienced in the Canarian archipelago over the last two decades (Falcón *et al.*, 2015). Examples of this process are the ocean triggerfish (*Canthidermis sufflamen*) and goldspot goby (*Gnatholepis thomsoni*) (Brito *et al.*, 2005). In contrast, there are evidences showing the regression of temperate species such as the seastar *Marthasterias glacialis* (Authors *pers. obs.*) and fucoid algae, *i.e. Fucus guiryi* (Riera *et al.*, 2015). Even a decreasing trend in coral populations has been observed in the black coral *Antipathella wollastoni* at shallow seabeds throughout the Archipelago, with most of the remaining populations at higher depths (> 45 m) forming dense aggregations (Martín-García 2013).

3.2 Fisheries collapse

Catcheries have shown clear symptoms of overfishing throughout the last decades (García-Mederos *et al.*, 2015). The steadily increasing pressure from recreational fishermen is a factor of utmost importance to take into account for future works on fisheries effort and landings. Professional fishers are currently limited to certain coastal settlements, focused on a very limited array of target species (Balguerías 2001), and represent a minor fraction of the landings. The collapse of several commercial species is eminent in the near future since the fishing pressure is not decreasing and stocks are on the brick of sustainability for most of the species, with special emphasis on groupers and snappers (García-Mederos *et al.*, 2015).

Riera *et al.*, (2016) have shown a dramatic decrease of two commercial limpet sizes (*Patella candei crenata* and *Patella aspera*), indicative of overharvesting, mainly conducted by recreational harvesters. The limpet size is considered a surrogate of the state of the conservation of their stocks, and thus, the low viability of limpet populations is evident in the Canarian archipelago.

3.3 New pollution concerns

Increasing coastal human presence and seaside recreation at a global scale is related to pollution through microplastics, metals, ultraviolet (UV) filters and other pollutants that bioaccumulate in marine organisms with potential risks for both humans and the marine environments (*e.g.* coral bleaching by viral infections, among other effects) (Danovaro *et al.*, 2008; Sánchez Rodríguez *et al.*, 2015). The rise of plastic in human industrial chains means a new quanti- and qualitative leap in our way of damaging marine ecosystems. It represents the epitome of an unsustainable use of the oceans as a garbage dump.

In this sense, recent studies have alerted of pervasive presence of microplastics in sandy substrata, although biotic effects are still poorly known (Baztan *et al.*, 2014). Little is known of the long-term effects of plastics and microplastics at different levels of the marine food chain. The type of compounds derived from the degradation of plastics, associated with the type of damage they can cause in different types of organisms, including humans, is scarcely known. Methods to characterize and identify composition and origin of plastic debris are being developed (La Daana *et al.*, 2017, Serranti *et al.*, 2018). A low prevalence of microplastics has been found in fish caught in coastal waters of the South Pacific. This has been explained by low human presence in that region and by the great dynamism of the currents that move coastal water outwards through upwelling and (Ory *et al.*, 2018). However, contamination by plastics in coastal waters and beaches of the Atlantic islands (*e.g.* Makaronesia and the Caribbean) is already an important problem (Monteiro *et al.*, 2018). The information again seems to be more focused on sandy beach environments because these are the most accessible for human use. Other coastal configurations, dominant in certain volcanic islands, are poorly studied. In a study from beaches in the Canary Islands, the characteristic debris types collected pointed to open-sea sources rather than land-based or local ones (Herrera *et al.*, 2018). However, the relative importance of diffuse sources compared with point sources of plastic discharge to the sea is not well known, although certainly a very high percentage would come from river discharge and land-based industries and cities (Herrera *et al.*, 2018).

The long-term global effects of microplastics and other pollutants on biodiversity and the integrity of marine trophic networks are difficult to predict. Certain questions remain thus open. How do microplastics interact with other types of contaminants? What are its short and long-term effects on food webs and finally on human health? How can pollution by microplastics affect fisheries? Above all, what mechanisms and strategies are we applying to tackle this global problem?

3.4 Extensive coastal degradation

Coastal development affects over 80% of the coastline in many European regions, and it is responsible for much of the observed habitat degradation and loss (Airoldi and Beck, 2007). Coastal occupation underpins profound changes in ecosystem functioning, which are not limited to short-term effects since drivers or consequences that may be developed because of these changes (Biggs *et al.*, 2009). Ecological consequences of coastal transformation have not been fully appreciated or brought into a proper conservation context, due to a preponderance of management implementations and infrastructure schemes applied in response to emergence situations, or in exclusive dedication to human welfare, leisure or economic criteria (Semeoshenkova and Newton 2015).

Degradation of coastal and marine ecosystems through urbanization has been increasing on the islands during the last half century, long before the first marine reserves were firstly conceived and proposed in the 1980s (*e.g.* Bacallado *et al.*, 1989). Coastal development, urbanization, intensive agriculture and mass tourism pressure have contributed to an irreversible transformation of several coastal habitats from oceanic archipelagos (Otto *et al.*, 2007; Bertocci *et al.*, 2016). Such activities interact (and compete) with other activities in the primary sector such as both recreational and traditional fisheries, as well as aquaculture industry (Chuenpagdee *et al.*, 2013).

On the other hand, the construction of "mega" infrastructures such as ports affects extensive coastal areas (Sundblad and Bergström, 2014). The development plans of several islands include the construction of future harbor infrastructures, as well as the creation of artificial beaches with subsequent beach nourishment actions. The environmental affections of these infrastructures are not exclusively limited to the coastal area but to the surrounding environment, with changes in hydrodynamics, physico-chemical and biological factors, among others (Martins *et al.*, 2016).

Besides, impacts of land logistic facilities and transport infrastructure associated to harbors and marinas would add additional impacts (*i.e.* changes in water runoff affecting sedimentation, and pollution sources) at different levels upon mesolittoral and sublittoral habitats. Moreover, such land changes add and accumulate with landscape transformation (urbanization, agriculture; see Otto *et al.*, 2007); this implies indirect effects on the marine environments. The stakeholders have traditionally shown reticence against the creation of marine reserves and protection of relevant marine environments around the islands, although already even some fishermen associations have acknowledged the benefits on the long term, for example in the MPA Mar de las Calmas (El Hierro) (Jentoft *et al.*, 2012).

Finally, the worldwide approach to the conservation schemes of species is becoming growingly reactive and species-focused, rather than adaptive and habitat- or ecosystem-focused. This approach seems to favor species over habitats as protection subjects, and responds to a generalized prioritization of the development of coastal infrastructure and logistic schemes over efficient habitat conservation. In terrestrial coastal ecosystems, it has been suggested that protection of relatively small areas does not avoid massive loss of natural areas (Otto *et al.*, 2007). Far from slowing down, the last decades have seen further irreversible coastal degradation and destruction of natural areas due to coastal development. In the Canary Islands, this has occurred simultaneously to reductions in protection status of concrete figures for both species and habitats, and to the inclusion of "selected" taxa in protection lists within the category "Species of Interest for the Canary Islands", rather than enhancing protection of certain idiosyncratic and singular environments whose preservation would render improved conservation status of species in the future, if basic principles of biological conservation were applied.

Nevertheless, in the last decade, conservation efforts have shown some positive results, with the recovery of populations of endangered species such as, cetaceans, turtles and seals (Bejder *et al.*, 2016, Piacenza *et al.*, 2016, Magera *et al.*, 2013), as well as signs of reverse of the decline of exploited species, *e.g.* fish commercial stocks in several areas where they were depleted by industrial fleet (*e.g.* Froese *et al.*, 2018). Recently, Duarte *et al* (2020) outlined the importance of fulfilling the Sustainable Development Goal 14 of the United Nations aims to "conserve and sustainably use the oceans, seas and marine resources for sustainable development". This goal is achieved by rebuilding not only species on decline but ecosystems that have disappear or in at risk of destruction from human-driven perturbations. In the last decade, an increase number of restoration projects and initiative have been developed worldwide on declining coastal ecosystems, namely mangroves, seagrass meadows, kelp forests and coral reefs.

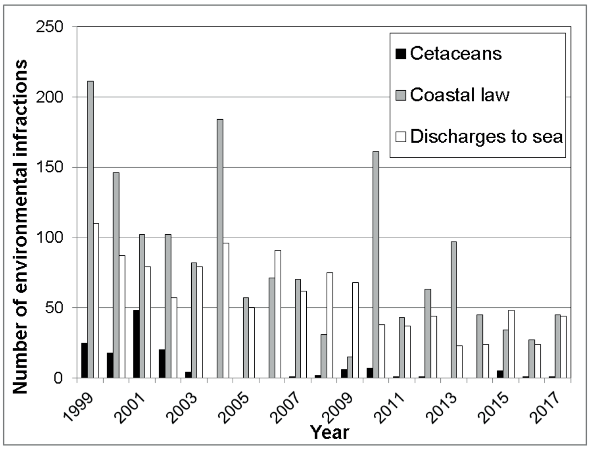

Figure 2. Evolution of the absolute number of environmental infractions related to marine ecosystems in the Canary archipelago. The data relating to infractions on cetacean conservation, illegal discharges to the sea and infractions of the Coast law are shown. Data source: ISTAC (Instituto Canario de Estadística; Estadística de Vigilancia Ambiental / Resultados principales. Islas de Canarias. 1999-2017).

4. CONCLUSIONS

The increase in resource exploitation, extractive activities and use and abuse of the coastline entail onerous environmental costs in the Canarian archipelago and other islands worldwide. Society-environment positive synergies should be pursued to ameliorate exacerbated pressure on island coastal resources. Involvement of public administrations, citizenship environmental education to minimize diffuse pollution, updated and effective fisheries management supported by a strict surveillance and gremial participation, monitoring of ballast waters and hull fouling of vessels and oil platforms, and legal control of development within integrated coastal zone management (ICZM), are all urgently needed measures for sustainable future scenarios in most oceanic archipelagos.

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ASSESSMENT IMPACT OF THE DAMIETTA HARBOUR (EGYPT) AND ITS DEEP NAVIGATION CHANNEL ON ADJACENT SHORELINES

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ABSTRACT:

Deep navigation channels have a great impact on adjacent beaches and crucial economic effects because of periodic dredging operations. The navigation channel of the Damietta harbour is considered a clear example of the sedimentation problem and deeply affects the Northeastern shoreline of the Nile Delta in Egypt. The aim of the present study is to monitor shoreline using remote sensing techniques to evaluate the effect of Damietta harbour and its navigation channel on the shoreline for the last 45 years. Also, the selected period was divided into two periods to illustrate the effect of man-made interventions on the shoreline. Shorelines were extracted from satellite images and then the Digital Shoreline Analysis System (DSAS) was used to estimate accurate rates of shoreline changes and predict future shorelines evolution of 2030, 2040, 2050 and 2060. The Damietta harbour created an accretion area in the western side with an average rate of 2.13 m year^{-1}. On the contrary, the shoreline in the eastern side of the harbour retreated by 92 m on average over the last 45 years. So, it is considered one of the main hazard areas along the Northeastern shoreline of the Nile Delta that needs a sustainable solution. Moreover, a detached breakwaters system is predicted to provide shore stabilization at the eastern side as the implemented one at Ras El-Bar beach. Predicted shoreline evolution of 2060 shows a significant retreat of 280.0 m on average.

Keywords: Navigation Channel; Shoreline; Damietta Harbour; Remote Sensing; DSAS.

RESUMO:

Os canais de navegação profundos têm um grande impacto nas praias adjacentes e efeitos econômicos cruciais devido às operações de dragagem periódicas. O canal de navegação do porto de Damietta é considerado um exemplo claro do problema de sedimentação e afeta profundamente a costa nordeste do Delta do Nilo, no Egito. O objetivo do presente estudo é monitorar a linha costeira usando técnicas de sensoriamento remoto para avaliar o efeito do porto de Damietta e seu canal de navegação na linha costeira nos últimos 45 anos. Além disso, o período selecionado foi dividido em dois períodos para ilustrar o efeito das intervenções feitas pelo homem na costa. As linhas costeiras foram extraídas de imagens de satélite e, em seguida, o Digital Shoreline Analysis System (DSAS) foi usado para estimar taxas precisas de mudanças na linha costeira e prever a evolução futura das linhas costeiras em 2030, 2040, 2050 e 2060. O porto de Damietta criou uma área de acreção no lado oeste com uma taxa média de 2.13 m ano^{-1}. Em contrapartida, a linha da costa no lado oriental do porto recuou 92 m em média nos últimos 45 anos. Portanto, é considerada uma das principais áreas de risco ao longo da costa nordeste do Delta do Nilo que precisa de uma solução sustentável. Além disso, um sistema de quebra-mares isolado está previsto para fornecer estabilização da costa no lado leste como o implementado na praia de Ras El-Bar. A evolução da linha costeira prevista para 2060 mostra um recuo significativo de 280 m em média.

Palavras-chave: Canal de navegação; Costa; Porto de Damietta; Sensoriamento remoto; DSAS.

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1. INTRODUCTION

Navigation channels are a determined path dredged in the sea bed to enable ships to access harbours or to pass through countries. The channel extends from port location to the required water depth in the sea, where depth and width of the channel depend on the ship dimensions. Navigation channels usually act as sediment traps and interrupt sediment movement from various directions due to waves and currents. Sedimentation happens when suspended load reaches the channel location with bigger water depth and low current velocity. Hence, construction of deep navigation channels usually affects the contiguous beaches, because of losing suspended material that is trapped in navigation channels.

In deep navigation channels, periodic dredging is usually needed to ensure the safe passage of vessels and maintain the water depth of navigation channel at the required level, but the volume of dredged material varies from a location to another. In Argentina, El-Toro navigation channel required periodic dredging to remove 300 000 m^3year^{-1} of sediment (Perillo and Cuadrado, 1991). In Thailand, the approach channel of Bangkok harbour is maintained through continuous dredging of 5 million m^3year^{-1} of sediment (Deguchi *et al.*, 1994). Moreover, the entrance channel at Blankenberge harbour, Belgium, had a total amount of sediment of 165 000 m^3year^{-1} (Zimmermann *et al.*, 2012) and the estimated rate of sediment in the navigation channel of LNG port at Abu-Qir bay on the northwestern coast of Nile Delta, Egypt, was 1 977 000 m^3year^{-1} (Deabes, 2010). Furthermore, the international navigation channel in Dinh An Estuary, Vietnam, used to be dredged twice a year, where siltation volume reached 1 250 000 m^3year^{-1} (Nguyen *et al.*, 2013), and the average rate of siltation in Saint John harbor, Canada, was 4 200 000 m^3year^{-1} (Leys *et al.*, 2011). Also, the navigation channel at Damietta port trapped longshore sediment and gained sedimentation with an average rate of 2 000 000 m^3 $year^{-1}$ (Frihy *et al.*, 2016).

Dredging activities in navigation channels and man-made interventions in the coastal system contribute to the erosion of the adjacent beaches (Da Silveira *et al.*, 2012). Changing the alignment and depth of the navigation channel at Dar el Sallam harbor, Tanzania, caused high erosion in Ras Makabe and the sedimentation rate increased to 168 000 m^3year^{-1} (Sanga and Dubi, 2004). So, the construction of a deep navigation channel has clear environmental effects on the adjacent shoreline.

The shoreline is the interface between sea and land and represents one of the main natural borders for many countries, and can be used for tourism, recreation activities, urbanization and industrial and infrastructures activities. Assessment and analysis of shoreline changes are very important in coastal area management, understanding of morphological processes, computing sediment budget and identification of hazard zones (Danforth and Thieler, 1992). Monitoring shoreline changes can be used to illustrate for instance the environmental effect of navigation channels on adjacent beaches. Also, an erosion/ accretion pattern reveals the natural processes of wave-induced longshore current and sediment transport (El Banna and Herehe, 2009).

From 1807 to 1927 ground surveying was the only way used for shoreline mapping. Aerial photographs were used for that purpose in the period from 1927 to 1980, and then from 1972 Landsat and other satellites provide digital imagery in infrared spectral bands that facilitate monitoring operations. Using satellite images in shoreline monitoring has many advantages since optical images are simple to interpret and easily obtainable. In addition, absorption of infrared wavelength region by the water make such images an ideal combination for mapping. Besides, it is not time-consuming, inexpensive to implement and it has large ground coverage.

Landsat images were used in shoreline monitoring in the last decades to evaluate shoreline response to coastal structures and determine accretion and erosion pattern. El Banna and Hereher (2009) used satellite images for analyzing the coast of Sini, Egypt, in the period between 1986 and 2001. Dewidar and Frihy (2010) used ten scenes of Landsat satellite images for 35 years from 1972 to 2007 to quantify erosion and accretion along Northeastern shoreline of Nile delta in Egypt. El-Sharnouby *et al.* (2015) used ten Landsat images for monitoring shoreline changes of Gamasa beach, Egypt, with 30 km length over 30 years. Moreover, the impact of port construction, siltation, land reclamation and urban development on Ningbo shoreline, China was evaluated by seven Landsat images for 40 years from 1976 to 2016 (Wang *et al.* 2017). Also, the rates of change of the North-Holland coast, Netherlands, were estimated by 13 Landsat images from 1985 to 2010 (Do *et al.* 2019). El nabwy *et al.* (2020) used eight Landsat satellite images for 33 years from 1985 to 2018 to estimate the shoreline changes in the northeast shoreline of the Nile Delta, Egypt, and the effect of the seawalls on it.

The Northern coastline of the Delta in Egypt has suffered from erosion in many spots in the last decades due to blockage of sediment discharge as a consequence of the Aswan High

Dam construction and constructed barrages across Nile River. Damietta harbour and its navigation channel are considered a clear example of sedimentation problem and impacts on adjacent beaches.

El-Asmar and White (2002) used remote sensing and field survey to assess shoreline changes consequent to harbour construction from 1983 to 1993. It was found that harbour jetties interrupted eastward-moving littoral drift. Therefore, the western beach had accretion of fine sand with an average rate of 25 m $year^{-1}$ from 1983 to 1993. In contrast, the eastern shoreline was losing coarse sand that was trapped in the navigation channel and the erosion rate reached -26 m $year^{-1}$. Frihy *et al.* (2004) evaluated the construction of large-scale detached breakwater systems on the Nile Delta coast of Egypt at Ras El Bar beach with a monitoring program spanning the years 1990 to 2002 and beach nearshore profiles. The preconstruction beach erosion at Ras El Bar (-6 m $year^{-1}$) was replaced by the formation of a sand tombolo (35 m $year^{-1}$) and salient (9 m $year^{-1}$). On the other hand, beach erosion substantially increased in the downdrift sides and reached -9 m $year^{-1}$.

Abo Zed (2007) used 18 bathymetric profiles to study the erosion/accretion pattern before the construction of Damietta harbour (1978-1982) and after the construction (1988 – 1997). The area was marked as an accretion zone before construction of harbour and then after the construction, it was observed that there was an accretion zone in the western side and an erosion zone in the eastern side of the harbour. The annual net rate of littoral drift on the western side was about 143 000 m^3 (accretion) and on the eastern side was about 254 000 m^3 (erosion).

El-Asmar *et al.* (2016) used 5 scenes from 1973 to 2015 to monitor the shoreline in the eastern side of Damietta harbour, that was an erosive segment with high shoreline retreat. Khalifa *et al.* (2017) used 32 bathymetric profiles to evaluate shoreline changes in the surrounding area of Damietta harbour from 2010 to 2015. The data revealed an accretion zone in the western side with an average rate of 15 m $year^{-1}$ and erosion pattern at eastern part with an average rate of – 6.5 m $year^{-1}$.

Previous literature proves that man-made interventions such as the construction of dams, deep navigation channels and breakwaters deeply affect shoreline morphology.

This research aims to evaluate the impact of harbours and deep navigation channels on neighboring beaches. Damietta harbour with deep channel could be a good example and case study for this purpose. The rate of shoreline change needs to be updated and estimated using high precision and accurate method. Furthermore, prediction of future shoreline could clarify the effect of the harbour on the shoreline in the next decades. Remote sensing technique was used in shoreline monitoring and then, DSAS was used to determine the rates of shoreline change using statistical approaches.

2. THE CASE STUDY OF THE DAMITTA HARBOUR

2.1. Introduction

In the 1980s, the Egyptian authority decided to establish the Damietta harbour on the northeastern side of the Nile Delta near the New Damietta city to improve trade and economy potential along the Mediterranean Sea. The selected location 9.7 km west of the Damietta Nile branch is characterized by a minimum wave and current effect. However, the location was described as one of long-term coastal accretion area (El-Asmar and White, 2002).

In 1982, the breakwaters of Damietta harbour were constructed to protect the harbour entrance from siltation. The western breakwater was constructed parallel to the navigation channel with 1500 m length and extends to 7.0 m water depth and the eastern breakwater was constructed perpendicular to the shoreline with 500 m length and extends to 3.0 m water depth. The navigation channel was completed in 1984, with a total length of 20 km, 15 m average water depth and 200 m width of the inner part that increases to 300 m in the outer part (Figure 1).

As a result of the harbour construction, the shoreline in the eastern side suffered erosion and Ras El-Bar beach began to retreat, threatening tourism and recreational activities. So, from 1991 to 2002, a detached breakwater system consisting of eight breakwaters was constructed in Ras El-bar area to protect the shoreline. Breakwaters are 200 m long with 200 m gaps between each other and were constructed 400 m offshore at 4.0 m water depth.

2.2. Study area description

The study area is the shoreline around Damietta harbour with a total length of 26.2 km, and it was divided into three zones (Figure 2). Zone A extends from Gamasa drain to the western breakwater of Damietta Harbour with a length of 18.50 km. Zone B extends from the eastern breakwater to the location of detached breakwaters with a length of 4.5 km. Zone C extends beyond the detached breakwaters of Ras El-Bar beach with a length of 3.2 km. The aim of the present study is to monitor

Figure 1. Layout of Damietta harbour.

shoreline changes to evaluate the effect of Damietta harbour and its navigation channel on the shoreline for the last 45 years. The selected period was divided into two periods, the first period was from 1973 to 1994 to analyze the changes in shoreline before and after Damietta harbour construction and before the construction of shore protection structures. The second period was from 1995 to 2018 and it illustrates shoreline changes after the construction of detached breakwaters at zone C.

2.3. Meteorological analysis

Coastal processes depend on meteorological factors such as wind, waves, tide, and current. So, analysis of the metrological data is very important for the assessment of coastal changes. The predominant wind direction at northeastern coast of Nile Delta is between N and NW direction with an average wind speed of about 3.75 m s^{-1} (Khalifa, 2017).

According to records from 2001 to 2004, the predominant waves in the Damietta harbour come from NW direction (76.8%) and a small portion of waves comes from NE direction (13.6 %) and other waves come from SE (5.3%) and SW (4.3%), Figure 3. The maximum wave height was 4.8 m with significant wave height and a period of 1.10 m and 6.7 s, respectively. Collected data was measured by Coastal Research Institute (CoRI) and Hydraulic Research Institute (HRI), Egypt. It is obvious that, the prevailing wave direction is NW and it is responsible for generating net longshore sediment transport from west to east. However, a reverse current is generated toward SW direction in winter seasons with an average velocity of 0.30 m s^{-1}. Also, referring to the measurement of water surface elevations, the North-eastern shoreline of Nile Delta has semidiurnal tide and maximum tidal range (H.H.W.L – L.L.W.L) is about 0.75 m.

From the previous analysis, it is obvious that sediment flux is transporting in both directions from west to east and from east to west, due to currents and waves. So, N-E orientation of the navigation channel interrupts moving littoral drift from both directions and affects the shoreline in both western and eastern sides.

3. MATERIALS AND METHODS

The monitoring program followed many steps such as data acquisition, shoreline extraction and shoreline analysis, analyze

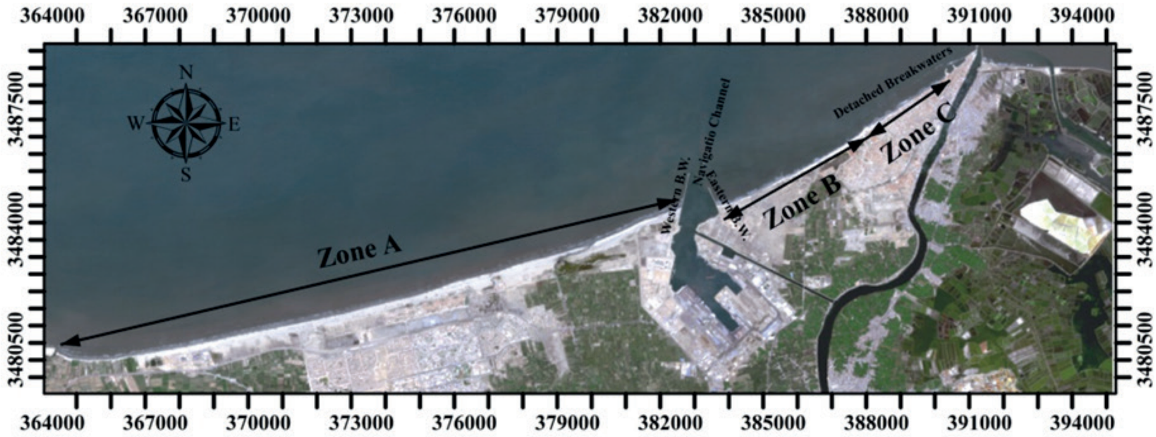

Figure 2. Study Area Zones.

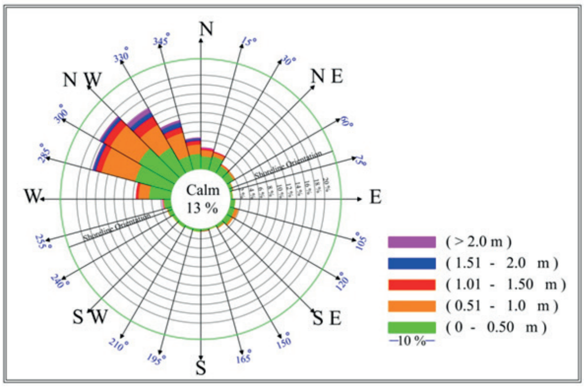

Figure 3. Wave Rose at Damietta harbor.

shoreline changes, determination of hazard areas among the study area and prediction of future shorelines (Figure 4).

3.1. Data acquisition

Satellite images were acquired from earth explores sites maintained by United States Geological Survey (USGS) and from Landsat satellite dataset. Landsat images were acquired in unequal intervals to ensure data acquisition in good weather condition with low cloud cover to reduce the error due to variability of shoreline position and minimize the influence of the tide (Moore, 2000). Table 1 illustrates scenes properties such as date, satellite name, sensor type and image resolution.

3.2. Image processing

Acquired images were subjected to image processing such as geometric correction, radiometric correction and atmospheric correction using ENVI V5.30 software. Geometric correction is performed to eliminate distortion related to scale variation, tilt

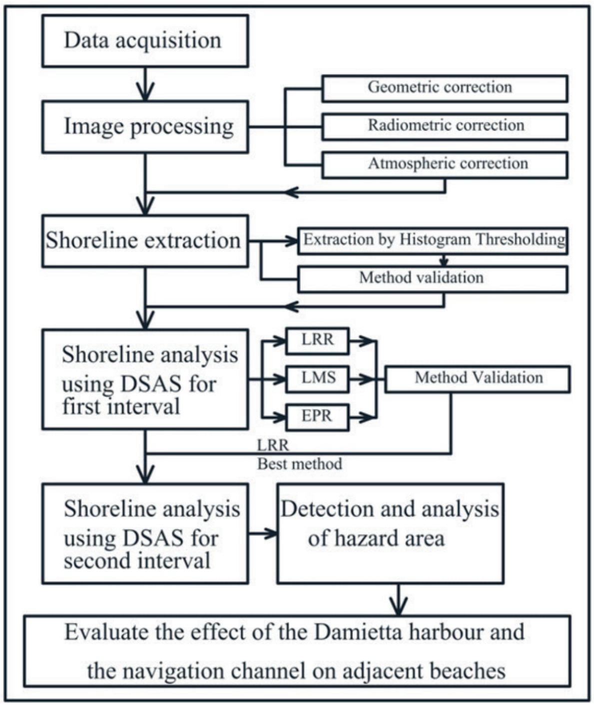

Figure 4. Flowchart of the monitoring program.

and lens distortion, and it is carried out from Ground Control Points (GPCs). Satellite images were geometrically rectified to the Universal Transverse Mercator (UTM) with a spheroid and datum of WGS 84, Zone 36. Radiometric calibration means a conversion of recorded DN values (0-255) to at-surface reflectance. Atmospheric correction is removing the effects of the atmosphere to produce surface reflectance values and can significantly improve the interpretability and use of an image.

Table 1. Satellite images properties.

Period	Acquisition Date	Satellite	Acquisition Time	Water Level (m)	Sensor Type	Scene Path/row	Spatial Resolution (m)
I	1973-01-04	Landsat 1	8:00	0.65	MSS	190/038	60.0
	1978-05-20	Landsat 3	7:47	0.53	MSS	190/038	60.0
	1984-04-29	Landsat 5	7:50	0.65	TM	176 / 038	30.0
	1987-03-21	Landsat 5	7:45	0.39	TM	176 / 038	30.0
	1990-03-29	Landsat 5	7:43	0.24	TM	176 / 038	30.0
	1992-04-03	Landsat 5	7:47	0.57	TM	176 / 038	30.0
	1994-04-09	Landsat 5	7:43	0.60	TM	176 / 038	30.0
II	1995-03-11	Landsat 5	7:33	0.51	TM	176 / 038	30.0
	2000-11-11	Landsat7	8:13	0.82	ETM	176 / 038	30.0
	2005-05-25	Landsat 5	8:10	0.52	TM	176 / 038	30.0
	2010-04-05	Landsat 5	8:14	0.54	TM	176 / 038	30.0
	2013-04-13	Landsat 8	8:25	0.37	OLI/TIRS	176 / 038	15.0
	2016-03-04	Landsat 8	8:23	0.55	OLI/TIRS	176 / 038	15.0
	2018-03-26	Landsat 8	8:22	0.60	OLI/TIRS	176 / 038	15.0

3.3. Shoreline extraction

There are different methods for shoreline extraction from satellite images. Histogram thresholding on one of the infrared bands is widely used method in shoreline delineation since the reflectance of water is nearly equal to zero and reflectance of land is greater than water. For example, band 5 is the best for extracting land-water interface in case of TM or ETM satellite images (Niya *et al.,* 2013, Alesheikh *et al.*, 2007). After the image processing, final binary images were obtained and were processed in ArcGIS10.2 software to extract the shoreline. Each image was converted from raster to vector to obtain the clear interface between land and water as the shoreline. Figure 5 shows sample of the extracted shoreline over the period of study.

3.4. Uncertainty in of shoreline change

Using satellite images in the analysis of shoreline changes has a range of error. The source of error may be due to geo-referencing error and short-term variability of shoreline position, (Louati *et al.*, 2015, Kumar *et al.* 2010). For the selected satellite images the RMSE in geo-referencing process did not exceed 0.6 pixels, that is considered an acceptable error (Kabir *et al.* 2020). So, the maximum geo-referencing error is ±28.5 m for the period (1973-1984), ±9.4 m for the period (1984-2013) and ±7.5 m for the period (2013-2018). On the other hand, most of Landsat scenes were acquired in spring period and during calm sea condition that tidal ranging 0.1 to 0.50 m. Hence, the predicted shift in shoreline position varies from movement ±2.50 m to ±12.5 m according to the beach slope. Regarding the spatial resolution of Landsat images that varied from 60×60 m, 30×30 m and 15×15 m, the Landsat images could be used in analysis of shoreline changes neglecting the tidal effect (Louati *et al.*, 2015; El-Sharnouby *et al.* 2015; Kuleli, 2010; Dewidar and Frihy 2007; Guariglia *et al.* 2006). So, the total error based on geo-referencing error is ±3.86 m, ±1.01 m, ±3.06 m and ±0.97 m for the period (1973-1984), (1984-2013), (2013-2018) and (1973-2018) respectively.

3.5. Validation of Shoreline Extraction Method

To validate the shoreline extraction method and determine its accuracy, the extracted shoreline from the satellite image of 2013 was compared with another shoreline that was generated by a field survey for the same year. The comparison between shorelines was carried out by the Digital Shoreline Analysis System (DSAS) using a statistical approach of End Point Rate (EPR). Results showed a good agreement between the extracted shoreline from the satellite image and that from the field survey with Mean Absolute Error (MAE) of 6.3 m (Figure 6.a). Figure 6.b shows that 90 % of the values have small errors ranging between -5 and +15 m (20 m). The error may be because of geo-referencing process or short-term variability of shoreline position, however it can be considered an acceptable error according to the used the pixel size of the satellite image (15.0 m) (Wang *et al.* 2017).

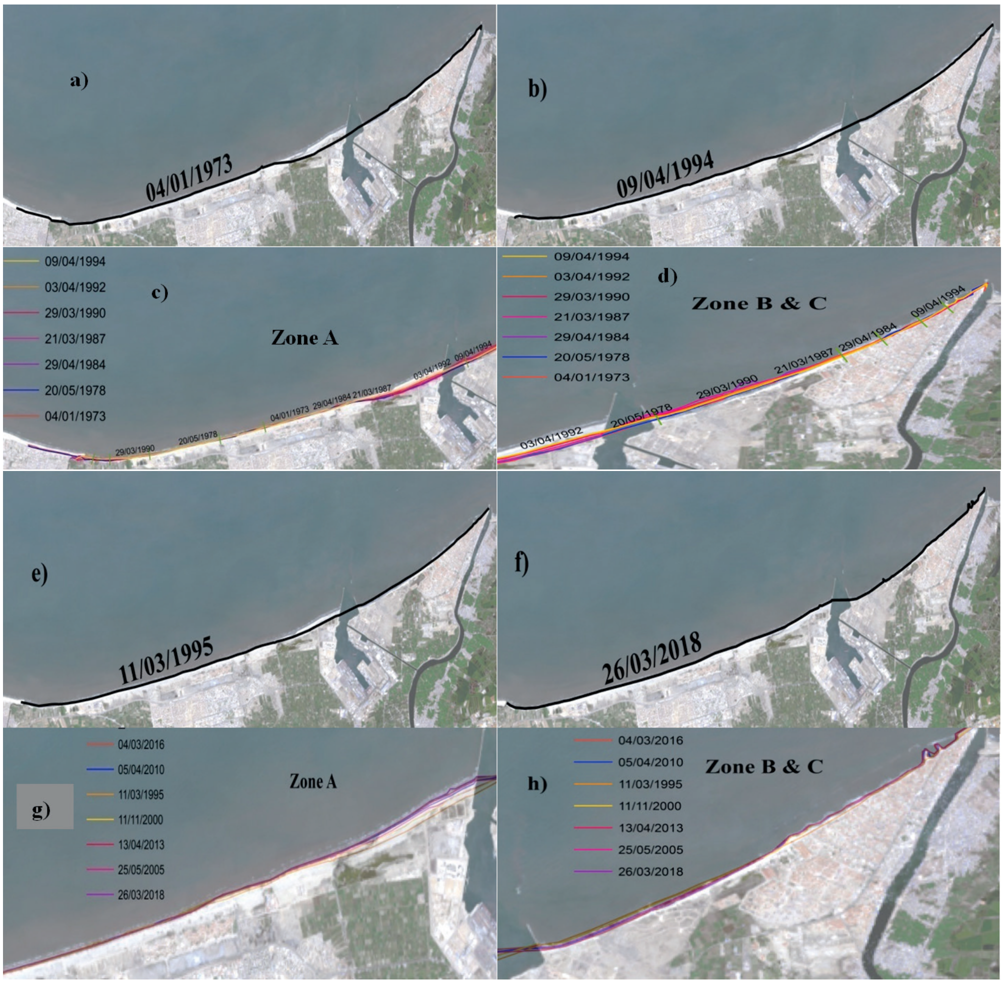

Figure 5. a,b) Satellite images for the two years 1973 and 1994, c) Extracted shorelines for the first period (1973–1994) For zone A, d) Extracted shorelines for the same period for zone B and C e,f) Satellite images for the two years 1995 and 2018, g) Extracted shorelines for the second period (1995–2018) for zone A, h) Extracted shorelines for the same period for zone B&C.

3.6. Shoreline analysis using DSAS

DSAS is a software extension within the Environmental System Research Institute (ESRI) ArcGIS. DSAS can be used in the analysis of shoreline changes, computing erosion/accretion rates over historical timescales and prediction of the shoreline evolution as an indicator of future trend assuming constant physical nature. Different statistical approaches such as End Point Rate (EPR), Linear Regression Rate (LRR) and Least Median of Squares (LMS) were used in the calculations. Shorelines are merged into one feature class to perform statistical analysis. Transects with 10 m spacing, were generated perpendicular to this baseline and intersected the shorelines to establish measurement points. Statistical approaches were used to calculate the average rate of shoreline changes.

4. ANALYSIS OF RESULTS AND DISCUSSION

4.1. First period (1973 – 1994)

Statistical methods of EPR, LRR and LMS were used to calculate the rates of change in both study periods. Table 2 summarizes the results and the rates of change for each zone for the first period.

Also, Figure 7 shows the variable rates of shoreline change along the study area using the different methods.

To validate the statistical results and determine the most accurate method in calculating shoreline rates of change, the results from statistical calculations were compared with other results that were generated from field ground survey by Frihy and Komar (1993) at the same locations (Figure 8). Frihy and

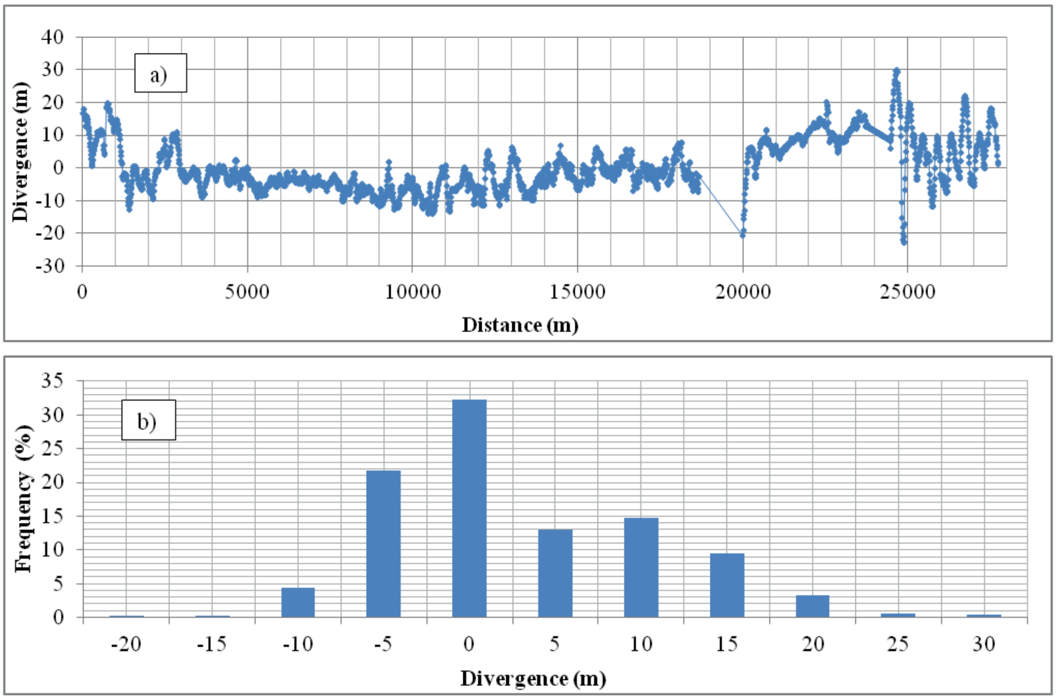

Figure 6. a) Differences between shoreline from satellite image and field survey, b) Frequency for each divergence value.

Table 2. Summary of statistical results for the first period (1978 – 1994)

Zone	Total Length (km)	Transect Lines	Average Rates of Change (m/year)			Zone Description	Net Shore Movement (NSM) (m)		
			EPR (m year^{-1})	LMS (m year^{-1})	LRR (m year^{-1})		Max.	Min.	Average
A	18.50	1822	1.50	3.08	1.95	Accretion	+ 179.8	+ 0.09	+ 75.0
B	4.50	426	0.32	-4.98	1.98	Accretion	+ 83.7	+0.09	+ 32.70
C	3.20	282	-1.35	-0.68	-1.20	Erosion	- 74.36	-1.84	- 36.27
Total	26.20	2530							

Komar (1993) used 65 beach profiles along 240 km of the Delta coastline that has been obtained annually from 1971 to 1990. Profile lines were perpendicular to shoreline that were spaced by 0.5 to 10 km and extend to a water depth of 6 m.

Table 3 shows a comparison between estimated rates from DSAS and computed rates from the field survey profiles at certain locations. It seems that, both EPR and LRR methods give better results (Figure 9), with correlation coefficients of 0.60 and 0.58, respectively. Although LRR had lower correlation coefficient than EPR, it will be used in calculations as it uses all data points in the rate calculation to reduce the influence of spurious data counter to EPR method.

LRR results show that zone A includes an accretion area representing 65 % of the total zone and characterized by a variable change rate from 0 to 9.22 m year^{-1} with an average annual rate of 3.44 m year^{-1}, and another erosion area representing 35 % of the total zone with change rate ranging between 0 and -4.72 m year^{-1} and an average annual rate of -0.90 m year^{-1} (Figure 10). However, Zone A can be considered generally an accretion zone since shoreline stepped forward by 75.0 m in average and an average annual rate of change of +1.95 m year^{-1}. Zone B is also considered an accretion zone (with a small erosion zone corresponding to 18% of zone B) with a maximum accretion rate of 6 m year^{-1} and an average rate of change of +1.98 m year^{-1}. Zone B was used as a dumping

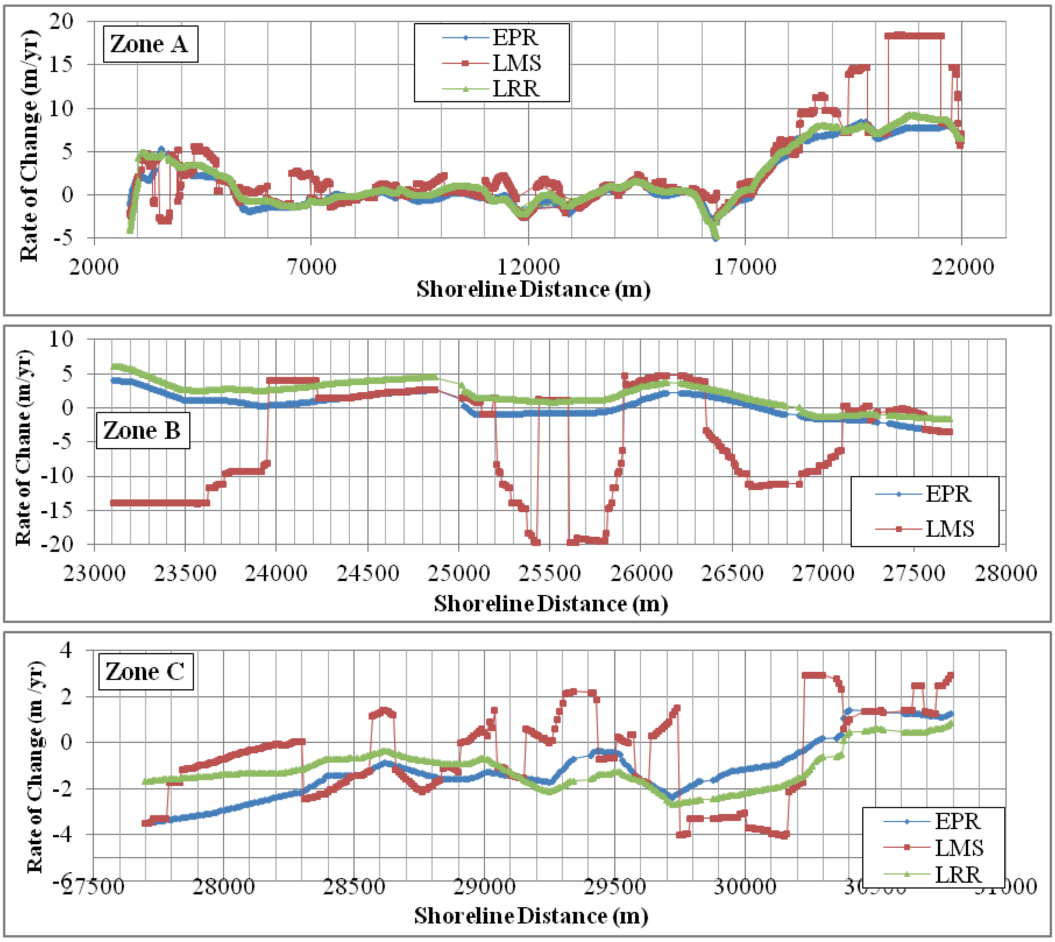

Figure 7. The rates of shoreline change using different methods for the first period (1978 – 1994).

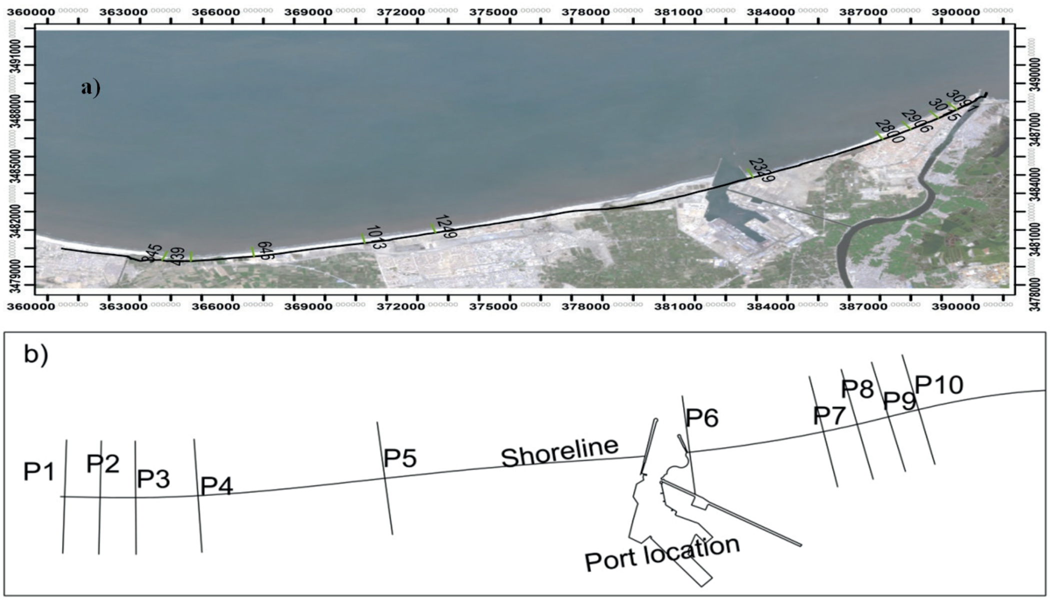

Figure 8. a) Location of transects used in the statistical analysis in the present study, b) Profile numbers and location of survey profiles by Frihy and Komar, (1993).

Table 3. The comparison between DSAS results and field survey results.

Present Study			The annual rate of shoreline change (m/year)					Frihy and Komar (1993)	
Transect ID	Location		No. Profile Location						
	East (UTM)	North (UTM)	DSAS Results				Ground Survey	East (UTM)	North (UTM)
			EPR	LMS		LRR			
345	363795.2	3479581	4.18	4.29	4.33	5.8	P1	363743.1	3479671
439	364650.3	3479545	2.22	5.19	3.46	3.2	P2	364662.2	3479649
646	366662.2	3479801	-1.36	-1.11	-1.26	2.2	P3	366687.4	3479860
1013	370241.8	3480528	-0.02	0.35	0.77	0.9	P4	370207.9	3480584
1249	372513.1	3481085	-0.79	1.24	0.02	-1.2	P5	372571.3	3481104
2329	382790.4	3484094	3.18	-13.92	4.84	1.2	P6	382762.1	3484184
2800	386972.5	3486162	-2.97	-0.75	-1.4	0.3	P7	387019.2	3486278
2906	387863.6	3486702	-1.35	-1.07	-0.99	-0.1	P8	387937.8	3486786
3015	388753.3	3487301	-0.86	-4.02	-1.89	-0.2	P9	388812.6	3487380
3091	389341.4	3487764	1.88	1.88	1.8	-0.6	P10	389493.4	3487997
Correlation coefficient			0.60	0.30	0.58				

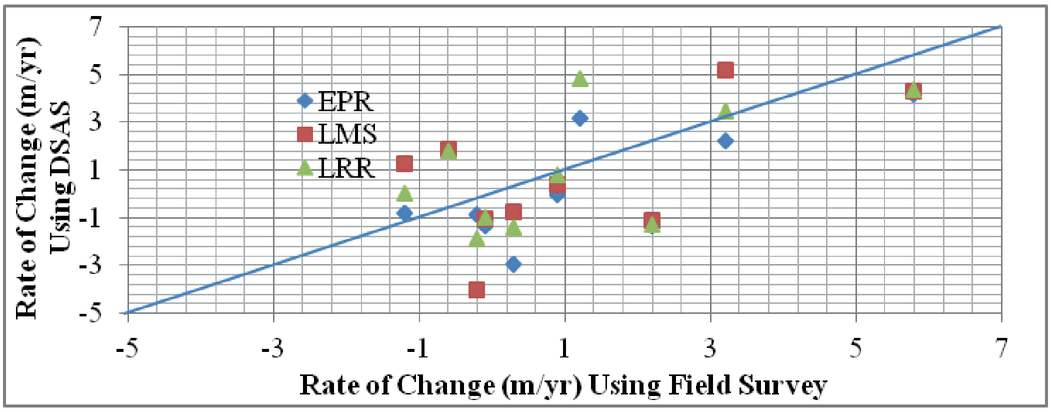

Figure 9. Validation of statistical results of the present study with ground survey results by Frihy and Komar (1993).

area of the dredged material from the navigation channel. So, the accretion rate of this zone after the harbour construction is related to the nourishment with the dredged material. The selection of this site to dump the dredged materials was not the best solution because the southwesterly reverse currents redistributed the dredged material and carried it back to the navigation channel. So, authorities changed the dumping site and decided to use the dredged material in recovering eroded beaches. On the contrary, zone C suffered from erosion that reached -2.7 m year^{-1} with an average rate of -1.20 m year^{-1} and the shoreline retreated with -74 m in some locations and -36.2 m on average. So, Egyptian authorities adopted a shore protection project composed of detached breakwaters in zone C to protect the valuable shoreline of Ras El-Bar.

4.2 Second period (1995 – 2018)

According to DSAS results, zone A followed the same trend in the second interval and 70 % of the area was under accretion and with change rates varying from 0 to 11.88 m year^{-1} with an average annual rate of 3.78 m year^{-1}, where 30 % of the zone has erosion with variable rate that ranging from 0 to -5.3 m year^{-1} and an average rate of -1.64 m year^{-1}. So, zone A is still considered generally an accretion zone with an average annual rate of change of +2.13 m year^{-1} and the shoreline had

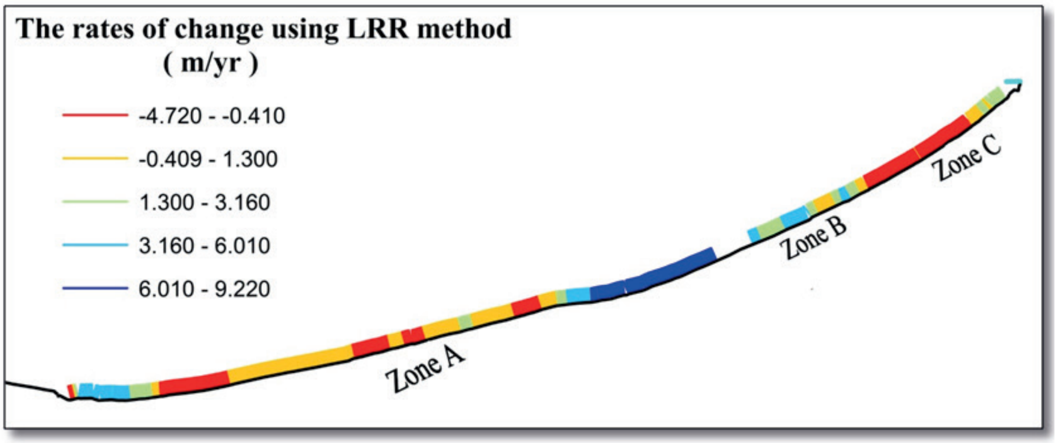

Figure 10. The rates of shoreline change using LRR method for the First period (1978 – 1994).

Table 4. Summary of statistical results for the second period (1995 – 2018).

Zone	Total Length (km)	Transect Lines	Average Rate of Change (m year^{-1})			Zone Description	Net Shore Movement NSM (m)		
			EPR (m/ year)	LMS (m/ year)	LRR (m/ year)		Max.	Min.	Average
A	18.50	1817	2.67	1.12	2.13	Accretion	+ 293.80	+ 0.06	+ 85.70
B	4.50	413	-4.26	-3.96	-4.30	Erosion	- 143.56	- 11.10	- 98.40
C	3.20	299	2.26	-0.17	1.81	Accretion	+ 106.8	+ 0.16	+ 56.30
Total	26.20	2529							

an average progressive of 85.70 m. On the other side, zone B was converted to erosion zone with a change rate ranging from 0 to -6.6 m year^{-1} and an average of -4.30 m year^{-1}. In addition, shoreline retreated 143.5 m in some points with an average regressive of -98.40 m. On the other hand, in zone C as a result of the implemented detached breakwaters, the erosion rate changed to an accretion rate varying from 0 to 3.97 m year^{-1} with an average value of + 1.81 m year^{-1} and the shoreline stepped forward 106.8 m in some points and 56.30 m in average. Table 4 summarizes the results for each zone.

Figure 11 also shows the variable rates of shoreline change along the study area using LRR method for each zone.

From the data analyzed, it can be concluded that the construction of the Damietta harbour significantly impacted the adjacent shoreline, increasing the accretion rate in zone A with 10 %, Figure 12. The western breakwater of the harbour blocks the littoral drift from the west and accumulated sand generating an accretion zone on the western side. However, Zone B was converted from an accretion zone to an erosion zone with a high erosion rate as the western breakwater of the harbour and the navigation channel block the littoral drift moving toward the east. On the other hand, the implemented detached breakwaters at Zone C showed a good performance since the erosion problem was solved and the shoreline was improved with an average rate of 1.81 m year^{-1}.

Therefore, zone B is considered to be one of the main hazard areas along the Northeastern shoreline of Nile Delta and needs a comprehensive and intensive study to determine the proper solution for this area.

4.3 Analysis of hazard area

Damietta harbour has a deep effect on zone B leading to high erosion rate. The area was analyzed with DSAS to measure the change in the total period from 1973 to 2018, Figure 13. The average rate of change varies from -4.64 m year^{-1} to -1.16 m year^{-1} with an average value of -3.39 m year^{-1}. Also, NSM shows that the shoreline retreated from 1978 to 2018 by 156.3 m in some points and 92 m on average, Figure 14.

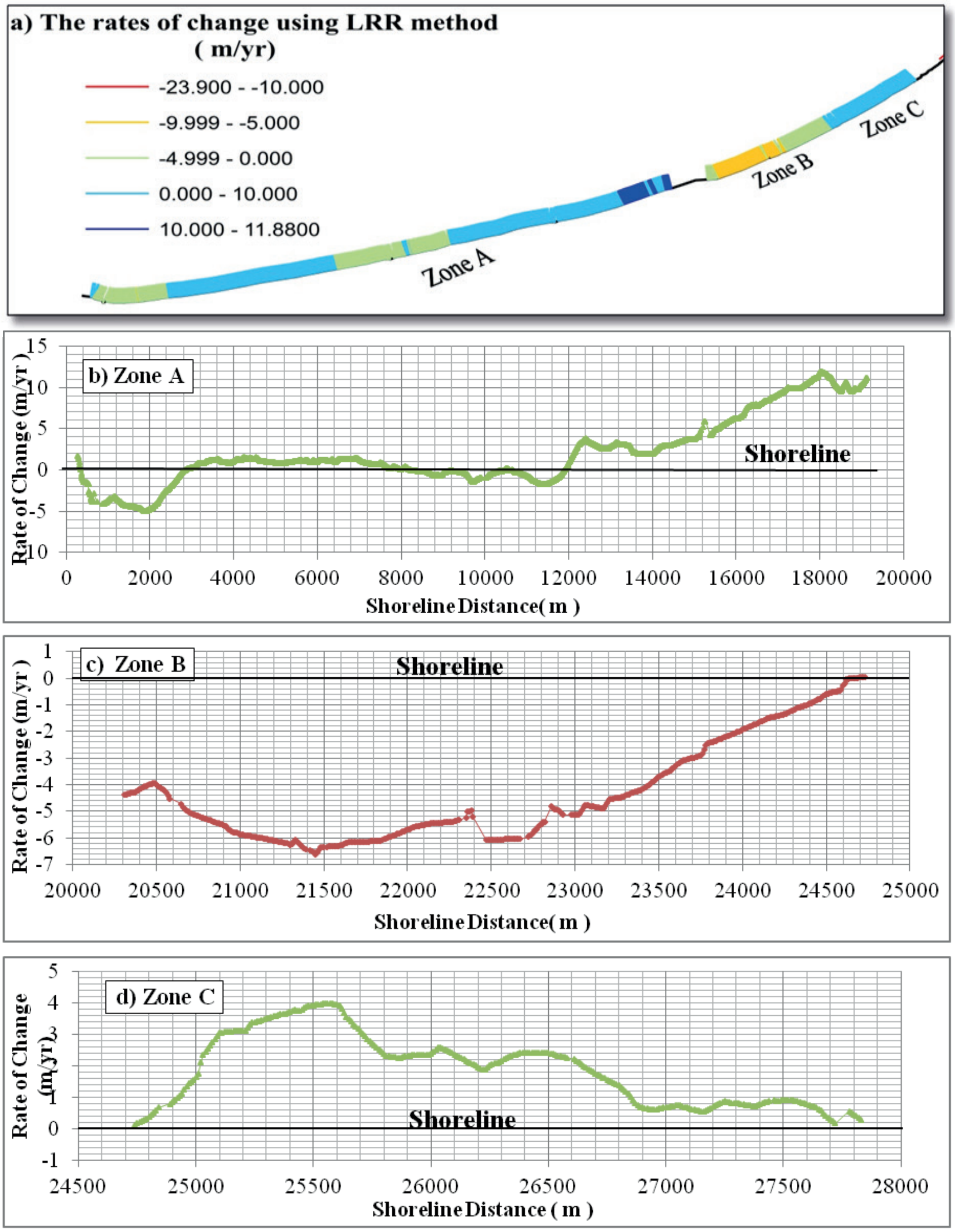

Figure 11. a) Statistical results using LRR method for the study area during (1995 – 2018), b) Rates of shoreline change for Zone A, c) Rates of shoreline change for Zone B, d) Rates of shoreline change for Zone C.

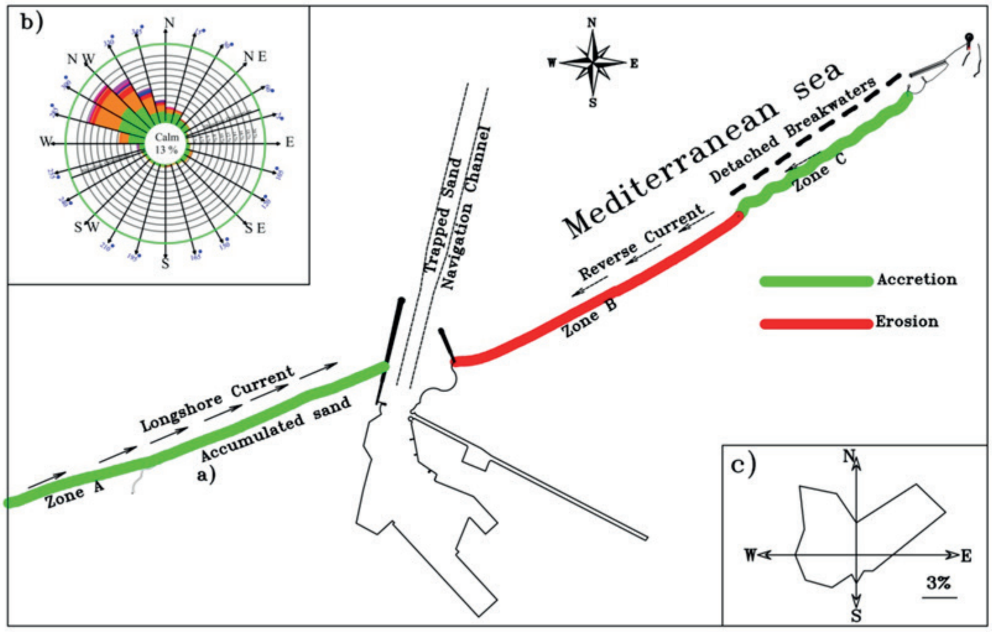

Figure 12. a) Erosion/accretion pattern of the study area, b) Wave rose at Damietta harbour; c) Current rose recorded at Damietta harbour.

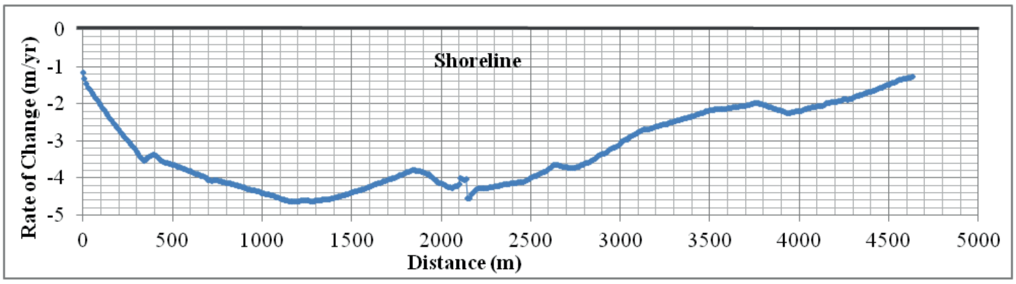

Figure 13. Average rates of shoreline change at zone B.

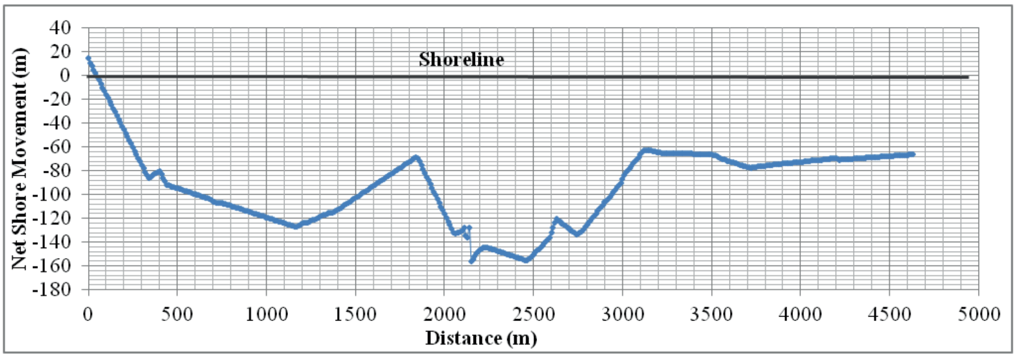

Figure 14. Net shore movement at zone B.

The erosion rate of this zone was a result of blockage of sediment discharge from both the western side and the eastern side. The western breakwater and the navigation channel trapped sediments from Burullus, and detached breakwaters at Ras El-Bar beach settle down any incoming sediment from Damietta promontory. So, zone B is considered one of the main hazard areas along the Northeastern shoreline of Nile delta and needs a sustainable solution. Sand nourishment is the solution current used in the eastern coast (Figure 15) to restore eroded area of the beach and to keep it available for recreation uses. However, it is a conventional solution that has rising periodic cost. Satellite image of 2019 shows four small groins that were constructed by Egyptian authorities in the first part of zone B. However, groins will not be effective without continuous sand nourishment, since sediment flux is trapped in both sides.

4.4 Prediction of future shoreline evolution

Shoreline prediction was carried out for zone B to clarify its expected position in the future if there no sustainable solution for this area is applied. The prediction accuracy of shoreline position is based on the historical processes that could be determined by satellite images. Numerous methods have been adopted for future prediction of shoreline position or sea-level rise like non-linear mathematical models, and the simplest and useful ones are the End Point Rate (EPR) and the Linear Regression Rate (LRR) models, (Li *et. al.*, 2001). So, in this case the EPR model has been used to predict the future shoreline positions. The position of the future shoreline for a certain date is predicted using the rate of shoreline movement, the time interval between predicted and observed shoreline which can be expressed as:

$$Y_{pre} = r_{EPR} X_{pre} + b \tag{1}$$

where Y_{pre} represents the predicted distance from the baseline in meters, X_{pre} the time interval between predicted and observed shoreline, r_{EPR} the rate of change given from DSAS for each transect and b the Y-intercept given by,

$$r_{EPR} = (Y_n - y_0) / (X_n - X_0) \tag{2}$$

$$b = Y_n - m_{EPR} \times X_n = Y_0 - m_{EPR} \times X_0 \tag{3}$$

where Y_n represents the predicted distance from the baseline in meters at the last shoreline date X_n and Y_0 the initial distance from the baseline in meters at the first shoreline date X_0.

To validate this technique, shorelines of 1995 and 2010 and corresponding r_{EPR} were used to predict the shoreline of 2013. Root mean square error (RMSE) was calculated to evaluate the reliability of this method. The delineation between predicted values and actual values varied from 33.0 m to 0.02 with RMSE of 12.54 m and relative error of 0.63 m year^{-1}, that is considered a reasonable value for such a large shoreline.

Figure 15. Sand nourishment in the eastern coast of Damietta harbour (April 2019).

To enhance prediction accuracy, the rate of change should be corrected as,

$$r_{EPR\ corrected} = r_{EPR} + (\text{error} / \text{time interval}) \qquad (4)$$

Shorelines of 2030, 2040, 2050 and 2060 were predicted after correction and shown in figure 16. From 2018 to 2060, the shoreline was estimated to retreat approximately 390.0 m in some points with an average regression of 280.0 m, which is considered a significant loss of area.

5. SUMMARY AND CONCLUSIONS

Remote sensing techniques were used to evaluate the impact of the Damietta harbour and its deep navigation channel on adjacent shorelines. DSAS was used to determine the accurate rate of change and predict the future shoreline evolution. The Damietta harbor and navigation channel have a deep impact on neighboring beaches as the navigation channel acts as a sediment trap that interrupts the long-shore transported sediment. The western breakwater prevents sediment movement partially from west to east, so the accretion zone is created on the western side of the harbour with accretion rate of 2.13 m year^{-1}. Hence, shoreline monitoring deduced that the eastern shoreline had an average erosion rate of -3.1 m year^{-1} and it is considered a fatal hazard area along Northeastern shoreline of Nile Delta. Shoreline retreated from 1978 to 2018 by 156.3 m in some points and 92 m on average. Shoreline prediction shows that shoreline will retreat by 390.0 m in some points with an average regression of 280.0 m in 2060 if no sustainable solution for this area is applied.

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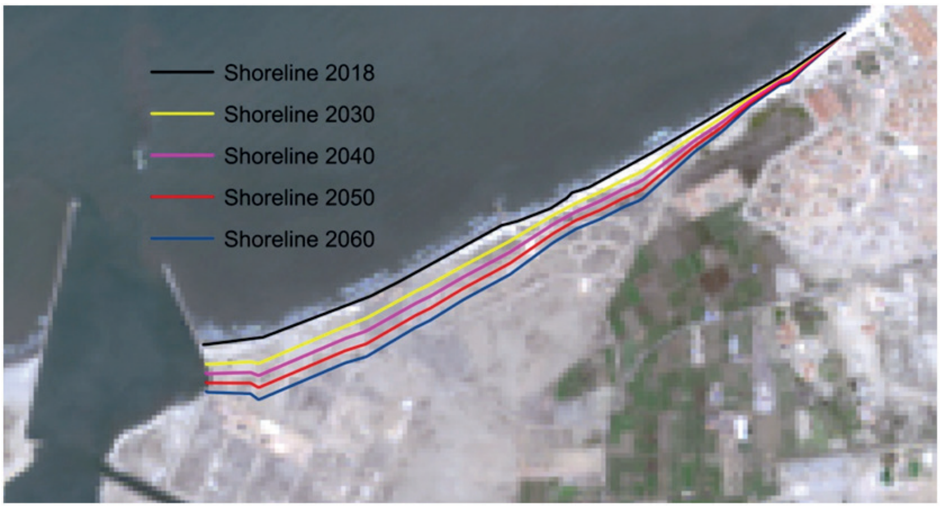

Figure 16. The predicted shoreline evolution of zone B.

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