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Results of the problem based science analysis

3.2.1) The Ria de Aveiro Lagoon, Portugal



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Integrated water resources and coastal zone management in European lagoons in the context of climate change



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CO Confidential, only for members of the consortium (including the Commission Services)

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List of abbreviations

AMD	Acute Marsh Dieback
AMBI	AZTI's Marine Biotic Index
APA	Portuguese Environmental Agency
ARHCentro	Administração da Região Hidrográfica do Centro I.P.
DAISIE	Delivering Alien Invasive Species Inventory for Europe
DPSIR	Drivers-Pressures-State-Impacts-Response
ICNB	Portuguese Institute for Nature Conservation and Biodiversity
ICNF	National Institute for Environment and Forest Conservation
INAG	Portuguese Water Institute
IPCC	Intergovernmental Panel on Climate Change
MAMAOT	Ministério da Agricultura, Mar, Ambiente e Ordenamento do Território
TBT	Tributyltin
WFD	Water Framework Directive
WORMS	World Register of Marine Species
WWTP	WasteWater Treatment Plant

Summary

This report constitutes a knowledge analysis based on existing literature regarding the Biology and Ecology of Ria de Aveiro lagoon.

The Ria de Aveiro is located on the Portuguese Atlantic coast, comprising a high number of habitats which support high biological diversity. Ria key benthic habitats are salt marshes, seagrass meadows and mudflats. Salt marshes and seagrass meadows have crucial functions on stabilizing the sediment, protecting intertidal areas from tidal action and erosion. All together, these habitats provide shelter and resources for ecological and economical valuable species of invertebrates and fishes.

Updated lists of Ria Flora and Fauna (macroinvertebrates and mobile megafauna and fish) are provided. The presence of alien/invasive species is also reported.

Regarding the application of WFD, Ria is subdivided in five transitional water bodies (4 natural and 1 heavily modified): Two classified as Good, one Moderate and one Poor, whilst the potential ecological status of the heavily modified water body is Moderate.

The report also includes a description of the main fisheries, and finalizes with the identification of some open questions.

1 General introduction

The main objective of this report is to systematize the knowledge base regarding the Biology and Ecology of Ria de Aveiro lagoon.

Chapter one starts with a brief description of the system, highlighting the importance of benthic communities in the functioning of the lagoon and the support of key ecosystem processes and services. The reader will also get an overview of the major environmental changes that took place in Ria in the last decades, including the presence of invasive/alien species already recorded. Chapter one finalizes with an overview regarding the Ria water quality status based on publications from the last ten years. Chapter two focuses on the description of Ria main habitats and their ecological importance. This chapter includes a short description of each key habitat, including a list of the main macrophyte, macroinvertebrate and mobile megafauna species recorded in the lagoon, and finalizes with an overview of these habitats in the context of climate change.

Chapter three summarizes the published results regarding benthic biological diversity and benthic ecological quality, and a map illustrating the qualitative distribution of the benthic macrofauna biological diversity is provided. In Chapter four readers will find a detailed list of the main fish species that can be found (or were found in the past) in Ria de Aveiro lagoon, based on literature review, followed by Chapter five where a description of the main fisheries is provided. The report finalizes with the identification of some open questions (Chapter 6).

1.1 Ria de Aveiro

The Ria de Aveiro (40°38'N, 08°45'W) is a shallow coastal lagoon connected to the Atlantic Ocean through a single inlet (Figure 1.1), and is located in the central coastal zone of Portugal integrating the Vouga river catchment area (<http://www.arhcentro.pt/>). For a more detailed description of the catchment area please see LAGOONS project webpage (<http://lagoons.web.ua.pt/> - LAGOONS 2012, Report D2.1b).

The lagoon is approximately 45 km long (NNE-SSW), 10 km wide and, during spring tide, covers an area of approximately 83 km² and 66 km² of wetland at high and low water, respectively (Dias *et al.*, 2000). The bathymetry of the Ria de Aveiro consists of four main channels with several branches forming islands, inner basins and mudflats (Figure 1.2). The Mira and Ílhavo channels are narrow and shallow; the S. Jacinto - Ovar channel is wider and

deeper in its southern part, forming secondary narrow and shallow channels and basins northwards. The Espinheiro channel is characterised by secondary channels, islands, marshes and basins (Figure 1.2). Other smaller rivers also discharge into the lagoon, namely the river Boco in the Ílhavo channel; the Cáster river in the Ovar channel, and the Mira river in the Mira channel (<http://lagoons.web.ua.pt/> - LAGOONS Report D2.1b). The Vouga river is the most important, accounting for 2/3 of the total freshwater input (Moreira *et al.*, 1993). Due to the combined effects of the freshwater discharge and tidal propagation, the central area of the Ria de Aveiro exhibits a longitudinal salinity gradient from about 0 in the upper reaches of the Espinheiro channel to about 36 at the bar entrance (e.g. Vaz and Dias, 2008). The average depth of the lagoon relative to chart datum is about 1 m, except in the navigation channels where dredging operations are frequently carried out (<http://lagoons.web.ua.pt/> - LAGOONS 2012, Report D6.1).

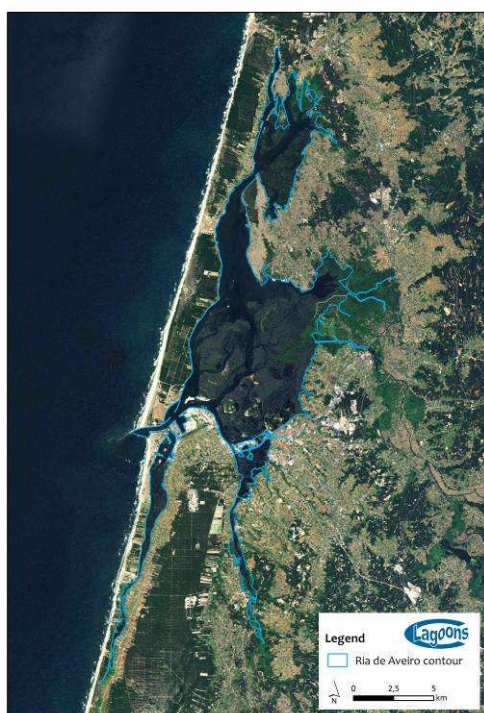


Figure 1.1 – Aerial view of Ria de Aveiro with its contour highlighted in blue (source: map from Google Earth; contour from WFD).

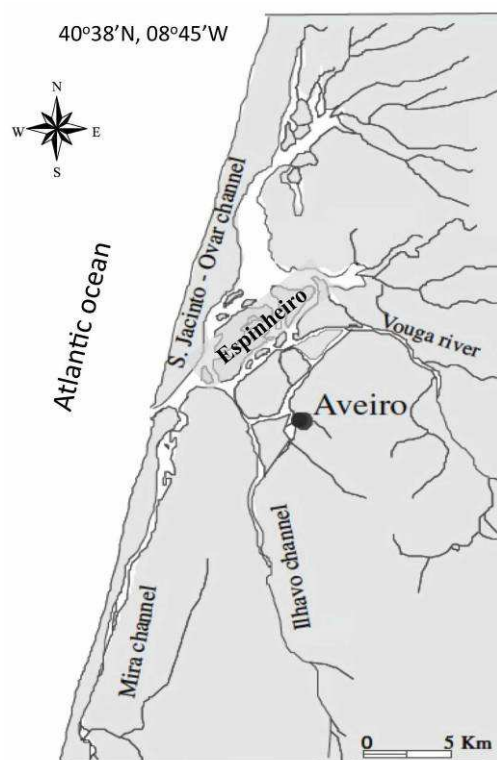


Figure 1.2 – The main channels in the Ria de Aveiro coastal lagoon.

1.2 The benthic communities

Benthic communities at Ria de Aveiro lagoon play a key role in the environmental health and biodiversity, contributing to the well-being of the surrounding populations. Seagrass meadows and salt marshes, which are among the most productive habitats in coastal areas (McLusky and Elliott, 2004), have important roles in the functioning of the lagoon, namely through the ecological processes occurring within these transitional areas (e.g. estuaries and coastal lagoons). Salt marshes and seagrass meadows have crucial functions on stabilizing the sediment, protecting intertidal areas from tidal action and erosion, as well as providing habitat for ecological and economical valuable species of invertebrates and fishes (Orth *et al.*, 2006; Gedan *et al.*, 2009; Lillebø *et al.*, 2011). Moreover, these habitats are crucial in primary productivity, nutrients' sequestration and cycling (Sousa *et al.*, 2008, 2010, 2012; Silva *et al.*, 2009; Lillebø *et al.*, 2011), metals' sequestration and cycling (e.g. Válega *et al.*, 2008; Pereira *et al.*, 2009; Lillebø *et al.*, 2010a; Marques *et al.*, 2011; Anastácio *et al.*, 2013), and still can mediate organic pollutants degradation and bioavailability (e.g. petroleum hydrocarbons, pesticides and surfactants) (Almeida *et al.*, 2008, 2009a, 2009b; Martins *et al.*, 2008; Ribeiro *et al.*, 2013). The mentioned ecological processes support the ecological functions and services provided (provisional, regulation and maintenance) by Ria de Aveiro benthic communities.

Considering Ria recent past, namely the last three decades, the major changes holds up to ecohydrology of the system.

In the 1950s, the inlet channel was improved by construction of breakwaters and deepened by dredging as far upstream as the port of Aveiro (Silva *et al.*, 2004). During the 1990s, most of the dredging operations were carried out in the main navigation channel but, during the 1990s, large quantities of sediment ($> 2 \times 10^6 \text{ m}^3$; Administração do Porto de Aveiro) were also removed from the inner channels (Silva *et al.*, 2004). Changes in the system's hydrodynamics have altered the tidal prism and increased the water velocity (Picado *et al.*, 2010), resulting in the loss of subtidal seagrass meadows and reducing the intertidal meadows extension and biodiversity (Silva *et al.*, 2004). Deepening of the major channels due to dredging activities and the construction of inlet piers (Silva and Duck, 2001; Silva *et al.*, 2004, 2009) led to the increase in tidal water velocity and turbidity, sediment resuspension, loss of fine sediments and nutrients (Silva *et al.*, 2004), which ended in the decline in intertidal area and lastly in the seagrass meadows decline (Silva *et al.*, 2009). Currently, seagrass meadows of this lagoon are

restricted to intertidal areas and *Zostera noltii** is the only expressively present seagrass species (Silva *et al.*, 2009; Azevedo *et al.*, 2013; Cunha *et al.*, 2013).

*According to WORMS – World Register of marine Species (<http://www.marinespecies.org/index.php>, Accessed [6/13/2013]) the name is *Zostera noltei* Hornemann, 1832, as originally described by the author (<http://images.algaebase.org/pdf/562DF4FA11a772CADEOyVIDFDDA3/48367.pdf> Accessed [6/13/2013]). However, throughout this report we will keep the most commonly used name according to the literature cited).

Until the 1960's, the traditional activity of harvesting “moliço” (a mixture of seagrasses and macroalgae, including *Zostera noltii*, *Zostera* (*Zostera*) *marina* Linnaeus, 1753, *Ulva* Linnaeus, 1753 and *Gracilaria* Greville, 1830) and its use as fertilizer for agricultural fields had a great impact in the surrounding human-populations (Santos and Duarte, 1991). However, after a remarkable decrease in the subtidal population and due to the increased use of chemical fertilizers in agriculture, “moliço” lost its economical value and commercial harvesting of seagrass was totally ceased (Silva *et al.*, 2004). To sum up, since the 1980's, the combined pressures in Ria de Aveiro lagoon led to the decrease of vertical distribution and species diversity of these meadows (detailed data in section 2). In addition, seagrass meadows coverage greatly decreased over the last three decades: namely, in 1984, *Z. noltii* had a cover distribution of 8 km² in the Ovar channel, but only 1 km² was recorded by 2003 (Silva *et al.*, 2004). Accordingly, macroalgae that populated subtidal areas in the past (e.g. *Chaetomorpha ligustica* (Kützing) Kützing, 1849, previously described as *Lola lubrica* (Setchell & N.L.Gardner) A.Hamel & G.Hamel, 1929) had disappeared. In Azevedo *et al.* (2013), authors applied the generic DPSIR (Drivers-Pressures-State-Impacts-Response) framework to the Ria Zosteraceae seagrass communities, and results are shown in table 1.1. Regarding salt marshes, there are evidences of the decrease of the salt marsh area in the Baixo Vouga Lagunar due to the increase of the tidal prism and consequently the increase of the saltwater flooding period, also putting the unique and typical landscape, “bocage”, under pressure (Project F:ACTS! webpage - <http://www.factsproject.eu/>). Furthermore, changes of the lagoon hydrodynamics and the abandonment of salt works, one of ancient traditional activities, have led to the collapse of the saltpan walls (Picado *et al.*, 2010) and consequently changing the landscape.

Table 1.1 – Application of DSPIR framework to the *Zosteraceae* seagrass communities of Ria de Aveiro (data from Azevedo *et al.*, 2013).

The framework	The evidence	References
Driving Forces	Harbour activities (commercial and fishing)	(Duck and Silva, 2012)
Pressures	Dredging activities;	(Silva and Duck, 2001)
	Breakwater construction	(Silva and Duck, 2001; Silva <i>et al.</i> , 2005)
	Fishing/bait digging	(Silva <i>et al.</i> , 2005).
State	System hydrodynamics changed	(Silva and Duck, 2001; Silva <i>et al.</i> , 2004, 2007, 2009; Araújo <i>et al.</i> , 2008)
	Channel depth increased	(Silva and Duck, 2001; Silva <i>et al.</i> , 2004, 2009)
	Nutrient cycling modified	(Silva <i>et al.</i> , 2009)
	Modifications in sediment dynamics	(Silva and Duck, 2001; Silva <i>et al.</i> , 2004, 2005)
	Seagrass decline	(Silva <i>et al.</i> , 2004, 2005, 2009; Cunha <i>et al.</i> , 2013)
	Light penetration	(Silva <i>et al.</i> , 2004; 2009)
	Alterations of water physicochemical characteristics	(Silva <i>et al.</i> , 2004)
Impacts	Decrease in seagrasses species diversity	(Silva <i>et al.</i> , 2004, 2005, 2009; Cunha <i>et al.</i> , 2013)
	Habitat loss and fragmentation	(Silva <i>et al.</i> , 2004, 2005, 2009; Cunha <i>et al.</i> , 2013)
	Decrease protection against erosion	(Silva <i>et al.</i> , 2009)
	Decrease settling of fine sediments	(Silva <i>et al.</i> , 2007, 2009)
	Increased water current	(Silva <i>et al.</i> , 2004, 2007, 2009)
	Decrease nutrient retention	(Silva <i>et al.</i> , 2009)
	Decrease of other associated ecosystem services, processes and functions	(Barbier <i>et al.</i> , 2011)
Responses	<p>Complementary studies to evaluate the resilience of Ria de Aveiro seagrass meadows to biogeochemical and physical stressors;</p> <p>Development of a decision-support tool for management strategies (e.g. biological, water quality and hydrodynamic models of intertidal seagrass communities' processes in Ria de Aveiro, including future climate change scenarios);</p> <p>Contribution to the implementation of the WFD, i.e., to the knowledge towards a good ecological status of Ria de Aveiro</p>	(Azevedo <i>et al.</i> , 2013)

1.3 Presence of invasive/alien species

The presence of invasive/alien species has also been recorded in Ria de Aveiro.

(All species names were verified and updated according to WORMS – World Register of marine Species (<http://www.marinespecies.org/index.php>, Accessed [6/10/2013])).

Relevant examples of macroinvertebrates are:

Venerupis philippinarum (A. Adams & Reeve, 1850) – It is native distribution is South China Sea, East China Sea, Yellow Sea, Sea of Japan, Sea of Okhotsk and southern Kuril Islands (European Network on Invasive Alien Species – NOBANIS - <http://www.nobanis.org/> Accessed [6/10/2013]). The status in Portugal is Alien/Established (DAISIE – Delivering Alien Invasive Species Inventory for Europe (<http://www.europe-aliens.org/> Accessed [6/10/2013])). *V. philippinarum* (previously known as *Ruditapes philippinarum* (Adams & Reeve, 1850)) was recently introduced in Ria where it lives in sympatry with the native species *Venerupis decussata* (Linnaeus, 1758) (previously known as *Ruditapes decussatus* (Linnaeus, 1758)) (Figueira and Freitas, *in press*).

Mya arenaria Linnaeus, 1758 – It is native to the North American east coast from Labrador, Canada to Cape Hatteras (Global Invasive database (<http://www.issg.org/> Accessed [6/10/2013])). The status in Portugal is Cryptogenic/Established (DAISIE – Delivering Alien Invasive Species Inventory for Europe (<http://www.europe-aliens.org/> Accessed [6/10/2013])). In Ria de Aveiro *M. arenaria* has a low density and a restricted distribution, but has an established reproductive population (Conde *et al.*, 2012).

Corbicula fluminea Müller, 1774 – the genus *Corbicula* lives in temperate to tropical southern Asia west to the eastern Mediterranean; Africa, except in the Sahara desert; and southeast Asian islands south into central and eastern Australia (Morton 1986 in U.S. Geological Survey. [2013]. Nonindigenous Aquatic Species Database. Gainesville, Florida. Accessed [6/10/2013]). The status in Portugal is Alien/Established (DAISIE – Delivering Alien Invasive Species Inventory for Europe (<http://www.europe-aliens.org/> Accessed [6/10/2013])). In Ria de Aveiro *C. fluminea* has been recorded in Casal de São Tomé channel (municipality of Mira) that runs to the Ria de Aveiro (Gabriel, 2011), in Vouga River (Reis 2006, in Morais *et al.*, 2009) and in the most inner channel areas of Ria, except Ílhavo and Ovar channels (Rodrigues *et al.*, 2011).

Relevant examples of macrophytes are:

Spartina versicolor E. Fabre – the origin, nomenclature and systematics of this species has been controverse and is not totally clear. It was firstly described in the French Mediterranean coast (Fabre 1849, in Prieto *et al.*, 2011), then in the coast of Italy and on the Portuguese coast (Prieto *et al.*, 2011 and references therein). Thus, while some authors consider it as a Mediterranean native species, others suggest that its native distribution is the Atlantic European coasts and others still suggest that this species is native at the North American Atlantic coast exhibiting invasive behaviour on the European Atlantic coast (SanLéon *et al.*, 1999; Prieto *et al.*, 2011 and references therein). This species has also been reported as *Spartina patens* (Aiton) Muhlenberg, and according to Prieto *et al.* (2011) it should be considered the same species as *Spartina versicolor*. Flora Europaea (<http://rbg-web2.rbge.org.uk/FE/fe.html> Accessed [6/13/2013]) classifies *Spartina versicolor* E. Fabre, *Spartina patens* auct., non (Aiton) Muhl., *Spartina durieui* Parl., *Spartina juncea* auct., non (Michx.) Willd, as synonyms. In Ria de Aveiro, *S. versicolor* has been considered an invasive species by the National Institute for Environment and Forest Conservation – Instituto da Conservação da Natureza e das Florestas, I. P. (ICNF) - (PSRN 2000; IDAD, 2008; ICNF, 2012). *S. versicolor* was not mentioned in a monitoring study performed by Crespo (2003 in IDAD, 2008) but was later identified in the high marshes of Ria (IDAD, 2008; ICNF, 2012).

1.4 Water quality status

Regarding the Ria water quality status the following summarizes publications from the last ten years on: i) the overall eutrophic condition of Ria (Ferreira *et al.*, 2003; Lopes *et al.*, 2007); ii) the characterization of Ria habitats (AMBIECO, 2011); iii) the classification of Ria water masses following the Water Framework Directive (MAMAOT/ARHCentro, 2012); iv) the historical contamination of the system (Castro *et al.*, 2006; Sousa *et al.*, 2007; Pereira *et al.*, 2009).

One decade ago the former Portuguese Water Institute (INAG), which is now Part of the Portuguese Environmental Agency (APA), promoted a study aiming the Identification of sensitive areas and vulnerable zones in transitional and coastal Portuguese systems. The resulting report concluded that the Ria de Aveiro has a moderate degree of eutrophication and low overall human influence in comparison to other estuarine systems, and that less than 10% of nutrient inputs were coming from point sources (Ferreira *et al.*, 2003). The improvement of

the WWTP systems and the construction of a submarine outfall have reduced nutrient loads (SIMRIA webpage - <http://www.simria.pt/>), however some upstream areas still show higher concentrations of dissolved inorganic nutrients, namely nitrogen (Lopes *et al.*, 2007). As the pressure from nutrient loads were not expected to increase, to what concern eutrophication Ria was not recommended to list as a sensitive area (Directive 91/271/EEC) or vulnerable zone (Directive 91/676/EEC) (Ferreira *et al.*, 2003). More recently, i.e. in 2012 (MAMAOT/ARHCentro, 2012), an area of 23.36 Km² (including the municipalities of Aveiro, Vagos, Mira, Cantanhede and Oliveira do Bairro) and an area of 81,38 km² (including the municipalities of Estarreja-Murtosa) were classified as vulnerable zones, respectively: Zona Vulnerável Litoral Centro and Zona Vulnerável de Estarreja-Murtosa. Nevertheless, and although the Ria de Aveiro is quite urbanized and industrialized in some areas, it has been classified to be in a reasonable good state of environmental preservation. The study leading to this conclusion was done in the scope of the Polis Litoral Ria de Aveiro programme (<http://www.polisriadeaveiro.pt/>) reported by AMBIECO (2011).

In the scope of the WFD, Ria de Aveiro lagoon is divided in five transitional water bodies (Figure 1.3). These water bodies were classified as follow (MAMAOT/ARHCentro, 2012);

WB1 – A natural (unmodified) water body that includes the Mira channel and Barra - the connection to the Atlantic ocean. The water ecological status is **Good**.

WB2 – A heavily modified water body corresponding to the central area of the lagoons. The water potential ecological status is **Moderate**.

WB3 – A natural (unmodified) water body that corresponds to the Ílhavo channel. The water ecological status is **Good**.

WB4 – A natural (unmodified) water body that includes the Murtosa channel and the Laranjo basin. The water ecological status is **Moderate**.

WB5 – A natural (unmodified) water body that corresponds to the Ovar channel. The water ecological status is **Poor**.

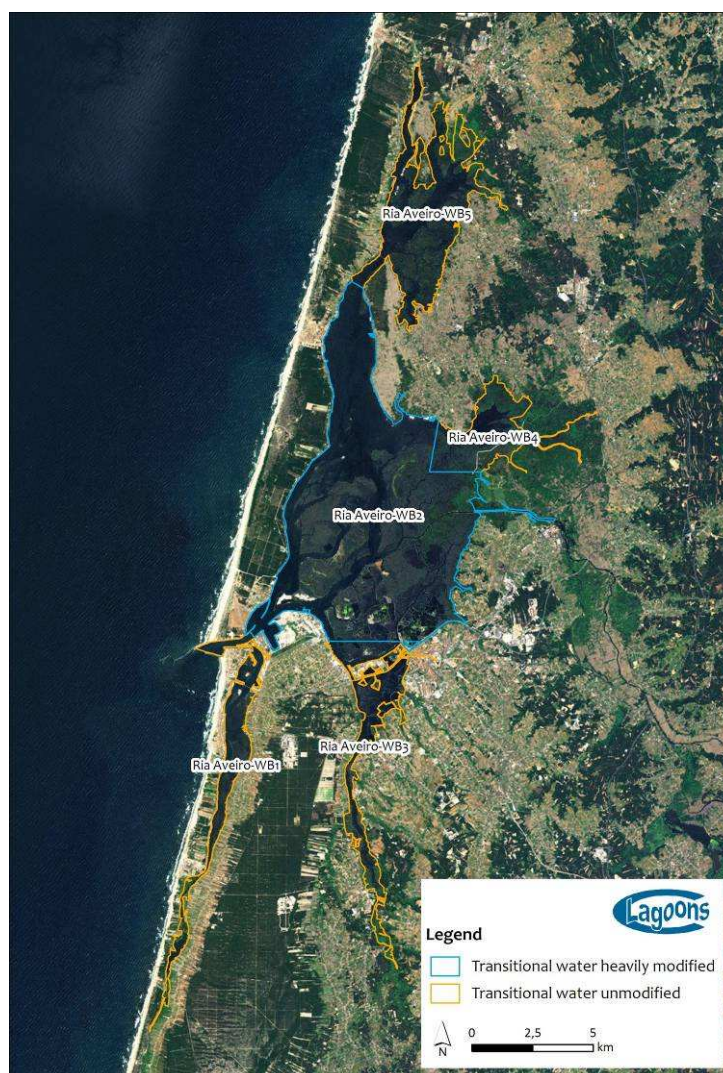


Figure 1.2 – Aerial view of Ria de Aveiro with the contour of the water masses according to WFD (source: map from Google Earth; contour from WFD).

The implementation of EU environmental policies has furthermore aided in reducing anthropogenic sources of potentially toxic elements (e.g. [Pereira et al., 2009](#)). However, there is substantial contamination in bottom sediments as a result of historical contamination. In fact, the most impacted areas are a comparatively small basin in the northern part of the estuary (Coroa basin) ([Castro et al., 2006](#)), and the Laranjo basin, a shallow area measuring 2 km², that is historically contaminated with mercury ([Pereira et al., 2009](#)). From the 1950's until 1994, the Laranjo basin received a highly contaminated effluent that was discharged from a mercury cell chlor-alkali plant located in the Estarreja industrial complex ([Pereira et al., 2009](#)). The sediment pool of mercury is still a cause of concern since it may constitute a

potential source to biota (Pereira *et al.*, 2009). Outside the Laranjo basin, Hg levels are much lower and below the European threshold concentration for fish and seafood consumption (0.5 mg kg⁻¹ for seafood and 1 mg kg⁻¹ for certain fish, EC No. 466/2001) (Pereira *et al.*, 2009). Another study (Sousa *et al.*, 2007) denoted a decrease in TBT pollution in the Ria over the last years. Changes became particularly clear after 2003 and might be associated to the EU ban of TBT.

2 Description of the main lagoon habitats and their ecological importance

2.1 Ria habitats and their classification under international conventions:

The lagoon watershed area is surrounded by a high number of habitats supporting high biological diversity (Figure 2.1). It is classified as a special area of conservation under the EU directive on the conservation of wild birds (79/ 409/EEC). Under the Berne Convention it has several species classified as protected, strictly protected or as endangered. In its northern part, between S. Jacinto and Torreira, there is a nature reserve—“Reserva Natural das Dunas de S. Jacinto”. This natural reserve (Natural Reserve of S. Jacinto Dunes) covers an area of approximately 960 ha with 210 ha corresponding to the maritime area, and has a vast diversity of habitats and species. The Decree-Law No. 47/79 of 6th March established the “Nature Reserve” status in order to create the necessary conditions for the preservation of the coastal dunes (ICNB webpage, <http://portal.icnb.pt/>). From the conservational point of view this system is considered a high priority since it is a fundamental step in the migration of aquatic birds and an ideal place for winter shelter and nesting. The whole area of the Ria de Aveiro also supports agriculture farms and activities intrinsically associated with major towns in coastal areas: port facilities, industries, aquacultures, salt production and fishing. Since the 19th century, the settled population has shaped the ecosystem by creating salt pans and drainage marshes, opening small channels for navigation, and by creating farmlands such as the smallholdings named “bocage”, thus contributing to the increase of habitat diversity and associated biodiversity (BIORIA webpage, <http://www.bioria.com>). Within the lagoon the key habitats include saltmarshes, seagrass meadows and mudflats (Figure 2.2, *seagrasses not shown*).

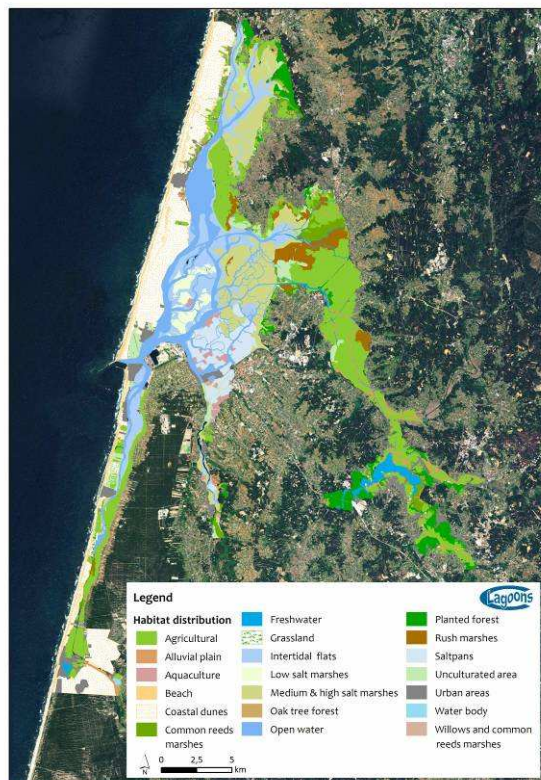


Figure 2.1 – Aerial view of Ria de Aveiro habitats distribution (source: map from Google Earth; data from AMBIECO report for Polis Littoral Ria de Aveiro, 2011).



Figure 2.2 – Aerial view of the lagoon main benthic habitats: intertidal mudflats; low-, medium- and high salt marshes (seagrasses not shown) (source: map from Google Earth; data from AMBIECO report for Polis Littoral Ria de Aveiro, 2011).

2.2 Ria key benthic habitats:

Salt marshes are located in transitional areas between the land and the estuarine/brackish water. These habitats are characterized by a typical zonation and spatial distribution of the halophytes species, depending on the marsh's topography. Consequently, tidal submersion (frequency and duration), physical and chemical characteristics of the sediment, and the interspecific competition depend on the halophyte's location (Lefeuvre *et al.*, 2003). Accordingly, halophytes colonizing low salt marsh are adapted and tolerant to different physical and chemical parameters, when compared to the plant species colonizing mid- and high marshes. Other parameters, such as competition and facilitation among salt marsh plants, as well as grazing, are also critically important in controlling community structure (Barbier *et al.*, 2011). Furthermore, all salt marsh plants have important ecosystem functions, namely nutrient cycling, high primary productivity, protection against tidal action and coastal erosion

(due to their contribution to the sedimentation process and detritus retention), important habitats and nursery areas for different estuarine fauna (e.g. fishes) (Sousa *et al.*, 2008, 2010, 2012; Gedan *et al.*, 2009; Lillebø *et al.*, 2010; Barbier *et al.*, 2011; Deegan *et al.*, 2012).

As mentioned above, salt marshes are characterized by a zonation pattern following the marsh topography and the salinity gradient. Accordingly, Ria de Aveiro low salt marshes are colonized by pioneer vegetation such as *Spartina maritima* (Curt.) Fernald and *Salicornia ramosissima* J.Woods (IDAD, 2008; AMBIECO, 2011; <http://www.biorede.pt/>), able to support long inundation and anaerobic periods, as well as high salinities since they are daily affected by the tidal action. Contiguously, in a higher elevation but still frequently inundated, mid-marshes are mainly populated by *Sarcocornia perennis* subsp. *perennis* (Mill.) A.J.Scott and *Halimione portulacoides* (L.) Aell. (IDAD, 2008). These species preferably occupy creek margins, channels and banks in the mid-marshes, being bi-daily tidally inundated (IDAD, 2008). They are characterized by low to moderate soil salinities, without permanent inundation, but still flooded by the tidal water (IDAD, 2008). High marshes are mainly populated by the sea rush (*Juncus maritimus* (Lam.) and upstream, where the salinity is lower, *Phragmites australis* (Cav.) Trin. ex Steud. is the dominant species. *Phragmites australis* colonizes transitional areas wherein freshwater predominates to the saline water. According to the salinity gradient, other species also occur, namely *Bolboschoenus maritimus* (L.) Palla, previously known as *Scirpus maritimus* L., *Halimione portulacoides* (L.) Aell., *Aster tripolium* subsp. *pannonicus* (Jacq.) Soó, 1925, *Paspalum vaginatum* Sw., *Triglochin maritima* L. and *Limonium vulgare* Mill. (IDAD, 2008; AMBIECO, 2011). Considering that *Spartina versicolor* wasn't mentioned in monitoring studies performed in 2003 (Crespo, 2003; in IDAD, 2008), this invasive species has recently populated Ria de Aveiro high marshes (PSRN, 2000; IDAD, 2008; ICNF, 2012), occurring both as the dominant species or accompanied by *Juncus maritimus* and *Phragmites australis*.

According to a monitoring study performed at the Baixo Vouga Lagunar (IDAD, 2008), the most frequent (and usually more abundant) species in the Ria de Aveiro salt marsh habitat are summarized in Table 2.1.

Table 2.1 – Most common salt marsh plants occurring in low, mid and high marshes of Ria de Aveiro lagoon (data from IDAD, 2008 and from <http://www.biorede.pt/> Accessed [06/13/2013]). All species names were verified and updated according to WORMS – World Register of Marine Species (<http://www.marinespecies.org/index.php>, June 2013) and/or Flora Iberica (<http://www.floraiberica.es/>, June 2013).

Salt marsh type/ elevation	Species	Portuguese common name
Low marsh	<i>Spartina maritima</i> (Curtis) Fernald, 1916	morraça
	<i>Salicornia ramosissima</i> J.Woods, 1851	salicornia
Mid marsh	<i>Salicornia ramosissima</i> J.Woods, 1851	salicornia
	<i>Sarcocornia perennis</i> subsp. <i>perennis</i> (Mill.) A.J. Scott, 1978, previously known as <i>Arthrocnemum perenne</i> (Mill.) Moss, 1948	gramata
	<i>Halimione portulacoides</i> (L.) Aellen, 1938	gramata branca
	<i>Aster tripolium</i> L. subsp. <i>pannonicus</i> (Jacq.) Soó, 1925 ^(a)	malmequer-da-praia
	<i>Inula crithmoides</i> L., 1753 ^(b)	campana-da-praia
	<i>Spergularia salina</i> J. & K. Presl, previously known as <i>Spergularia marina</i> (Linnaeus) Besser, 1822 ^(c)	sapinhos-das-areias
	<i>Spergularia media</i> (L.) C.Presl, 1826	-
	<i>Triglochin maritimum</i> L., 1753 ^(d)	erva-do-brejo
	<i>Triglochin striata</i> Ruiz López & Pavón, 1802	-
	<i>Cotula coronopifolia</i> Linnaeus, 1753	-
	<i>Limonium vulgare</i> Mill., 1955	limónio
High marsh	<i>Juncus maritimus</i> Lam., 1789	junco-das-esteiras
	<i>Bolboschoenus maritimus</i> (L.) Palla, 1955, previously known as <i>Scirpus maritimus</i> L.	triângulo
	<i>Schoenoplectus pungens</i> (Vahl) Palla, 1889, previously known as <i>Scirpus pungens</i> Vahl.	-
	<i>Schoenoplectus lacustris</i> (L.) Palla, 1889, previously known as <i>Scirpus lacustris</i> L.	bunho
	<i>Phragmites australis</i> (Cav.) Trin. ex Steud., 1955	caniço
	<i>Typha latifolia</i> L., 1753	tábua
	<i>Typha angustifolia</i> L., 1753	
	<i>Typha domingensis</i> Pers., 1807	
	<i>Tamarix africana</i> Poir., 1789	-

^(a) WORMS reports *Aster tripolium* L., but not the subspecies mentioned in the literature; ^(b) only the genus *Inula* is recorded on WORMS; ^(c) according to WORMS; however, Flora Iberica accepts *Spergularia marina* (L.) Besser, 1822; ^(d) according to WORMS; Flora Iberica reports *Triglochin maritima* L., 1753, which is due to an update to the feminine gender.

Seagrass meadows are located in lower elevation/height. These meadows are colonized by intertidal and subtidal seagrasses, providing important ecosystem services, namely high primary productivity, carbon and nutrient cycling and sequestration, sediment stabilization and protection against erosion, food for grazers, habitat and nursery areas for fish species (Orth *et al.*, 2006; Lillebø *et al.*, 2011; Fourqurean *et al.*, 2012), being crucial for the equilibrium of the entire system. Seagrasses are widely recognized as engineering organisms (e.g. Hastings *et al.*, 2007; van der heide *et al.*, 2012), considering their ability to modify the abiotic environment and perform biogeochemical changes within the sediment. The ecosystem engineering was firstly described by Jones *et al.* (1994). Accordingly, seagrass meadows can affect the water flow and dynamics, contribute to sediment trapping and reduce

water turbidity, as well as affect nutrient dynamics, carbon sequestration and biogeochemistry of the sediments (Lillebø *et al.*, 2011; van der Heide *et al.*, 2012; Fourqurean *et al.*, 2012).

Until the 1960's, Ria de Aveiro subtidal areas were colonized by transitional waters' seagrass species (*Ruppia cirrhosa* (Petagna) Grande, 1918, *Stuckenia pectinata* (L.) Börner, 1912, previously known as *Potamogeton pectinatus* and by *Zostera* Linnaeus, 1753) (Silva *et al.*, 2004). Later on, a survey performed in the 1980's at Ovar channel recorded that subtidal areas were vegetated by the seagrasses *Stuckenia pectinata*, *Ruppia cirrhosa*, *Zostera marina* and *Zostera noltii* (also present at intertidal area), and by the macroalgae *Enteromorpha* Link, 1820 accepted as *Ulva* Linnaeus, 1753, *Chaetomorpha ligustica* (Kützinger) Kützinger, 1849 and *Gracilariopsis longissima* (S.G.Gmelin) M.Steentoft, L.M.Irvine & W.F.Farnham, 1995, previously known as *Gracilaria verrucosa* (Hudson) Papenfuss, 1950 (Silva *et al.*, 2004) (Table 2.2).

Table 2.2 – Seagrass and macroalgae species occurring in Ria de Aveiro lagoon over time (from the 1980's till present); (data from Silva *et al.*, 2004, 2009; Cunha *et al.*, 2013). All species names were verified and updated according to WORMS – World Register of marine Species (<http://www.marinespecies.org/index.php>, June 2013) and/or Flora Iberica (<http://www.floraiberica.es/>, June 2013). * Insignificant biomass most of the year (Silva *et al.*, 2009).

Tidal zonation	Macrophytes	Until the 1980's	In the 2000s'	Nowadays/Present
Subtidal	Seagrasses	<i>Ruppia cirrhosa</i> (Petagna) Grande, 1918		Unknown/not recorded
		<i>Stuckenia pectinata</i> (L.) Börner, 1912, previously known as <i>Potamogeton pectinatus</i> L.		
		<i>Zostera marina</i> Linnaeus, 1753		
		<i>Zostera noltii</i> Hornemann, 1832 ^(a)		
	Macroalgae	<i>Ulva</i> Linnaeus, 1753, previously known as <i>Enteromorpha</i> L., 1753	<i>Ulva</i> Linnaeus, 1753 ^(c)	
		<i>Chaetomorpha ligustica</i> (Kützinger) Kützinger, 1849, previously known as <i>Lola lubrica</i> (Setchell & N.L.Gardner) A.Hamel & G.Hamel, 1929 and as <i>Rhizoclonium lubricum</i> Setchell & N.L.Gardner, 1919		
		<i>Gracilariopsis longissima</i> (S.G.Gmelin) M.Steentoft, L.M.Irvine & W.F.Farnham, 1995, previously known as <i>Gracilaria verrucosa</i> (Hudson) Papenfuss, 1950	<i>Gracilariopsis longissima</i> (S.G.Gmelin) M.Steentoft, L.M.Irvine & W.F.Farnham, 1995, previously known as <i>Gracilaria verrucosa</i> (Hudson) Papenfuss, 1950	
Intertidal	Seagrasses	<i>Zostera noltii</i> ^(a)	<i>Zostera noltii</i> ^(a)	<i>Zostera noltii</i> ^(a)
	Macroalgae	<i>Ulva</i> Linnaeus, 1753, previously known as <i>Enteromorpha</i> L., 1753	<i>Ulva intestinalis</i> Linnaeus, 1753	

Tidal zonation	Macrophytes	Until the 1980's	In the 2000s'	Nowadays/Present
			<i>Ulva lactuca</i> Linnaeus, 1753	<i>Ulva</i> Linnaeus, 1753, previously known as <i>Enteromorpha</i> L., 1753
		<i>Ulva</i> Linnaeus, 1753, previously known as <i>Enteromorpha</i> L., 1753	<i>Ulva</i> Linnaeus, 1753, previously known as <i>Enteromorpha</i> L., 1753	
		<i>Gracilariopsis longissima</i> (S.G.Gmelin) M.Steentoft, L.M.Irvine & W.F.Farnham, 1995, previously known as <i>Gracilaria verrucosa</i> (Hudson) Papenfuss, 1950	<i>Gracilariopsis longissima</i> (S.G.Gmelin) M.Steentoft, L.M.Irvine & W.F.Farnham, 1995, previously known as <i>Gracilaria verrucosa</i> (Hudson) Papenfuss, 1950	<i>Gracilaria</i> Greville, 1830
			<i>Gracilaria gracilis</i> (Stackh.) Steentoft, L. M. Irvine & Farnham, 1995	
			<i>Fucus spiralis</i> Linnaeus, 1753 *	<i>Fucus</i> Linnaeus, 1753
			<i>Cladophora</i> Kützinger, 1843 *	
			<i>Bryopsis</i> J.V.Lamouroux, 1809*	
			<i>Blidingia</i> Kylin, 1947 *	
			<i>Ceramium virgatum</i> Roth, 1797, previously known as <i>Ceramium rubrum</i> C.Agardh, 1811 *	
			<i>Polysiphonia</i> Greville, 1823 *	
			<i>Bostrychia scorpioides</i> (Hudson) Montagne, 1842, previously known as <i>Fucus scorpioides</i> Hudson, 1762 *	
			<i>Stylonema</i> Reinsch, 1875, previously known as <i>Goniotrichum</i> Kützinger, 1943 *	
			<i>Sargassum muticum</i> (Yendo) Fensholt, 1955 *	

^(a) *Zostera noltei* Hornemann, 1832 according to WORMS; however, throughout the present report it will be kept the most commonly used name (*Zostera noltii*) according to the literature cited.

However, as mentioned in section 1, throughout the last decades of the 20th century, Ria de Aveiro lagoon suffered a great decline in the seagrass meadows extension and biodiversity. Thus, in the beginning of the 21st century, at Ovar channel, *Zostera noltii* (dominant species) was restricted to the intertidal zone, and mixed with the macroalgae *Ulva* sp., and *Gracilariopsis longissima* (S.G.Gmelin) M.Steentoft, L.M.Irvine & W.F.Farnham, 1995 (Silva *et al.*, 2004). According to Silva *et al.* (2009), at that time, *Gracilaria gracilis*, *Ulva intestinalis* and *Ulva lactuca* were the most abundant macroalgae species, but *Cladophora* Kützinger, 1843, *Chaetomorpha* Kützinger, 1845, *Blidingia* Kylin, 1947, *Bryopsis* J.V.Lamouroux, 1809, *Ceramium virgatum* Roth, 1797, previously known as *Ceramium rubrum* C.Agardh, 1811, *Polysiphonia* Greville, 1823, *Bostrychia scorpioides* (Hudson)

Montagne, 1842 (previously known as *Fucus scorpioides* Hudson, 1762), and *Stylonema Reinsch*, 1875, previously known as *Goniotrichum* Kützinger, 1943 were present as well (Table 2.2). However, their biomass was considered insignificant during most of the year. *Fucus spiralis* Linnaeus, 1753 and *Sargassum muticum* (Yendo) Fensholt, 1955 were recorded attached to human-made structures (Silva *et al.*, 2009). The subtidal zone was only vegetated by macroalgae (*Gracilariopsis longissima* (and *Ulva* sp.) (Silva *et al.*, 2004). Currently, recent assessments (e.g. Silva *et al.*, 2009; Cunha *et al.*, 2013) indicate that *Zostera noltii* (Figure 2.3) is restricted to the intertidal flats, whose main meadows' distribution is shown in Figure 2.4. *Gracilaria* Greville, 1830 and *Ulva* sp. are the main macroalgae genera colonizing the intertidal mudflats, namely in Ovar and Mira channels. Even though with lower abundance, *Fucus* Linnaeus, 1753 is also present in some areas of the exposed intertidal areas of Ria de Aveiro.



Figure 2.3 – *Zostera noltii* meadow in Mira channel (photo: ©AI Lillebø, 2012).



Figure 2.4 – Aerial view of Ria de Aveiro habitats distribution and the location of the main *Zostera noltii* meadows in 2012 (team personal observation); (source: map from Google Earth; data from AMBIECO report for Polis Littoral Ria de Aveiro, 2011).

Mudflats are colonized by benthic microalgae (microphytobenthos) forming extensive biofilms on the sediment surface (e.g. [Vieira *et al.*, 2013](#)), being highly productive ([Underwood and Kromkam, 1999](#)). Biofilms also contribute to the protection against erosion through the compaction and biostabilization of the sediment due to the exopolyssacharides production ([Tolhursf *et al.*, 2002](#); [Lundkvist *et al.*, 2007](#)). Mudflats provide habitat to many macroinvertebrate species and are an important feeding source for bird species. Actually, mudflats can support extremely high densities of invertebrates ([McLusky and Elliott, 2004](#)) providing high densities of polychaetes, molluscs, crustaceans and other invertebrates as a resource for many birds and fish predators. Bait digging is frequent in Ria de Aveiro mudflats, showing their social-economical role in the adjacent human population ([Cunha *et al.*, 2005](#); [Freitas *et al.*, 2011](#)). These benthic communities have the ability to influence and change the physical and chemical parameters of their adjacent sediment. Through bioturbation, these organisms are responsible for physical displacement of water and particles within the sediment ([Kristensen *et al.*, 2012](#)). Burrows construction and maintenance leads to chlorophyll a (microphytobenthos), organic matter and other particles mixing and relocation within the sediment layer. As a result, biogeochemical changes occur, which, according to [Kristensen *et al.* \(2012\)](#), corresponds to an ecosystem engineering process, rather than bioturbation. Moreover, temperate mudflats and its benthic organisms, namely primary producers (e.g. [Sundbäck *et al.*, 2000](#)), microbial communities (e.g. [Rysgaard *et al.*, 1995](#); [Risgaard-Petersen, 2004](#)) and macrofaunal organisms (e.g. [Rysgaard *et al.*, 1995](#); [Fanjul *et al.*, 2011](#)), have an important role on the nutrient cycling at the sediment level, contributing to the overall system equilibrium.

Regarding benthic macroinvertebrates and mobile megafauna in Ria de Aveiro lagoon, Table 2.4 show the list of species that can be found based on literature review of the following authors: [Moreira *et al.*, 1993](#); [Nunes *et al.*, 2008](#); [Rodrigues *et al.*, 2011](#); MAMAOT/ARHCentro, 2012 and from <http://www.biorede.pt/>.

Table 2.4 – Macroinvertebrates and mobile megafauna species occurring in Ria de Aveiro lagoon (Moreira *et al.*, 1993; Nunes *et al.*, 2008; Rodrigues *et al.*, 2011; MAMAOT/ ARHCentro, 2012 and from <http://www.biorede.pt/>). All species names were verified and updated according to WORMS – World Register of marine Species (<http://www.marinespecies.org/index.php>, June 2013).

Class	Family	Specie
Anthozoa	Hormathiidae	<i>Calliactis parasitica</i> (Couch, 1842)
Anthozoa	Sagartiidae	<i>Cereus pedunculatus</i> (Pennant, 1777)
Asciidiacea	Molgulidae	<i>Molgula manhattensis</i> (De Kay, 1843)
Bivalvia	Cardiidae	<i>Cerastoderma edule</i> (Linnaeus, 1758)
Bivalvia	Cyrenidae	<i>Corbicula fluminea</i> (O. F. Müller, 1774)
Bivalvia	Donacidae	<i>Donax semistriatus</i> (Poli, 1795)
Bivalvia	Mactridae	<i>Spisula solida</i> (Linnaeus, 1758)
Bivalvia	Myidae	<i>Mya arenaria</i> (Linnaeus, 1758)
Bivalvia	Nuculidae	<i>Nucula nucleus</i> (Linnaeus, 1758)
Bivalvia	Semelidae	<i>Scrobicularia plana</i> (da Costa, 1778)
Bivalvia	Semelidae	<i>Abra ovata</i> (Philippi, 1836) - (unaccepted) <i>Abra segmentum</i> (Récluz, 1843)
Bivalvia	Semelidae	<i>Abra alba</i> (W. Wood, 1802)
Bivalvia	Solenidae	<i>Solen marginatus</i> (Pulteney, 1799)
Bivalvia	Tellinidae	<i>Tellina tenuis</i> (da Costa, 1778) - (unaccepted) <i>Angulus tenuis</i> (da Costa, 1778)
Bivalvia	Veneridae	<i>Ruditapes decussatus</i> (Linnaeus, 1758) - (unaccepted) <i>Venerupis decussate</i> (Linnaeus, 1758)
Bivalvia	Veneridae	<i>Turtonia minuta</i> (Fabricius, 1780)
Bivalvia	Veneridae	<i>Venerupis pullastra</i> (Montagu, 1803) - (unaccepted) <i>Venerupis corrugata</i> (Gmelin, 1791)
Cephalopoda	Sepiidae	<i>Sepia officinalis</i> (Linnaeus, 1758)
Clitellata	Tubificidae	<i>Tubificoides benedii</i> (Udekem, 1855)
Gastropoda	Haminoeidae	<i>Haminoea orbignyana</i> (Férussac, 1822)
Gastropoda	Hydrobiidae	<i>Hydrobia ulvae</i> (Pennant, 1777) - (unaccepted) <i>Peringia ulvae</i> (Pennant, 1777)
Gastropoda	Nassariidae	<i>Nassarius reticulatus</i> (Linnaeus, 1758)
Gastropoda	Tateidae	<i>Potamopyrgus jenkinsi</i> (E. A. Smith, 1889) - (unaccepted) <i>Potamopyrgus antipodarum</i> (J. E. Gray, 1843)
Malacostraca	Ampithoidae	<i>Amphithoe ferox</i> (Chevreux, 1901)
Malacostraca	Ampithoidae	<i>Ampithoe valida</i> (Smith, 1873)
Malacostraca	Anthuridae	<i>Cyathura carinata</i> (Krøyer, 1847)
Malacostraca	Aoridae	<i>Aora typica</i> (Krøyer, 1845)
Malacostraca	Aoridae	<i>Microdeutopus anomalus</i> (Rathke, 1843)
Malacostraca	Bathyporeiidae	<i>Bathyporeia tenuipes</i> (Meinert, 1877)
Malacostraca	Chaetiliidae	<i>Saduriella losadai</i> (Holthuis, 1964)
Malacostraca	Corophiidae	<i>Corophium acherusicum</i> (Costa, 1853) - (unaccepted) <i>Monocorophium acherusicum</i> (Costa, 1853)
Malacostraca	Corophiidae	<i>Corophium insidiosum</i> (Crawford, 1937) - (unaccepted) <i>Monocorophium insidiosum</i> (Crawford, 1937)
Malacostraca	Corophiidae	<i>Corophium multisetosum</i> (Stock, 1952)
Malacostraca	Crangonidae	<i>Crangon crangon</i> (Linnaeus, 1758)
Malacostraca	Gammaridae	<i>Gammarus</i> sp.
Malacostraca	Idoteidae	<i>Idotea chelipes</i> (Pallas, 1766)
Malacostraca	Melitidae	<i>Melita palmata</i> (Montagu, 1804)

Malacostraca	Mysidae	<i>Diamysis bahirensis</i> (Sars G.O., 1877)
Malacostraca	Mysidae	<i>Gastrosaccus spinifer</i> (Goës, 1864)
Malacostraca	Mysidae	<i>Mesopodopsis slabberi</i> (van Beneden, 1861)
Malacostraca	Mysidae	<i>Paramysis (Longidentia) nouveli</i> (Labat, 1953)
Malacostraca	Mysidae	<i>Paramysis (Pseudoparamysis) bacescoi</i> (Labat, 1953)
Malacostraca	Portunidae	<i>Carcinus maenas</i> (Linnaeus, 1758)
Malacostraca	Sphaeromatidae	<i>Lekanesphaera</i> sp.
Malacostraca	Tanaidae	<i>Zeuxo (Zeuxo) holdichi</i> (Bamber, 1990)
Malacostraca	Urothoidae	<i>Urothoe</i> sp.
Polychaeta	Ampharetidae	<i>Amage adspersa</i> (Grube, 1863)
Polychaeta	Ampharetidae	<i>Alkmaria romijni</i> (Horst, 1919)
Polychaeta	Ampharetidae	<i>Melinna palmata</i> (Grube, 1870)
Polychaeta	Capitellidae	<i>Capitella</i> sp.
Polychaeta	Capitellidae	<i>Heteromastus filiformis</i> (Claparède, 1864)
Polychaeta	Cirratulidae	<i>Tharyx</i> sp.
Polychaeta	Fabriciidae	<i>Manayunkia aestuarina</i> (Bourne, 1883)
Polychaeta	Glyceridae	<i>Glycera tridactyla</i> (Schmarda, 1861)
Polychaeta	Hesionidae	<i>Microphthalmus</i> sp.
Polychaeta	Lumbrineridae	<i>Lumbrineris tetraura</i> (Schmarda, 1861)
Polychaeta	Nephtyidae	<i>Nephtys cirrosa</i> (Ehlers, 1868)
Polychaeta	Nereididae	<i>Nereis diversicolor</i> (Müller, 1776) - (unaccepted) <i>Hediste diversicolor</i> (O.F. Müller, 1776)
Polychaeta	Onuphidae	<i>Diopatra marocensis</i> (Paxton, Fadlaoui & Lechapt, 1995)
Polychaeta	Onuphidae	<i>Diopatra micrura</i> (Pires, Paxton, Quintino & Rodrigues, 2010)
Polychaeta	Onuphidae	<i>Diopatra neapolitana</i> (Delle Chiaje, 1841)
Polychaeta	Opheliidae	<i>Ophelia neglecta</i> (Schneider, 1892)
Polychaeta	Orbiniidae	<i>Scoloplos armiger</i> (Müller, 1776) - (unaccepted) <i>Scoloplos (Scoloplos) armiger</i> (Müller, 1776)
Polychaeta	Oweniidae	<i>Owenia fusiformis</i> (Delle Chiaje, 1844)
Polychaeta	Pectinariidae	<i>Pectinaria koreni</i> (Malmgren, 1866)
Polychaeta	Phyllodocidae	<i>Eteone foliosa</i> (Quatrefages, 1866) - (unaccepted) <i>Hypereteone foliosa</i> (Quatrefages, 1865)
Polychaeta	Phyllodocidae	<i>Eteone picta</i> (Quatrefages, 1866) - (unaccepted) <i>Mysta picta</i> (Quatrefages, 1866)
Polychaeta	Phyllodocidae	<i>Eumida bahusiensis</i> (Bergstrom, 1914)
Polychaeta	Phyllodocidae	<i>Eumida sanguinea</i> (Örsted, 1843)
Polychaeta	Protodrilidae	<i>Protodrilus</i> sp.
Polychaeta	Serpulidae	<i>Pomatoceros triqueter</i> (Linnaeus, 1758) - (unaccepted) <i>Spirobranchus triqueter</i> (Linnaeus, 1758)
Polychaeta	Serpulidae	<i>Mercierella enigmatica</i> (Fauvel, 1923) - (unaccepted) <i>Ficopomatus enigmaticus</i> (Fauvel, 1923)
Polychaeta	Sigalionidae	<i>Pisone remota</i> (Southern, 1914)
Polychaeta	Spionidae	<i>Prionospio fallax</i> (Söderström, 1920)
Polychaeta	Spionidae	<i>Polydora ligni</i> (Webster, 1879) - (unaccepted) <i>Polydora cornuta</i> (Bosc, 1802)
Polychaeta	Spionidae	<i>Pygospio elegans</i> (Claparède, 1863)
Polychaeta	Spionidae	<i>Spio</i> spp.
Polychaeta	Spionidae	<i>Pseudopolydora paucibranchiata</i> (Okuda, 1937)
Polychaeta	Spionidae	<i>Malacoceros</i> sp.
Polychaeta	Spionidae	<i>Spiophanes bombyx</i> (Claparède, 1870)
Polychaeta	Spionidae	<i>Streblospio shrubsolii</i> (Buchanan, 1890)

Polychaeta	Syllidae	<i>Autolytus aurantiacus</i> (Claparède, 1868) - (unaccepted) <i>Proceraea aurantiaca</i> (Claparède, 1868)
Polychaeta	Syllidae	<i>Autolytus langerhansi</i> (Gidholm, 1967) - (unaccepted) <i>Myrianida langerhansi</i> (Gidholm, 1967)
Polychaeta	Terebellidae	<i>Polycirrus</i> sp.
Polychaeta	Terebellidae	<i>Lanice conchilega</i> (Pallas, 1766)

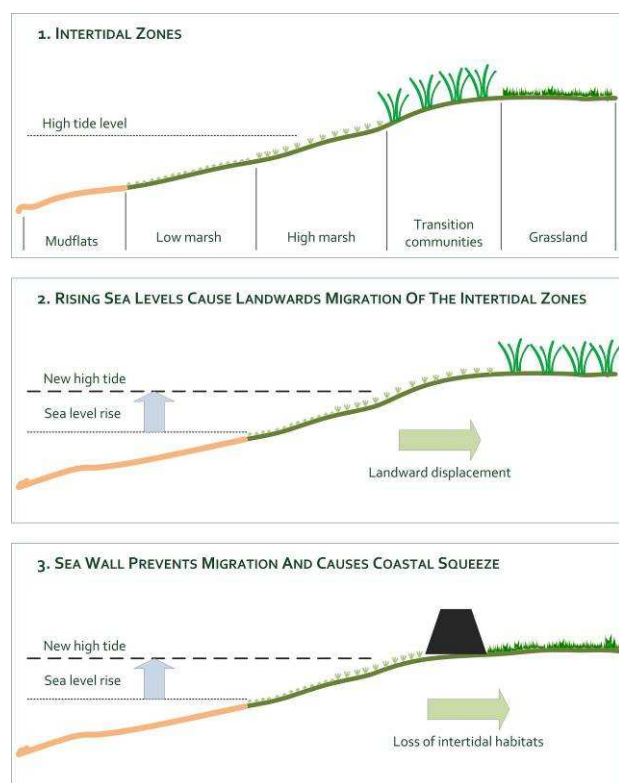
Other specimens (not identified to the species level) have also been reported for Ria de Aveiro by Rodrigues *et al.*, (2011), namely from the Phylum Nematoda and Nemertina; the Class Ascidiacea, Anthozoa, Ophiuroidea and Oligochaeta; and the crustacean amphipods from the Family Caprellidae, Isaeidae and Aoridae. *Diopatra marocensis* occurs sympatrically with *Diopatra neapolitana* and with *Diopatra micrura*, in Ria de Aveiro (Pires *et al.*, 2012).

2.3 Ria key benthic habitats in the context of Climate change:

According to recent climate change scenarios (IPCC, 2007, <http://www.pnud.cl/recientes/IPCC-Report.pdf>) southern Europe will tend to become dryer despite winter torrential rain events becoming more frequent, increasing freshwater discharges. Consequently, erosion processes and material loads into transitional areas are expected to increase, leading to an aggravation of light-limiting conditions to phytoplankton and consequently to primary production in the intertidal habitats. This will inevitably disturb the trophic structure of the system, and consequently affect the system secondary production including fishery-stocks. Sea level rise is also expected to cause changes inside estuaries (intertidal areas), affecting the hydrodynamics of these transition systems.

Salt marshes – Salt marshes, as intertidal habitats, are likely to face many effects of climate change in the next decades, which consequently will affect its ecosystem services. Considering their narrow and transitional function between land and sea, these ecosystems are particularly sensitive. The potential effects of these changes on the salt marsh dynamics, structure and communities have been increasingly studied in the last years/decades (e.g. Allen, 1990,1997; Simas *et al.*, 2001; Feagin *et al.*, 2010; Thorne *et al.*, 2012). The retreat of shorelines due to sea level rise, and the blockage of coastal areas inland migration (e.g. due to human-made sea walls to protect agricultural lands) was named “coastal squeeze” (Doody, 2004). This phenomenon is the result of many combined factors, namely, accelerated rates of sea level rise, coastal population increase and its inherent activities in coastal areas (anthropogenic climate change). Accordingly, “coastal salt marsh squeeze” (Figure 2.1) leads

to a retreat of seaward edge of wetlands and simultaneous steep gradient occurrence, which can block migration of tidal wetlands inland (e.g. [Torio and Chmura, in press](#)). As a result of



sea level rise, salt marshes flooding frequency and duration may increase beyond the tolerance of the vegetation, leading to soil accumulation ([FitzGerald et al., 2008](#)); tidal currents may also be stronger increasing coastal erosion and, consequently, the retreat of wetlands/salt marshes may occur. In addition, increasing coastal anthropogenic effects have disturbed and eliminated salt marshes ([Silliman et al., 2009 in Chmura, in press](#)) and anthropogenic barriers and steep gradients will prevent migration inland ([Feagin et al., 2010](#)).

Figure 2.1 – Schematic representation of “coastal squeeze” phenomena due to sea level rise (© J.A. Soares).

Salt marshes are widely recognized by their effective carbon sink capacity. However, this ecosystem service may be threatened by the climate change effects ([Kirwann and Mudd, 2012](#)), namely sea level rise ([Chmura, in press](#)). It has been suggested that the net impact of climate change on carbon accumulation in intertidal wetlands depend on the competition between mineral sediment deposition and organic-matter accumulation. Thus, it was suggested that this impact will increase carbon burial rates in the first half of the twenty-first century, but that carbon–climate feedbacks are likely to diminish over time ([Kirwan and Mudd, 2012](#)). The sustainability of salt marshes (and maintenance of its carbon sink capacity) will depend as well on the ability to migrate inland, thus, on the “coastal squeeze” inheritance ([Chmura, in press](#)). Hydrological changes in the system can lead to drought which, in turn, induces changes to porewater and sediment chemistry (hypersalinity or mobilization of metals, redox changes) leading to acute marsh dieback (AMD) ([Hughes et al., 2012](#)).

In case of increased inundation and salinity of South Africa salt marshes, either a landward migration of salt marsh can occur, or a dieback of salt marsh vegetation, an increase in water level and sensitivity to prolonged submergence; thus, reducing species diversity of the system (Tabot and Adams, 2013).

Particularly in Ria de Aveiro, sea level rise predictions show that salt marshes dynamics and stability will be closely related to/dependent on the hydrodynamic conditions under climate change context, namely regarding sea level rise (Valentim *et al.*, 2013). Thus, nutrients, sediment salinity and moisture, saline stress and relative vegetation patterns, are some of the parameters likely to be affected, compromising salt marshes stability (Valentim *et al.*, 2013).

Salt marsh plants are morphological and physiological adapted to its specific environmental conditions regarding location/zonation/elevation within the salt marsh. Therefore, plants show different adaptations to its photosynthetic pathway (either C₃ or C₄) (Table 2.3). Increased performance of C₄ plants when the CO₂ availability is low and the temperature is warm is due to the specialized adaptation from the C₃ ancestor's at the metabolic and morphologic level (Kranz anatomy) (Ehleringer *et al.*, 1997). Thus, increasing CO₂ availability in the atmosphere (as a consequence of climate change) might favour C₃ pathway rather than C₄ (Sousa *et al.*, 2010). In addition, as a result of CO₂ and temperature increase, productivity increase is much higher in C₃ species than in C₄ (Koch *et al.*, 2013).

Table 2.3 – Adaptations of salt marsh plant species regarding the C₃ and C₄ photosynthetic pathway (adapted from Sousa *et al.* (2010); summary based on the following publications: (a) Kakani *et al.*, 2008; (b) Ehleringer 1993; (c) Sage and McKown 2006; (d) Ehleringer *et al.*, 1991, 1997; (e) Nieva *et al.*, 1999 (f) Adam 1990)

Plant adaptations	Photosynthetic pathway	
	C ₃	C ₄
Temperature and CO ₂	Adapted to low temperatures ^(a) and high CO ₂ ^(b)	Adapted to warm temperature and low CO ₂ conditions ^(a,d)
Light	-	High light conditions ^(c)
Morphological and metabolic advantage	-	Kranz anatomy: specialized adaptation from C ₃ ancestors ^(d)
Ability to acclimate	High ability to acclimate to low light, temperature variation or elevated CO ₂ ^(c)	Low ability to acclimate to adapted/changed environments ^(c) ; At low salinity: higher photosynthetic capacity than C ₃ ^(e)
Other aspects	-	Higher potential productivity than C ₃ plants ^(f)

Seagrasses - Climate change is likely to have a strong impact on seagrass meadows productivity, distribution and function (Short and Neckles, 1999; Orth *et al.*, 2006; Jordá *et al.*, 2012; Koch *et al.*, 2013) making them highly vulnerable to these effects. Recent works have predicted increases in the maximum summer temperature and probably an increase in the frequency of heat waves in the same period (Sánchez *et al.*, 2004). In addition, by the end of the century, global mean temperatures are expected to rise by 1-3.5 °C (Watson *et al.*, 1996). Even though the increase in seawater temperature will most likely affect seagrasses, its effects are extremely difficult to preview, since the consequences of these changes greatly depend on the coastal lagoon characteristics and on the seagrass species characteristics (e.g. thermal tolerance and optimum temperatures for photosynthesis, respiration and growth). For instance, *Zostera marina*, a temperate subtidal seagrass was negatively affected by the temperature increases, with its shoot densities decreasing as a consequence of these events (Ehlers *et al.*, 2008). Global temperature increase might also lead to competition of tropical seagrass species with temperate species in temperate regions (Koch *et al.*, 2013). Considering the increase in ocean carbon dioxide and ocean acidification, seagrasses are likely to increase its photosynthetic activity and growth (Koch *et al.*, 2013).

Due to climate change, the frequency of extreme weather events are also predicted to increase, leading to the increase in the occurrence of flood and drought events. Consequently, benthic primary producers such as seagrasses might be affected by a stronger wave action and consequent erosion, but also by a reduction in light availability (eventually, light limitation) to the subtidal seagrass meadows (*“shading effect”*) due to sediment resuspension. The *“ability of seagrass ecosystems to recover from disturbance and return to its original state”* – its resilience (Elliott *et al.*, 2007 and references therein) – may be hard to achieve and depend on the interaction of many different factors (e.g. Lillebø *et al.*, 2011). In the context of climate change and as a result of its inherent drivers/pressures (e.g. global warming/heat waves), seagrass meadows resilience may depend on different factors, namely its genotypic diversity (Ehlers *et al.*, 2008). Based on modelling studies, Carr *et al.* (2012) suggested that the increase of high water temperature events might lead to more frequent eelgrass die-offs (anticipated/indicated by *“flickering”* and *“slowing down”* processes, as changes in the number of leaves per shoot) due to its inability to recover from this disturbance. In addition, changes in the hydrodinamism of water masses and salinities, as a result of extreme weather

events – floods and/or droughts (Lillebø *et al.*, 2011), will also have a profound effect on macrophyte distribution and survival within coastal ecosystems.

Sea-level rise, whose projection for the end of the 21st century is 50 cm as a “*best estimate*”, but can range between 15 and 95 cm (Watson *et al.*, 1996; IPCC, 2007; Rahmstorf, 2007), will affect the seagrass meadows functioning as well. As key factors for the seagrass productivity, survival and distribution (Livingston *et al.*, 1998; Moore and Wetzel, 2000; Plus *et al.*, 2001), the reduction of light availability and the increase in seawater temperature will directly affect seagrass meadows functioning and, consequently, affect other trophic levels in the coastal lagoons.

Mudflats - Coastal lagoons are considered as highly productive systems and usually show high abundances of benthic macrofauna species. This fauna plays an essential role in the maintenance of coastal ecosystem integrity by mediating exchanges and transformations of energy and materials, including nutrients, between the water column and the sediments (Fanjul *et al.*, 2011; Kristensen *et al.*, 2012). Furthermore, production by the macrobenthos provides an important vehicle of trophic transfer within the coastal ecosystem (Diaz and Rosenberg, 1995). Even though the increase in carbon dioxide and H⁺ positively affects photosynthesis, calcification processes will be negatively affected by climate change and ocean acidification, namely calcified macroalgae (Koch *et al.*, 2013). Regarding microphytobenthos, Vieira *et al.* (2013) showed that the photosynthetic capacity of these intertidal communities increases with transient temperature increases, anticipating an increase in its primary productivity as a result of global warming.

3 Benthic biological diversity and ecological quality

The data set used to qualitatively map the benthic macrofauna biological diversity in Ria de Aveiro was based mostly on the published results by Rodrigues *et al.* (2011), but also by Nunes *et al.* 2008, and by authors in *situ* observation of *Zostera noltii* meadows distribution. This way, the authors pretended to map the benthic macrofauna biological diversity taking into account not only the natural environmental factors that are important to define the structure of the infaunal assemblages, but also the interaction of both natural and anthropogenic factors in specific areas of Ria. This analysis and interpretation may therefore, slightly differ from the original publications. For the purpose of this report the authors will only consider the five main affinity groups (A, B₁, B_{2.1}, B_{2.2} and C) identified by Rodrigues *et*

al. (2011) (for a amore detailed description of the statistical methodology please see the original pubication). In the refereed study the environmental data matrix included sediment grain-size and organic matter content, Eh, some hydrodynamics descriptors and salinity. However, the correlation between biological data and environmental data was low, being the one with the sediment grain-size descriptors particularly low (Rodrigues *et al.*, 2011). Authors concluded that hydrodynamics descriptors and salinity were the main factors reflecting the benthic assemblage's distribution from the mouth upstream. Table 3.1 indicates the location of the five goupes considered, the original data on sediment grain-size, abundance and species richness (Rodrigues *et al.*, 2011) and the qualitative interpretation of the data used for the biological diversity mapping.

Table 3.1 – The location of the five goupes considered, summary of the original data on sediment grain-size, abundance and species richness (Rodrigues *et al.*, 2011) and the qualitative interpretation of the data used for the biological diversity mapping.

	B ₁	B _{2.1}	B _{2.2}	A	C
Location (original publication)	<i>Zostera</i> bed (Ovar channel)	Further upstera of group A	Further upstera of group B _{2.1}	Near the lagoons sea boundary	Inner most areas of the various channels, at each end, except the Ílhavo and Ovar channels
Abundance & Species richness (Quantitative)	671 (ind/0.05m ²) 17 (sp/0.05m ²)	395 (ind/0.05m ²) 16 (sp/0.05m ²)	398 (ind/0.05m ²) 9 (sp/0.05m ²)	65 (ind/0.05m ²) 7 (sp/0.05m ²)	129 (ind/0.05m ²) 3 (sp/0.05m ²)
Species richness & abundance (Qualitative)	Highest	Higher	Intermediate	Lower (lowest a part from site C)	Lowest
Attributed color	Blue	Green	Yellow	Orange	Red
Sediment type	Highest content in fine sediments (59%)	Intermediate content in fine sediments (34%)	Higher content in fine sediments (45%)	High gravel and sand content; Very low fine sediments content (7%)	Intermediate content in fine sediments (34%)

Nunes *et al.* 2008 performed a more detailed study restricted to the historical contamination Laranjo basin concluding that macrobenthic community structure changed significantly along the mercury gradient (for a more detailed information regarding the Hg historical contamination see Pereira *et al.*, 2009). Results showed that the increase of mercury contamination was associated with reduced total abundance, lower species diversity and dominance of taxa tolerant to mercury (Nunes *et al.*, 2008) (for a amore detailed description of the statistical methodology please see the original pubication). For the purpose of this

report authors will only consider the following qualitative levels for species richness and abundance, i.e., higher (green) corresponding to the reference site, intermediate (yellow) and lowest (red), corresponding to the most contaminated site (in this study no significant differences were found regarding the sediment composition). Furthermore, for the purpose of this report authors will also attribute the highest species richness and abundance (blue color) to other locations in Ria with *Zostera noltii* meadows, based on the authors personal observation in 2012 (see also Figure 2.4). The following map (Figure 4.1) illustrates the spacialization of the gathered information.

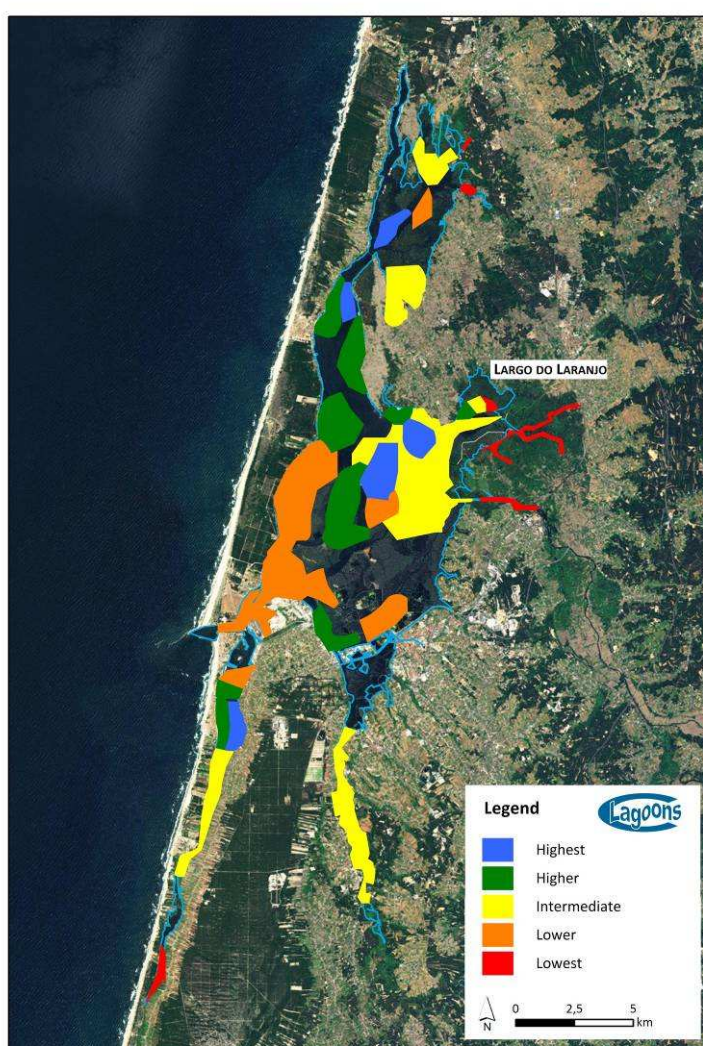


Figure 3.1. – Aerial view of Ria de Aveiro with the representation of the qualitative distribution of the benthic macrofauna biological diversity (source: map from Google Earth; data from [Rodrigues et al., 2011](#), [Nunes et al., 2009](#)). Note: the size of the represented areas are only an indication, they do not intend to represent the exact area.

Rodrigues *et al.* (2011) also assessed an approximation of the ecological quality status of Ria based on the benthic macroinvertebrates, as shown in Table 3.2 (for a amore detailed description of the statistical methodology please see the original pubication).

Table 3.2 – The location of the five goups considered and the respective ecological quality statement based on the benthic macroinvertebrates (Rodrigues *et al.*, 2011)

	B ₁	B _{2,1}	B _{2,2}	A	C
Location (original publication)	<i>Zostera</i> bed (Ovar channel)	Further upstera of group A	Further upstera of group B _{2,1}	Near the lagoons sea boundary	Inner most areas of the various channels, at each end, except the Ílhavo and Ovar channels
AMBI from the hole data set	Moderate	Moderate	Poor	Poor	Bad
AMBI from each assemblage data set	Good	Moderate	Moderate	Moderate	Poor

However, Rodrigues *et al.* (2011) concluded that none of the followed approaches satisfied the theoretical grounds, either by over-estimating the worst conditions or by potentially under-estimate the worst conditions, if impacted areas are erroneously included in the natural gradient. Taking into account the available data and information (*see* chapter 1.4) for the purpose of this report authors decided not to map the ecological quality statement based on the benthic macroinvertebrates.

4 Fish community

The pisces class (ichthyofauna) in Ria de Aveiro lagoon can be divided into four ecological categories: i) marine species occasionally entering the lagoon with the tides (e.g. *Sardina pilchardus*, *Gobius niger*, *Symphodus bairdii*, *Sparus aurata*, *Trigla lucerna*, *Callionymus lyra*, *Scophthalmus rhombus*, *Ammodytes tobianus*); ii) marine species dependent on the lagoon environment for food resources, shelter and nursery grounds (e.g. *Lisa aurata*, *Dicentrarchus labrax*, *Platichthys flesus*); iii) resident species, well adapted to the lagoon (e.g. *Atherina presbyter*, *A. boyeri*); iv) migratory species (e.g. *Anguilla Anguilla*, *Alosa alosa*, *Lampetra planeri*) (AMRia/CPU, 2007).

Table 4.1 shows in more detail the list of the main fish species that can be found (or were found in the past) in Ria de Aveiro lagoon, based on literature review of the following publications: [Rebelo and Pombo \(2001\)](#) and [Pombo \(2005\)](#).

Table 4.1 – Literature review of the main fish species that can be found (or were found in the past) in Ria de Aveiro lagoon based on the following publications: [Rebelo and Pombo \(2001\)](#) and [Pombo \(2005\)](#). (* The correspondent location is show in the map in figure 1; ** Observed only in the beginning of XX century).

Occurrence Status	Frequency	Portuguese common name	Scientific name	Higher density occurrence*
Resident in the lagoon	Frequent	Caboz-da-areia	<i>Pomatoschistus minutis</i> (Pallas, 1770)	Areão
		Caboz-negro	<i>Gobius niger</i> (Linnaeus, 1758)	Torreira
		Caboz-transparente	<i>Aphia minuta</i> (Risso, 1810)	Carregal
		Marinha-comum	<i>Syngnathus acus</i> (Linnaeus, 1875)	Areão Carregal
		Peixe-Rei-do-Mediterrâneo	<i>Atherina boyeri</i> (Risso, 1810)	Carregal Vagos Torreira Gafanha
		Solha-das-pedras	<i>Platichthys flesus</i> (Linnaeus, 1875)	Areão
		Tainha-olhalvo	<i>Mugil cephalus</i> (Linnaeus, 1758)	Carregal Areão
		Agulhinha	<i>Syngnathus abaster</i> (Risso, 1826)	Areão e Carregal
	Scarce	Bodião-vulgar	<i>Symphodus melops</i> (Linnaeus, 1758)	Torreira
		Caboz-comum	<i>Pomatoschistus microps</i> (kroyer, 1838)	Vagueira Areão Carregal
		Caboz-da-rocha	<i>Gobius paganellus</i> (Linnaeus, 1758)	
		Caboz-de-Belloti	<i>Gobius ater</i> (Belloti, 1888)	
		Galeota-menor	<i>Ammodytes tobianus</i> (Linnaeus, 1758)	
		Marinha-focinho-grosso	<i>Syngnathus typhle</i> (Linnaeus, 1758)	Carregal
		Marinha	<i>Nerophis ophidion</i> (Linnaeus, 1758)	
	Absent**	Caboz-de-Schmidt	<i>Gobius strictus</i> (Fage, 1907)	
		Caboz-manchado	<i>Pomatoschistus pictus</i> (Malm, 1865)	
		Cavalo marinho	<i>Hippocampus hippocampus</i> (Linnaeus, 1758)	
		Esganagata-marinha	<i>Spinachia spinachia</i> (Linnaeus, 1758)	
Marine juveniles	Frequent	Linguado-legítimo	<i>Solea solea</i> (Linnaeus, 1758)	
		Robalo-legítimo	<i>Dicentrarchus labrax</i> (Linnaeus, 1758)	Laranjo Vagos Gafanha
		Rodovalho	<i>Scophthalmus rhombus</i> (Linnaeus, 1758)	Barra Torreira
		Ruivo	<i>Chelidonichthys lucerna</i> (Linnaeus, 1758)	
		Sargo-legítimo	<i>Diplodus sargus</i> (Linnaeus, 1758)	Torreira
	Scarce	Choupa	<i>Spondyllosoma cantharus</i> (Linnaeus, 1758)	Barra
		Faneca	<i>Trisopterus luscus</i> (Linnaeus, 1758)	
		Linguado	<i>Solea senegalensis</i> (Kaup, 1858)	Gafanha

Occurrence Status	Frequency	Portuguese common name	Scientific name	Higher density occurrence*
		branco		
		Peixe-rei	<i>Atherina presbyter</i> (Cuvier, 1829)	Gafanha Barra Torreira
		Robalo-baila	<i>Dicentrarchus punctatus</i> (Bloch, 1792)	
		Sargo-alcorraz	<i>Diplodus annularis</i> (Linnaeus, 1758)	
		Sargo-safia	<i>Diplodus vulgaris</i> (E. Geoffrey Saint-Hilaire, 1817)	Gafanha
		Solha-legítima	<i>Pleuronectes platessa</i> (Linnaeus, 1758)	
	Absent**	Goraz ou besugo	<i>Pagellus bogaraveo</i> (Brunnich, 1768)	
		Juliana ou paloco	<i>Pollachius pollachius</i> (Linnaeus, 1758)	
Migratory (seasonal)	Frequent	Biqueirão	<i>Engraulis encrasicolus</i> (Linnaeus, 1758)	Areão
		Laíbeque-de-cinco-barbilhos	<i>Ciliata mustela</i> (Linnaeus, 1758)	
		Sardinha	<i>Sardina pilchardus</i> (Waulbaum, 1792)	Barra Torreira
		Taíinha-garrento	<i>Liza aurata</i> (Risso, 1810)	Torreira Vagos Areão
	Scarce	Agulha	<i>Belone belone</i> (Linnaeus, 1761)	
		Taíinha-liça	<i>Chelon labrosus</i> (Risso, 1826)	Areão e Vagos
	Absent**	Espadilha	<i>Sprattus sprattus</i> (Linnaeus, 1758)	
Adventitious visitors	Frequent	Bodião-de-bailloni	<i>Symphodus bailloni</i> (Valenciennes, 1839)	Torreira
		Dourada	<i>Sparus aurata</i> (Linnaeus, 1758)	Carregal
		Marachomba-babosa	<i>Parablennius gattorugine</i> (Linnaeus, 1758)	
		Peixe-aranha	<i>Echiichthys vipera</i> (Cuvier, 1829)	São Jacinto
		Peixe-pau-lira	<i>Callionymus lyra</i> (Linnaeus, 1758)	São Jacinto
		Salmonete-legítimo	<i>Mullus surmuletus</i> (Linnaeus, 1758)	Barra
	Scarce	Bodião-cinzento	<i>Symphodus cinereus</i> (Bonnaterre, 1788)	
		Bodião-fusco	<i>Labrus merula</i> (Linnaeus, 1758)	
		Bodião-reticulado	<i>Labrus bergylta</i> (Ascanius, 1767)	
		Boga	<i>Boops boops</i> (Linnaeus, 1758)	
		Caboz	<i>Pomatoschistus marmoratus</i> (Risso, 1810)	
		Caboz-de-quatro-manchas	<i>Deltentosteus quadrimaculatus</i> (Valenciennes, 1837)	
		Cangulo-cinzento	<i>Balistes carolinensis</i> (Gmelin, 1789)	
		Carapau	<i>Trachurus trachurus</i> (Linnaeus, 1758)	Carregal
		Congro	<i>Conger conger</i> (Linnaeus, 1758)	
		Xareu-azul	<i>Caranx crysos</i> (Mitchill, 1815)	Barra
		Galeota-maior	<i>Hyperoplus lanceolatus</i> (Le Sauvage, 1824)	
		Garó	<i>Scomber scombrus</i> (Linnaeus, 1758)	
		Linguado-da-areia	<i>Pegusa lascaris</i> (Risso, 1810)	
		Marachomba-babosa	<i>Parablennius sanguinolentus</i> (Pallas, 1814)	
		Pregado	<i>Scophthalmus maximus</i> (Linnaeus, 1758)	
		Taíinha-sabão	<i>Oedalechilus labeo</i> (Cuvier, 1829)	Carregal, Areão
	Absent	Besugo	<i>Pagellus acarne</i> (Risso, 1827)	

Occurrence Status	Frequency	Portuguese common name	Scientific name	Higher density occurrence*
		Bodião-tordo	<i>Labrus viridis</i> (Linnaeus, 1858)	
		Caneja ou caneja-lisa	<i>Mustellus mustellus</i> (Linnaeus, 1858)	
		Escorpião-roco	<i>Taurulus bubalis</i> (Euphrasen, 1786)	
		Galeota-da-areia ou ligueirão	<i>Gymnammodytes ciccerelus</i> (Rafinesque, 1810)	
		Laibeque-de-três-barbilhos	<i>Gaidropsarus vulgaris</i> (Cloquet, 1824)	
		Laibeque-do-Mediterrâneo	<i>Gaidropsarus mediterraneus</i> (Linnaeus, 1858)	
		Marachomba-frade	<i>Lipophrys pholis</i> (Linnaeus, 1858)	
		Peixe-balão	<i>Lagocephalus lagocephalus</i> (Linnaeus, 1858)	
		Piça-del-rei	<i>Coris julis</i> (Linnaeus, 1858)	
		Raia-pintada	<i>Raja asterias</i> (Delaroche, 1809)	
		Roncadeira-preta	<i>Sciaena umbra</i> (Linnaeus, 1858)	
		Salema	<i>Sarpa salpa</i> (Linnaeus, 1858)	
Migratory anadromous	Frequent	Esganagata	<i>Gasterosteus aculeatus</i> (Linnaeus, 1758)	
		Savelha	<i>Alosa fallax</i> (Lacepède, 1803)	Barra
	Scarce	Lampreia-do-mar	<i>Petromyzon marinus</i> (Linnaeus, 1758)	
		Sável	<i>Alosa alosa</i> (Linnaeus, 1758)	Rio Novo de Príncipe
		Taínha-de-salto	<i>Liza saliens</i> (Risso, 1810)	Carregal Areão Vagueira
		Taínha-fataça	<i>Liza ramada</i> (Risso, 1826)	Torreira Carregal Areão
Migratory catadromous	Frequent	Enguia	<i>Anguilla anguilla</i> (Linnaeus, 1758)	Areão Carregal
Freshwater	Frequent	Pimpão	<i>Carassius carassius</i> (Linnaeus, 1758)	Areão
	Scarce	Gambúsia	<i>Gambusia affinis</i> (Baird & Girad, 1853)	
	Absent**	Achigã	<i>Micropterus salmoides</i> (Lacépède, 1802)	
		Barbo	<i>Barbus bocagei</i> (Steindachner, 1865)	
		Peixe-encarnado	<i>Carassius auratus</i> (Linnaeus, 1758)	
		Ruivaca	<i>Rutilus macrolepidotus</i> (Steindachner, 1866)	
		Serpentina	<i>Cobitis taenia</i> (Linnaeus, 1758)	

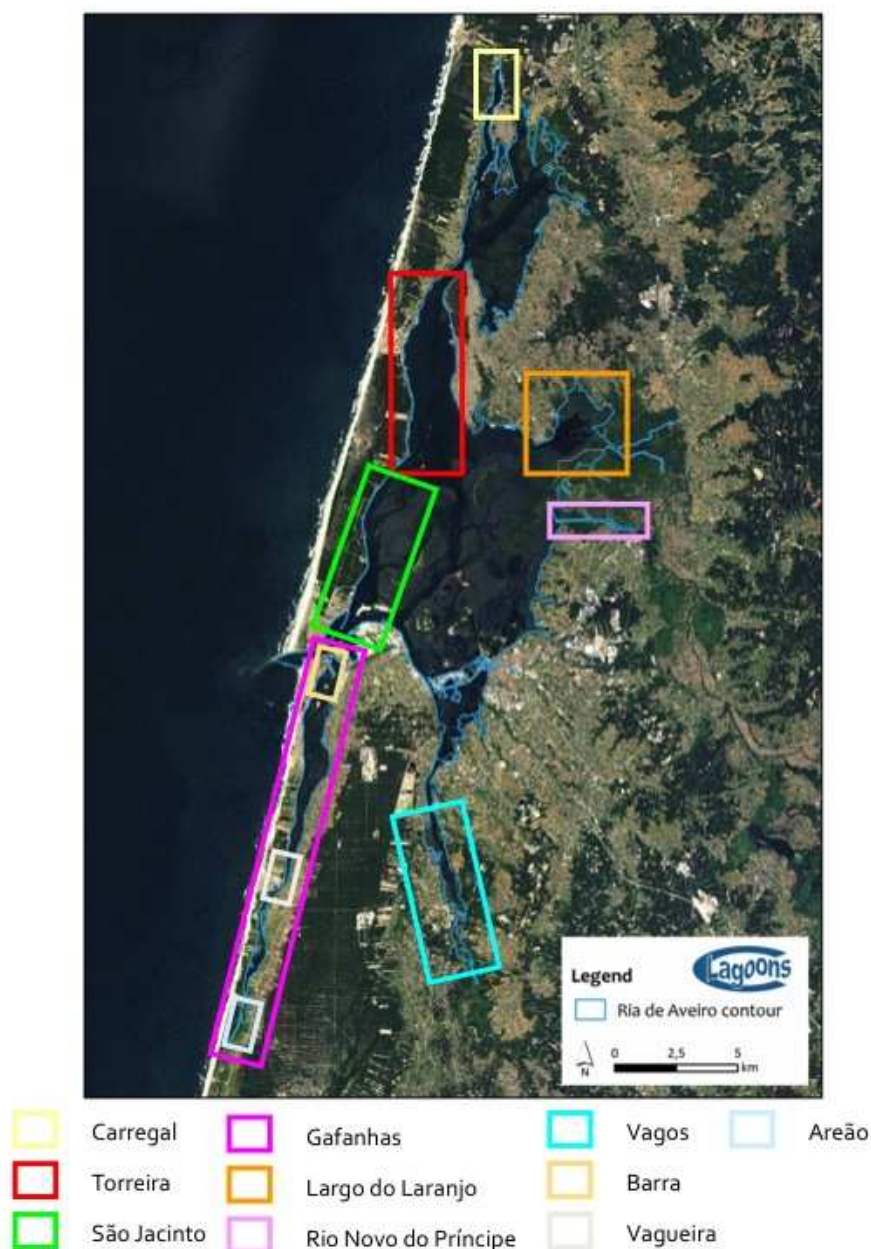


Figure 4.1. – The location of the of the main fish species that can be found (or were found in the past) in Ria de Aveiro lagoon.

5 Fisheries

Fishing, shellfish collecting and bait digging have traditionally been important sources of income in the Ria de Aveiro region, and continue so nowadays. Many families and individual fishermen depend exclusively on fishing, but in other cases this is a complementary source of income. Despite the several facilities that exist throughout the area of Ria de Aveiro for fish landing (known as “lotas”, in Portuguese) a very important part of the fish captured in these waters is not sold for the first time in any of these facilities, escaping taxation and reporting

(Carneiro *et al.*, 2002). Since keeping reliable records of these transactions and catch amounts is not possible, the economic, social and ecological evaluation of these fisheries value is difficult to make. Available studies on the ecological, social and economic importance of fisheries in the Ria de Aveiro (European Commission 2010) have not been able to disaggregate fisheries in terms of fishing vessel type, fishing grounds, types of gear or species group, and the real importance of fisheries, shellfish harvesting and bait digging in the lagoon remains unknown.

There are about 30 documented gears (from 1883 to 2000) and even though only less than half are contemplated in the fishing regulations in Ria de Aveiro (Portaria nº563/90 de 19 de Julho), the great majority are still being used nowadays (Carneiro *et al.*, 2002). These gears are essentially divided in the following general categories: digging, wounding gears, hooks and lines, traps, trawl nets and gillnets.

Polychaetes and bivalves are the main targets of the harvesting and digging activities, for their relative abundance along the shallow subtidal and intertidal flats of the lagoon, ease of capture, and growing commercial value (LAGOONS, 2012). These catches have two main final purposes: bait (e.g. solitary tube worm (*Diopatra neapolitana*), ragworm (*Hediste diversicolor* - *Nereis* (*Hediste*) *diversicolor* O.F. Müller, 1776 accepted as *Hediste diversicolor* (O.F. Müller, 1776)) and catworm (*Nephtys hombergii*)) and human consumption (e.g. common cockle (*Cerastoderma edule* - *Cardium edule* Linnaeus, 1758 accepted as *Cerastoderma edule* (Linnaeus, 1758)) european razor clam (*Solen marginatus*) and flat furrow clam (*Scrobicularia plana*)) (Cunha *et al.*, 2005). The estimations made by Cunha *et al.* (2005) show that bait harvesting in the Mira channel is very intense, with approximately 45 tons of *Diopatra neapolitana* (solitary tube worm) caught in one year, corresponding to a first sale value of more than 325 000 € (values estimated from catches between May 2001 and April 2002); more recently, Freitas *et al.* (2011) showed that, five years later, the annual catch was about that value, with the first-sale value reaching 224 000 €.

Fishing activities using wounding gears, hooks and lines, traps, trawl nets and gillnets aim for other species, such as mackerel, bream, sea bass, turbot, sole, mullet, salmon, eel, sardines, sea bream and plaice, recorded as the most commercialized fish species (Rebelo and Pombo, 2001). In 2000, there were 586 vessels licensed and only less than half made at least 1 registered landing (47.4%) (Carneiro *et al.*, 2002).

The species composition of the Ria de Aveiro's fish community changed between 1912 and the 80s (Rebelo and Macaringue, 2011). Many of the species recorded in the beginning of the last century were disappearing (from the families Ammodytidae, Blennidae, Cyprinidae, Gadidae, Gasterosteidae, Gobiidae, Labridae, Rajidae, Sciaenidae, Sparidae, Syngnathidae, Torpedinidae e Triakidae) and niches were left open to other species (from families Atherinidae, Mugilidae, Moronidae e Sparidae) (Rebelo, 1992; Rebelo and Pombo, 2001). In terms of numbers, these changes began with 57 registered species in 1912, a number that was only slightly oscillating until 1998, when the number of species decreased from 55 (registered in 1988) to 43 (registered in 1998); in 2000, studies showed that there were only 38 species in the Ria de Aveiro, a number that hasn't greatly changed, even though it has oscillated until these days. Throughout this time, there were 92 registered fish species in total (Rebelo and Macaringue, 2011).

Regardless the small amount of information available about fishing catches, it is known that the main source of income for the local fishermen are bivalves (cockle as the first one, then cross-cut carpet shell and then mussels), cuttlefish and shore crabs (Carneiro *et al.*, 2002).

6 Open questions

The following summarizes some open questions arise regarding the changes that Ria key benthic habitats are facing:

- *Presence of invasive/alien species*

The presence of invasive/alien species may interfere with the equilibrium, functioning and inter-relations of native species and its own habitats. Namely, the invasive macroinvertebrate species may affect the trophic web functioning, either through changes in the grazer species or new competition processes, which lastly affects the feeding cascade and all trophic chain/web. Accordingly, the macrophyte *S. versicolor*, considered an invasive species by ICNF, was recently recorded in the Ria high marshes. This may lead to adjustment/changes in the species competition among native and invasive salt marsh plants which, consequently, may affect the species dominance, not only in the high marsh, but in the low and mid marshes as well.

- *Loss of seagrass meadows ecosystem services*

Considering the remarkable decline of seagrass meadows in the last decades and the important ecosystem services they provide, it becomes relevant to perform the mapping of these

meadows in Ria. Then, the analysis of the seagrass decline extent at temporal and spatial scales, as well as its impacts on the ecosystem services provided, will provide a better knowledge of this important habitat of Ria de Aveiro lagoon.

- Resilience to climate change

Considering the potential climate change consequences, such as sea level rise and water temperature increase, Ria main habitats (salt marshes, seagrass meadows and mudflats) might face in the future some threats and changes. Thus, disappearance of some species and replacement of others due to competition (e.g. salt marsh macrophytes) may occur, which will therefore affect all the ecosystem dynamics.

- The future of fisheries (fish, macroinvertebrate and megainvertebrate fauna) in Ria

There is a strong dependence of professional, semi-professional and recreational fishing activities in Ria, which seems to be destabilized due to the foreseen overfishing and climatic scenarios for the area. However, there is a clear lack of systematic research on fishing activity inside the lagoon, the system carrying capacity and about the socio-economic importance of the activity in the lagoon. Therefore, the economic importance of these fishing practices is yet highly ignored and the contribution to the local economy remains obscure. The environmental effects of potential overfishing can hardly be verified since reliable catch data do not exist and trends cannot be ascertained. Policies to be applied for stock management and protection of socio-economic benefits from fishing, shellfish collecting and bait digging activities cannot be determined. Finally, the impacts of the anticipated climate change scenarios are of limited value in the current situation, since there is an abstract estimation of the current environmental and economic fishery conditions inside lagoon.

7 References

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