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ECONOMIC VALUE OF BLUE CARBON: CHALLENGES AND OPPORTUNITIES FOR PORTUGUESE COASTAL HABITATS

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Dissertation

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"*How inappropriate to call this planet earth when it is quite clearly Ocean.*"

Arthur C. Clarke

BIOGRAPHICAL NOTE

Carla Santos is graduated by the University of Porto since 2007, with a BSc in Environmental Sciences and Technology. Her sustainability and local development passion starts on the Project Local Agenda 21 in the *Nordeste Transmontano*, as she was doing her curricular internship. In 2008, she decided to increase her knowledge attending the Specialization Course in Sustainable Development and Local Agenda 21, at the Portuguese Catholic University - Porto.

Between 2001 and 2009, as an active member of *Orfeão Universitário do Porto*, she took responsibilities at the Group Board and had a very active participation in the ethnographic dance group. In 2003, was co-founder of NECTAR - an environmental student association, in which she was responsible for treasury activities and events organization.

In 2010, she collaborates in an environmental financial services company, in Florianópolis, Santa Catarina, Brazil, under the *INOV Contacto* Program, participating in projects of carbon assessment and management.

During her career she have worked in different companies, integrating several projects which main topics were: Strategic Environmental Assessment, Local Sustainability Strategies, Environmental Diagnosis (Water, Waste and Energy), Sustainable Tourism Plans, Energy Matrix, Waste Management Plans and Carbon Footprint, among others.

With 10 years of professional experience, in 2016, she decides to extend her multidisciplinary skills integrating the Master Course on Environmental Economics and Management, in Faculty of Economics in University of Porto.

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ABSTRACT

Coastal vegetated ecosystems like salt marshes and seagrass meadows, known as blue carbon ecosystems, are important natural carbon sinks by their high capacity to sequester and storage carbon. However, they are one of the most threatened ecosystems due to climate change and human pressure. This thesis aims to do an economic valuation of the sequestration and storage capacity service of Portuguese mainland blue carbon ecosystems. It is made calculations for the value of the current total carbon stock as well as for the predicted value tin2060 under two scenarios that have into account different ecosystems area loss rate. These ecosystems have a current economic value of 2.349.335€. The time evolution of blue carbon ecosystems changes is assed using two different approaches: the social cost of carbon and the marginal abatement cost. The results range between 218.412€ and 810.180€ in the *optimistic scenario*, and between 328.856€ and 1.219.863€, in the *pessimistic* scenario. Some challenges and opportunities are address in order to get a better management of these ecosystems in Portugal.

Keywords: social cost of carbon, marginal abatement cost, blue carbon, carbon storage, ecosystem services, salt marshes, seagrass meadows

Resumo

Os ecossistemas costeiros com cobertura vegetal, como os sapais e as pradarias de ervas marinhas, conhecidos como ecossistemas de carbono azul, são importantes reservatórios naturais de carbono pela elevada capacidade de sequestrar e reter carbono. Estes encontram-se, no entanto, entre os ecossistemas mais ameaçados pelos efeitos das alterações climáticas e pela pressão humana. Esta tese tem como objetivo a avaliação económica do serviço de sequestro e retenção de carbono associado aos ecossistemas costeiros como os sapais e as pradarias de ervas marinhas em Portugal continental. São efetuados cálculos para o valor económico do *stock* de carbono atual, assim como estimativas para o seu valor futuro no ano de 2060., sob dois cenários que consideram diferentes taxas de perda de área dos ecossistemas. Estes ecossistemas têm atualmente um valor económico de 2.349.335€. A sua evolução temporal é feita através de duas abordagens metodológicas: o custo social de carbono e o custo marginal de abatimento. Os resultados da perda económica associada à perda de ecossistema variam entre 218.412€ e 810.180€ no cenário otimista, e entre 328.856€ e 1.219.863€ no cenário pessimista. Alguns desafios e oportunidades são ainda destacados, com o objetivo de proporcionar uma melhor gestão dos ecossistemas costeiros em Portugal.

Palavras-chaves: custo social de carbono, custo marginal de abatimento, carbono azul, armazenamento de carbono, serviços de ecossistema, sapais, pradarias de ervas marinhas

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INTRODUCTION

Recent studies highlight coastal wetlands – mangroves, salt marshes and seagrasses - play an important role on climate change mitigation, being considered as long and effective carbon sinks (Howard *et al.*, 2017; Murray *et al.*, 2011; Nelleman *et al.*, 2009). These ecosystems can sequester and store a high amount of carbon in their biomass and more significantly in their soils and are pointed out as more efficient than most terrestrial forests (Howard *et al.*, 2017; Murray *et al.*, 2011; Nelleman *et al.*, 2009). In accordance with some studies, storage capacity of salt marshes sediments and seagrasses is respectively 48 times and 27 times higher than terrestrial temperate forest soils (Sousa *et al.*, 2017). These ecosystems are known as "blue carbon" and they have awaked the interest of scientific community and policymakers in the last few years due to its great potential for mitigating climate change damages.

Coastal ecosystems are among the most threatened natural environments worldwide, with alarming lost rates especially due to the human activity. So, they are becoming carbon sources instead of carbon sinks as it would be expected (Howard *et al.*, 2017; McLeod *et al.*, 2011). For this reason, it is internationally recognized the need to protect and restore the costal ecosystems, as it was being referred in international agreements like Paris Agreement on Climate Change and 2030 Agenda for Sustainable Development in SDG 14.

The possibility to link blue carbon interventions with conservation and climate finance is pointed out as an opportunity, and the valuation of the ecosystem services provided by coastal ecosystems, including carbon sequestration and storage, would benefit coastal managers in their policy decisions.

There are some global studies estimating economic value of blue carbon, highlighting the importance of protecting these ecosystems (Costanza *et al.*, 1989; Siikamäki *et al.*, 2012). However, there still needs to improve methodologies to calculate carbon sequestration and stocks (Carr *et al.*, 2018; Pendleton *et al.*, 2012; Siikamäki *et al.*, 2012), as well as to valuate this ecosystem service (Himes-Cornell *et al.*, 2018; Jerath *et al.*, 2016). In addition, there is a requirement for site specific data to support national and local scale policy (Himes-Cornell *et al.*, 2018).

In Portugal, some studies in this field are now emerging. There are some studies attempting to estimate carbon storage and sequestration capacity for some Portuguese salt marshes and seagrass meadows (Couto *et al.*, 2013; Nuñez, 2015; Sousa *et al.*, 2010; Sousa *et al.*, 2017), but there are no studies attempting to determine the economic value of these ecosystems. Furthermore, Portugal has the biggest salt marsh of Europe at Ria de Aveiro (Sousa *et al.*, 2017) and has an unique coastline considering European seagrass biodiversity (Cunha *et al.*, 2014), which makes this Country an important hotspot of blue carbon and its protection should not be forgotten.

This study presents the first economic valuation of Portuguese mainland coastal carbon ecosystems, aiming to support policy makers decisions about local ecosystems conservation. The research questions that guided this project are:

- What and where are the existing blue carbon ecosystems in Portugal?
- How much carbon is stored in these ecosystems, and how much it is at risk?
- What is the economic value associated to blue carbon in Portugal?
- What economic implications have the ecosystem losses?
- What kind of challenges and opportunities policy makers have to protect coastal wetlands in Portugal?

To answer these research questions, this dissertation is organized in seven sections. Firstly, it is explained the concept of blue carbon, its importance as carbon sink and other services provided by it, and it is also analysed their global threats. Secondly, it is described the importance of economic valuation of ecosystem services and it is made a literature review of the existing studies about economic valuation of blue carbon. Section three is dedicated to the methodology and Portuguese mainland case study and section four and five are completed with the results and discussion, respectively. Section six makes available some recommendations to integrate blue carbon in Portuguese climate policy. Last section gives some conclusions.

I. Blue Carbon Ecosystems

The study developed upon can be contextualized in the carbon cycle. This chapter is dedicated to explaining the different concepts of blue carbon among different authors or institutions, and the economic relevance of the coastal wetlands as blue carbon ecosystems, as well as, its key role on the climate change mitigation.

More than 55% of carbon captured by photosynthetic activity is captured by marine and coastal ecosystems and it is denominated as blue carbon (McLeod *et al.*, 2011; Nelleman *et al.*, 2009). For that process, the main contributors are the coastal vegetated habitats, like mangrove forests, salt marshes and seagrass meadows. Those habitats are identified as the most productive habitats worldwide (Nelleman *et al.*, 2009) and they are also considered to hold the greatest Greenhouse Gases (GHG) mitigation potential (Murray *et al.*, 2011).

These ecosystems remove carbon dioxide (CO_2) from the atmosphere photosynthetically and only a little percentage returns back again to the atmosphere through breathing. The remaining carbon is stored in ecosystems living biomass (both above- and below-ground) and in its soil organic carbon, but virtually all the sequestration carbon ends up buried in sediments, remaining stored for millennia if they are not destroyed (Murray *et al.*, 2011; Nelleman *et al.*, 2009).

Some studies (Howard *et al.*, 2017; Wylie *et al.*, 2016) highlight that although the blue carbon ecosystems occupy only 2% of global area, they are more effective sequestering and storing carbon in their soil and biomass than terrestrial forests. Annual average sequestration rates for salt marshes and mangrove were estimated between 6 and 8 $tCO_2e\ ha^{-1}\ y^{-1}$, while seagrasses carbon sequestration rate is lower, approximately 4 $tCO_2e\ ha^{-1}\ y^{-1}$. Even though, these rates are about two to four times larger than the rates observed in mature tropical forests that arounds 1.8-2.7 $tCO_2e\ ha^{-1}\ y^{-1}$ (Murray *et al.*, 2011).

Most of the carbon is stored in sediments, which also can be called as soil organic carbon, and this represents the unique possibility of long-term carbon sink. Its storage capacity depends on the type of ecosystem considered: an average of 500 t $CO_2e\ ha^{-1}$ for seagrasses, 917 t $CO_2e\ ha^{-1}$ for salt marshes, 1.060 t $CO_2e\ ha^{-1}$ for estuarine mangroves, and approximately 1.800 t $CO_2e\ ha^{-1}$ for oceanic mangroves (Murray et al., 2011). The main reasons for these differences are related with biomass production of each ecosystem and their capacity of produce surplus organic carbon (Nelleman *et al.*, 2009).

This great capacity to store amounts of carbon for longer periods than other ecosystems makes coastal vegetated habitats strong and efficient natural carbon sinks (Nelleman *et al.*, 2009). Besides this important service, these ecosystems provide a range of services with high importance for human well-being and economic development (Nelleman *et al.*, 2009; Russi *et al.*, 2013).

They are nursery habitats for fish, crustaceans, birds and marine mammals, as well as offers a sheltered living space for them, being a hotspots of biodiversity (Murray *et al.*, 2011; Nelleman *et al.*, 2009). Coastal wetlands have also a significant role in resilience capacity considering the climate change through their ability to coastal protection and erosion control (Murray *et al.*, 2011; Russi *et al.*, 2013). They are also capable to improve water quality since they uptake nutrients and contaminants preventing eutrophication, that is the consequence of excess of nutrients in water masses (Russi *et al.*, 2013). Furthermore, they are sources of raw materials and food and an unique and aesthetic landscape as well (Russi *et al.*, 2013). Table 1 summarizes some the services provided by coastal vegetated ecosystems.

Table 1. Services provided by coastal vegetated ecosystems

Provisioning services	Regulating services	Cultural Services	Supporting Services
Source of food Raw materials provision Genetic material Water supply	Carbon sequestration Water purification Flood protection Erosion control Coastal protection Climate regulation	Cultural heritage, spiritual and religious benefits Tourism, recreation, education and research Aesthetic	Primary production Nutrient cycling Nursery and habitat for fishes and other marine species

Adapted from: Mehvar et al. (2018); Russi et al. (2013)

Despite all the virtues of blue carbon ecosystem described above, they are among one of the most threatened natural environments worldwide, with alarming loss rates in recent years. It is estimated one third of the total lost had happened over the past decades and the remaining are identified as threatened (Nelleman *et al.*, 2009; Wylie *et al.*, 2016). It is estimated that the global loss rates are around 2 to 15 times faster than that of tropical forests, about annual loss between 0,7% and 3% (Howard *et al.*, 2017).

This loss is a consequence of human activities through the conversion and degradation of the land due mainly to coastal development: urban development, including for tourism and harbour infrastructure and for agriculture and aquaculture. Other pressures on these ecosystems that lead to their degradation is wood harvesting, overfishing, marine operations, water quality degradation and mechanical damage (such as dredging, trawling and anchoring).

Sea level rise (SLR) and coastal erosion also play a role in this loss (Murray *et al.*, 2011; Nelleman *et al.*, 2009). On other hand, blue carbon ecosystems can contribute to climate change itself, if they were degraded. Unfortunately, these ecosystems could shift from net sinks to sources of carbon releasing back into the atmosphere and/or into the ocean a large amount of carbon they are storing (Howard *et al.*, 2017; McLeod *et al.*, 2011). This happens when ecosystems are degraded or converted to other land uses, and the sediment carbon is degraded or exposed to oxygen, hence increase microbial activity (Pendleton *et al.*, 2012). It is projected that these ecosystems are releasing between 0,23 and 2,24 billion Mg of CO_2 *per* year (Howard *et al.*, 2017).

Healthy and productive coastal ecosystems have a key role in terms of biodiversity, in number and quality of services provided and in the mitigation of the climate change effects on coastal communities and economies. Revert current decline trends and recover the lost areas of blue carbon would enhance the ecological status of the global coastal environment and this could result in the recovery of important services such natural carbon sink (Nelleman *et al.*, 2009).

The above-mentioned reasons provide strong arguments for the protection and restoration of these ecosystems. Some authors (Howard *et al.*, 2017; McLeod *et al.*, 2011) argue that should be made an effort to integrate blue carbon into coastal management strategies and take them into GHG inventories and mitigation strategies, recommended by IPCC - International Panel on Climate Change (Howard *et al.*, 2017; McLeod *et al.*, 2011). This way it is guaranteed the carbon sequestering benefits are accounted for, along with all the additional services that are provided by the coastal ecosystems (Howard *et al.*, 2017).

There are some international reports underlining this need. By the 2015, the Paris Agreement refers the importance of insuring the integrity of all ecosystems, including the oceans. Under this Agreement, the Parties should "*conserve and enhance, as appropriate, sinks and reservoirs of greenhouse gases*" (UNFCCC, 2015). It also calls to include them in existing frameworks under United Nations Framework Convention on Climate Change (UNFCCC) and related climate financing mechanisms (Howard *et al.*, 2017). Similarly, the 2030 Agenda for Sustainable

Development highlights the importance "*to sustainably manage and protect marine and costal ecosystems to avoid significant impacts, including by strengthening their resilience, and take action for their restoration in order to achieve healthy and productive oceans*" and "*conserve at least 10 per cent of coastal and marine areas*", in their Sustainable Development Goal 14.

II. ECONOMIC VALUATION OF BLUE CARBON ECOSYSTEMS

Supporting background

The benefits provided by the ecosystems and their biodiversity to human well-being include provisioning, regulating, cultural and supporting services, as categorized by MEA (2005) as shown in Figure 1. The products that people get from ecosystems are defined as providing services as food, fiber, fuel, genetic material. Regulating services offer to people benefits such as air quality and water purification, climate regulation and regulation control, resulting from the regulation of ecosystems processes. The ecosystem services that provide benefits related with spiritual enrichment, recreation and aesthetical experiences are categorized as cultural services. Supporting services are those that support the production of all other ecosystem services, including primary production and soil formation (MEA, 2005).

Ecosystem services have been the subject of an increasing scientific interest and acknowledgement as they have crucial role on humanity and sustainable development (Cullen-Unsworth and Unsworth, 2013; Villa and Bernal, 2017).

Provisioning Services *Products obtained from ecosystems*	**Regulating Services** *Benefits obtained from regulation of ecosystems processes*	**Cultural Services** *Nonmaterial benefits obtained from ecosystems*
• Food • Fresh water • Fuelwood • Fiber • Biochemicals • Genetic resources	• Climate regulation • Disease regulation • Water regulation • Water purification	• Spiritual and religious • Recreation and ecotourism • Aesthetic • Inspirational • Sense of place • Cultural heritage
Supporting Services *Services necessary to produce all other ecosystems services* • Soil formation • Nutrient cycling • Primary production		

Figure 1. Classification of ecosystems services by Millennium Ecosystem Assessment

Adapted from: MEA (2005))

The international reports like *Millennium Ecosystem Assessment* (MEA, 2005) was the first time there was a global assessment of ecosystem services and *The Economics of Ecosystems and*

Biodiversity Initiative (TEEB, 2010), provided economic calculations and made the assessment of ecosystem services a leading subject. Later assessments use slight different organizations of these (e.g. CICES - Common International Classification of Ecosystem Services) and a new concept proposed by Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) is now under discussion but still does not have the consensus of the scientific community. These reports were based on scientific and economic knowledge that recognizes the importance of the benefits provided by ecosystems to people. They also underline the fact that the ecosystems are not being managed in a sustainable way, hence jeopardize human well-being (MEA, 2005).

The growth demand for ecosystem services, mainly provisioning services, has been compromising the ability of ecosystems supply other vital services such as, but not limited to, water and air purification, climate regulation, erosion control and human diseases control (MEA, 2005; TEEB, 2010). This is a result of the trade-off between services to get the most valuable for people, for instance, increase food supply converting a forest to agriculture(MEA, 2005).

There is a lack of knowledge and understanding about the benefits provided by regulating, supporting and cultural services. Furthermore, there is a difficulty in obtaining the value of ecosystem services in monetary terms. This means that these services are not considered into the decision-making process, reflecting a market failure that, consequently, leads to a degradation and destruction of those ecosystems.

Understanding the economic value of the ecosystems services could be an interesting mechanism to overcome this issue (Murray *et al.*, 2011). That was the purpose of MEA (2005) and TEEB (2010) which developed the assessment framework that makes possible to compare different services among each other (MEA, 2005), through the economic valuation.

Before selecting the economic method to value an ecosystem service, it is important to understand that ecosystems and their services have economic value and its measurement is based on attributes such as the utility people and their actual or potential use, in a directly or indirectly way (MEA, 2005).

The Total Economic Value (TEV) is the most extensively framework used to assess the utilitarian value of ecosystems (MEA, 2005; TEEB, 2010). It divides the total value (output) of ecosystems into use- and non-use value categories. Use values are related with the utility that people attribute to consumption or production purposes, while non-use values are those that do not involve direct or indirect use, although humans assign value for knowing their existence. These two types of values are disaggregated into different value components, as it is shown in Figure 2.

The use values can be direct use value that results from direct use of biodiversity, indirect use value derived from the regulation services provided by ecosystems and option value that are related with the importance that people give to the future availability of a given ecosystem (TEEB, 2010). Non-use value are related with existence value, that means the satisfaction derived from the knowledge of future existence of ecosystems.

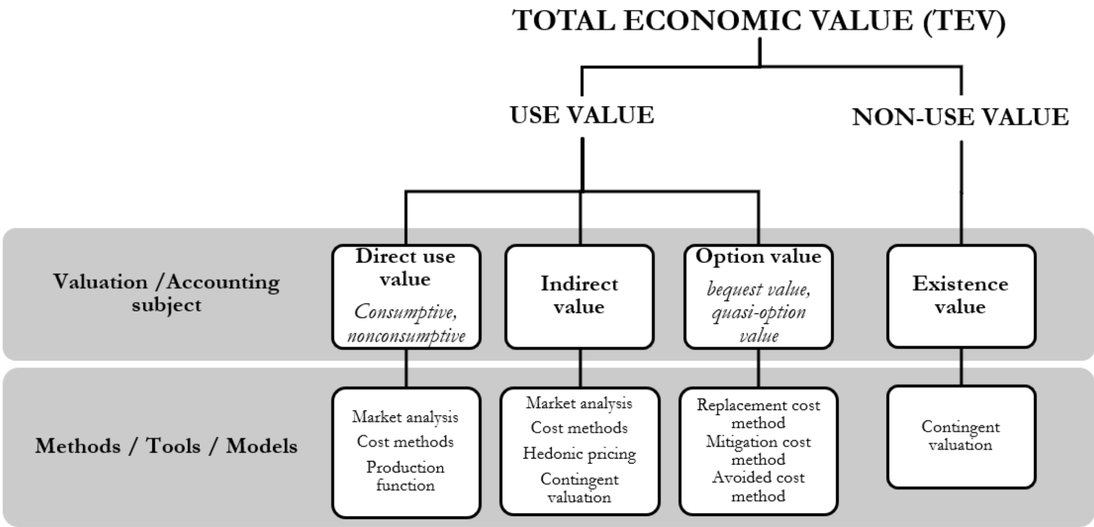

Figure 2. The Total Economic Value Framework.

Adapted from MEA (2005) and TEEB (2010).

As explained before, the economic valuation is the usual method of valuing the ecosystems significance and their services. At the same time, it has become an important method to sustain the protection of ecosystems and biodiversity, helping to promote conservation and sustainable use of natural resources (Villa and Bernal, 2017).

Nevertheless, this economic valuation is not just about money. It also identifies the consequences of ecosystems changes on human welfare, being a strong argument for better policy decisions (Himes-Cornell *et al.*, 2018; Murray *et al.*, 2011). This framework deeply contributes to the need to gather scientific and policy-relevant knowledge regarding the planet's biodiversity and ecosystems services, providing policymakers with accurate information. This idea has been materialized, in 2012, by the development of the IPBES.

Economic valuation methods

A variety of methods are available to assess ecosystem services that have been gathered in MEA (2005) and TEEB (2010) which is summarized in Figure 2. Based in assessment framework established under these reports, they also develop several guidelines to support assessment of different ecosystem services to global and local level. Ecosystem services assessment studies were made as well, despite it was not the first time that was performed this kind of assessment. Previous to MEA, some authors like Costanza *et al.* (1989) had valuated the ecosystems services using environmental valuation technics and ecological pricing methods, highlighting the benefits they provide to humans.

TEEB (2010) have categorized the methods used to value ecosystems services taking account the availability of direct or indirect price information, or the absence of both: (a) direct market valuation approaches, that reflect actual preferences or costs to individuals, through the use of data from actual markets; (b) revealed preference approaches which is based in observation of individual choices; and (c) stated preferences approaches through simulation of markets and demands of ecosystem services by means of surveys. Table 2 synthetize those different methods and their relationship with value types.

The market valuation approach has been identified by Himes-Cornell *et al.* (2018) as the most common method to estimate the economic value of climate regulation services such as carbon sequestration, and for this reason only those are deepened below. Through the market price, it is possible to do an approach to the monetary indicator of the direct uses of the ecosystem services like fish, raw materials, wood, among others. The process is based on the market price of the commodities that are made available from the ecosystem. Simultaneously, this price reflects the well-functioning market preferences and the marginal cost of production (TEEB, 2010).

Another alternative for the market price method is the cost-based approach, which reflects the monetary expenditure incurred by any ecosystem's alteration. It can be achieved through three different perspectives: the avoided cost, the replacement cost and the mitigation or restoration cost. The avoided cost method corresponds to the necessary budget to prevent an ecosystem loss. The replacement cost method measures the cost of replacing an ecosystem service with artificial technologies. At least, the effects induced by the ecosystem services loss and their restoration entails costs that can be accounted by the mitigation or restoration cost method (TEEB, 2010).

The *production function-based* method estimates the contribution value of that an ecosystem service to the income or productivity (TEEB, 2010).

Table 2. Relationship between valuation methods and value types

Approach		Method	Value
Market valuation	Price-based	Market prices	Direct and indirect use
	Cost-based	Avoided cost	Direct and indirect use
		Replacement cost	Direct and indirect use
		Mitigation / Restoration cost	Direct and indirect use
	Production-based	Production function approach	Indirect use
		Factor income	Indirect use
Revealed preference		Travel cost method	Direct (indirect) use
		Hedonic pricing	Direct and indirect use
Stated preference		Contingent Valuation	Use and non-use
		Choice-modelling / Conjoint Analysis	Use and non-use
		Deliberative group valuation	Use and non-use

Source: TEEB (2010)

These methods have some limitations because for most of the ecosystem services it is not possible to take direct prices since they are not marketed outputs (Himes-Cornell *et al.*, 2018). The absence of markets or the existence of distorted markets leads to problems related with data available and accurate needed for these approaches. This results in biased estimations that not provide reliable information to base policy decisions (TEEB, 2010).

Previous studies on economic valuation of carbon sequestration services in blue carbon ecosystems

The focus of this dissertation is the carbon sequestration service of coastal ecosystems. Coastal ecosystems as already pointed out has the most valuable on the planet exceeding US$25,000 billion *per* year (Nelleman *et al.*, 2009). In addition, carbon sequestration service, and its economic assessment is becoming increasingly important to better understand how this regulating service supports human well-being and, at the same time, plays another crucial role on mitigating climate change (Jerath *et al.*, 2016).

In this section, it is presented a literature review considering the purpose of the current dissertation. A few studies have been attempted so far, to assign a value of blue carbon ecosystem. Most of them were performed for developing countries and take mangrove ecosystems as subject of analysis (Camacho-Valdez *et al.*, 2013; Ganguly *et al.*, 2017; Himes-Cornell *et al.*, 2018; Zarate-Barrera and Maldonado, 2015), looking for their distribution in tropical and sub-tropical areas (Murray *et al.*, 2011; Pendleton *et al.*, 2012). A small number of blue carbon ecosystems valuation studies has been conducted for Europe (Beaumont *et al.*, 2014; Canu *et al.*, 2015; Cole and Moksnes, 2016; Luisetti *et al.*, 2013).

The choice of mangroves as study subject is because distribution of this habitat is well reported, including data at the country-scale and the potential of carbon storage is higher than salt marshes and seagrasses (Murray *et al.*, 2011). Mangroves also are one of the most threatened tropical coastal ecosystems. The focus in developing countries are related with the possibility of those countries be included in payment mechanisms from programs such Clean Development Mechanisms (CDM) and Reduction Emissions from Deforestation and Degradation (REDD+), both climate financial mechanisms that support these countries to reduce their emissions levels, while support their development.

Regardless of the ecosystem or area of study, the published studies employ a several valuation methods including benefits transfer, market price, opportunity cost, social cost of carbon and marginal abatement cost.

In a global analysis, Himes-Cornell *et al.* (2018) identifies that the most used method to value blue carbon ecosystems it is *benefits transfer*, with some concerns related with accuracy and validity of data used and the relevance of it for the case study of interest, since this method uses estimated values of other valuation studies and use these estimates to assess the value on another ‘similar’ site. Camacho-Valdez *et al.* (2013) apply this method using the statistical mean of individual estimates of each ecosystem service valuated in different countries to value the ecosystem services of coastal wetlands in Mexico. It is worth to note that coastal ecosystems can have distinct characteristics which results in different ecological outputs depending their geographical location.

Another common method to value blue carbon, reported by Himes-Cornell *et al.* (2018), is the *market price* that uses carbon offset prices from regulated and voluntary markets to estimate the value of carbon sequestration and storage, allowing the financial revenues estimate. These prices vary a lot across countries and markets and there are not a lot of payments made for blue carbon.

For instance, within the European Union Emissions Trading Scheme (EU ETS), over the last year (24/08/2017 to 24/08/2018) the prices of EU Allowances varied from 5,80 to 20,55 €/tCO_2 (EEX, 2018). On the voluntary markets, in 2016, carbon offsets prices varies from less than \$0.5/tCO_2e to more than \$50/tCO_2e (Hamrick and Gallant, 2017).

Opportunity cost approach was applied in cost-benefits analysis by Vázquez-González *et al.* (2017) and Chauhan *et al.* (2017) to analyse the economic viability of conserving mangroves instead of transform these ecosystems into economic activities, in Mexico and India, respectively. The results of Chauhan *et al.* (2017) reveals that the conversion of an ecosystem into an economic activity as rice paddy, increases the carbon equivalent emissions and reduces the net economic value of the same land. On other hand, Vázquez-González *et al.* (2017) concludes that ecosystems conservation is not always economically more attractive than it conversion. This relates with some local policies like subsidies for agriculture that benefits this economic activity.

Although ecosystem protection not always implies a win-win situation, a common figure of these studies (Chauhan *et al.*, 2017; Vázquez-González *et al.*, 2017) is the claim that the conservation of coastal ecosystems to reduce carbon emissions is economically viable.

Himes-Cornell *et al.* (2018) review also identify few studies that use methods that were not included in TEEB (2010) to estimate the value of carbon sequestration services. Those methods are *Social Cost of Carbon* (SCC) and *Marginal Abatement Cost* (MAC).

SCC reflects the economic cost of incremental unit of carbon dioxide (or equivalent amount of other GHG emitted (Nordhaus, 2017), this means, it represents what society is willing to pay today to avoid the future damages caused by one additional ton of carbon emissions (Price *et al.*, 2007). The SCC represents the external societal costs of climate change. This tool is useful to scale the expected economic damages resulting from climate change and to quantify the benefits of a policy to reduce GHG emissions (Prosperity, 2011).

Estimates of SCC are made through Integrated Assessment Models (IAMs), through the application of computer models which combines the best knowledge about how carbon emissions affects on climate system and consequently on society and economy (Carr *et al.*, 2018; Cole and Moksnes, 2016). Some assumptions are made in these models, including the discount rate or the estimates of future losses, that result in a higher or lower SCC. The SCC estimations ranges between $5 to $325 (Himes-Cornell *et al.*, 2018).

The SCC is used in the more recent studies to valuate economically the blue carbon (Canu *et al.*, 2015; Carr *et al.*, 2018; Cole and Moksnes, 2016; Ganguly *et al.*, 2017) on the premise that the loss of these ecosystems will take economic damages associated to the loss of carbon sequestration service. However, it is worth to mention that the selection of the SCC values varies between studies.

Most studies apply a global value of SCC since the damage of emit a ton of carbon have global implications (Cole and Moksnes, 2016; Luisetti *et al.*, 2013). Cole and Moksnes (2016) used an average value of SCC found in literature to valuate *Zostera marina* ecosystems, in Sweden. However, some studies try to apply IAMs models to estimate SCC to a country scale. That is the case of Ganguly *et al.* (2017) that applied the revised DICE Model from Nordhaus (2017) in India, to value the carbon sequestration regulation service delivered by seagrass ecosystems. The study of Canu *et al.* (2015) choose a SCC produced by European Commission in 2008, that ranges 19€ /tCO_2.

MAC represent the cost of reducing a ton of carbon emissions taking account a given target level (Price *et al.*, 2007). The MAC allows that policy-makers know the related costs to achieve a specific emission reduction target (Prosperity, 2011). The MAC curve shows how it increase for each additional unit of emission abated. The MAC curve usually derives from expert judgement, who evaluate the cost and emission reduction potential of individual technologies (Prosperity, 2011).

Beaumont *et al.* (2014) apply MAC approach to value the carbon sequestration and storage in UK coastal habitats. They use the recent CO_2 value appraisal guidance of UK government in order to be relevant for their policy. The UK's government changed for a MAC approach believing that it provides greater reliability that emission reduction targets can be met (Prosperity, 2011).

Table 3. Limitations of Social Cost of Carbon and Marginal Abatement Cost Methods

Social Cost of Carbon	Marginal Abatement Cost
Uncertainty	**Uncertainty**
- Unknown future of climate change damages and technological changes. - Subjectivity to choose inputs to SCC calculations (e.g. discount rate) based on researchers understanding about climate change.	- Unknown future of climate and technological changes.
Equity	**Omissions**
- Trend to benefit present generations, since future damages are highly discount due to future projections uncertainty; and rich people due their increased willingness and ability to pay now to avoid climate damages	- MAC omit factors like implementation and behaviour that influence if whether the technology can reach its reduction potential. - Excludes the social benefits of reducing carbon emissions.
Low Price	**Transparency**
- Trend to lower SCC in recent years due to high discount rates assumed.	- MAC curves depend on experts estimates which underlie assumptions which are not always explained or accessible.

Adapted from: Prosperity (2011)

Both SCC and MAC are valuable tools, yet they have some limitations associated with uncertainties about the future, inequity between developed and developing countries and with future generations, omission of some important factors and transparency about assumptions taken by experts (Table 3). These limitations contribute to different results of SCC and MAC that must be considered when chosen to economic valuation.

SCC was mentioned by Himes-Cornell *et al.* (2018) and Cole and Moksnes (2016) as describing the most appropriate to measure the net economic benefit of the avoided carbon emissions provided by blue carbon ecosystems, since the SCC method translates future damages into present monetary value.

Despite the limitation of these tools, both SCC and MAC could be used together as a complement to support the definition of climate mitigation and adaptation strategies. While SCC is suitable to estimate the benefits of any policy involving the potential reduction of GHG, and the MAC is a useful tool to calculate the costs associated with met an emission reduction target.

Some studies apply both SCC and MAC methods to valuate blue carbon (Jerath *et al.*, 2016; Luisetti *et al.*, 2013). Luisetti *et al.* (2013) use values of SCC from literature and DECC value (from UK government) to estimate economic value of carbon storage service loss in Europe. The study of Jerath *et al.* (2016) recurs to both SCC and MAC to evaluate the carbon legacy in a protected mangroves wetland area in Florida, USA, from two perspectives: the economic valuation of stored carbon and of sequestration carbon. For sequestration carbon service they have used the SCC, which reflects the economic value of the damage associated with incremental increase of carbon emissions. For the carbon assessment stored in biomass and soil they used the MAC, which represents the cost of preserving the existing mangrove forest and the associated stored carbon.

All the studies reinforce the importance of coastal vegetated ecosystems as carbon sinks aiming to encourage their protection and restoration.

III. METHODOLOGY

Background information

This study assessed the two types of blue carbon ecosystems present in the Portuguese mainland: salt marshes and seagrass meadows, which represents, also worldwide, the most relevant coastal blue carbon ecosystems (Nelleman *et al.*, 2009). Salt marshes and seagrass meadows comprise around 68% of Portuguese coastal wetlands, and around 19% of all coastal ecosystems[1].

Salt marshes are important intertidal ecosystems mainly linked to estuaries with a great ecological value. These ecosystems are well distributed in Portugal mainland appearing along the entire coastline, in almost all estuaries and coastal lagoons. They are composed of several vegetated species such as *Sarcocornia fruticosa, Sarcocornia perennis, Halimione portucaloides* and *Spartina maritima*, which are able to sequester and store carbon (Caçador and Duarte, 2012). Despite this, *S. maritima* is a usual plant reported in Portuguese salt marshes studies (Couto *et al.*, 2013; Sousa *et al.*, 2010; Sousa *et al.*, 2017).

In the Portuguese mainland coast, we found three of the four native seagrass species of Europe (they are the *Zostera noltii,* the *Zostera marina* and the *Cymodocea nodosa*), making it an exclusive coastline in terms of European seagrass biodiversity. Accordingly with Cunha *et al.* (2014), the most abundant seagrass specie in Portuguese coast is *Z. noltii* and the most endangered seems to be *Z. marina*.

Although the present legal protection imposed by the European Union (EU) aims to prevent significant losses rates of coastal ecosystems, in the last decades, there has been a reduction of these areas in Portugal due to the increasingly human pressure that had degraded and destroyed both salt marshes and seagrass meadows ecosystems. In Portugal around 60% of population lives near the sea, contributing for the coastal ecosystems' pressure through urbanization, tourism, industrial and agricultural uses and related pollution. In addition, natural processes like SLR and coastal erosion, most of the times associated with climate change, remain as a relevant threat, especially for seagrass meadows that are too sensitive to changes in environmental conditions (Pörtner *et al.*, 2014).

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[1] Own calculations with CORINE Land Cover 2012 database and Cunha *et al* (2014) estimations.

In the case of salt marshes, some of them are vulnerable to SLR only in the worst scenario because marshes accrete vertically and maintain themselves upon the sea level (Pörtner *et al.*, 2014). Almeida *et al.* (2014) have identified some cases of the Portuguese on salt marshes restoration, occasioned by agriculture abandonment, arguing this would be an important mechanism for re-creation of intertidal habitats.

The scenario of seagrass meadows is more pessimistic. Cunha *et al.* (2014) described losses greater than 75% in the last 20 years, emphasizing they have almost disappeared in some locations. As this problem is not exclusive for Portugal, this type of habitat is listed as a threatened and declining habitat under the OSPAR (Convention for the Protection of the Marine Environment of the North-East Atlantic).

The future of seagrass species in Portugal is uncertain, and some authors alert for a pessimistic future scenario associated with high costly restoration efforts (Cunha *et al.*, 2014; Luisetti *et al.*, 2013). Despite this, some restoration efforts were performed in Portugal. In Mondego Estuary, the recovery plan implemented by Water Management Authority which included physical protection and seagrass transplantation resulted in increased *Zostera noltii* meadows (Cunha *et al.*, 2014; Cunha and Serrão, 2011). Unfortunately, the results of the restoration program at Portinho da Arrábida was not successful as well due to winter storms which cover restoration sites with sand and coastal debris (Cunha and Serrão, 2011).

The continued implementation of measures of protection, recovery and restoration promoted by EU Environmental Directives will minimize future losses of both salt marshes and seagrasses.

Blue carbon area distribution, trends and drivers of change

There is a decreasing area on both salt marshes and seagrasses ecosystems in Portugal mainland in the last decades (Caetano and Marcelino, 2017; Cunha *et al.*, 2014). In salt marshes the main losses are related with land conversion, for agriculture, industry and tourism purposes as well as to coastal erosion (Airoldi and Beck, 2007; Caetano and Marcelino, 2017). The decline of seagrasses is mainly due to infrastructure development (such as marinas and ports), dredging operations, water degradation and natural phenomena's like winter storms (Cunha *et al.*, 2014). Table 4 summarizes the time evolution of these ecosystems area by NUT II[2] regions.

Table 4. Portuguese areas (hectares) of the two coastal ecosystems by NUT II Regions

	Area (ha)	Year				
		1985	2000	2006	2012	2060(4)
Salt marshes (1)	Norte	430,79	438,51	648,16	608,54	569,15
	Centro	6.310,86	6.671,37	6.575,96	6.563,48	5.786,46
	AML (1)	2.068,29	1.971,37	2.273,46	2.327,1	2.002,32
	Alentejo	1.732,18	1.714,54	1.429,59	1.439,59	1.266,84
	Algarve	8.641,96	7.764,83	7.061,81	7.053,91	6.214,79
	Mainland	19.184,07	18.560,27	17.998,98	17.992,69	15.838,91
Seagrasses (2)	Norte	0	0	0	0	0
	Centro	n.d.	n.d.	n.d.	57,04	0
	AML	n.d.	n.d.	n.d.	54,41	0
	Alentejo	n.d.	n.d.	n.d.	7,67	0
	Algarve	n.d.	n.d.	n.d.	1.566,82	0
	Mainland	n.d.	n.d.	n.d.	1.685,95	0

(1) Own calculations based on CORINE Land Cover 1990, 2000, 2006 e 2012.
(2) Adapted from Cunha et al, 2014
(3) Área Metropolitana de Lisboa
(4) Predicted under pessimist scenario; for more details, please read section "scenario analysis"
n.d. – no data available

In order to identify Portuguese mainland salt marshes areas distribution, it has been made own calculations recurring to CORINE land cover (CLC) maps for the years of 1985

[2] NUTS classification (Nomenclature of Territorial Units for Statistics) is a hierarchical system for dividing up the economic territory for statistical purposes, developed and regulated by the European Union. Under level II Portugal mainland is divided into five regions: Norte, Centro, Área Metropolina de Lisboa (AML), Alentejo and Algarve.

(CLC90), 2000 (CLC00), 2006 (CLC06) and 2012 (CLC12), provided by the European Environment Agency (see Appendix 1 for more information about own calculations). The current extent of salt marshes, based in the most recent available data of 2012, at Portugal mainland is estimated in 17.992 ha (Table 4), and are geographically located as shown in Figure 3. Algarve region have the major area of salt marshes, representing 39% of the total area, while North region have the lowest one, corresponding to 3,4% of the total area. Sousa *et al.* (2017) have pointed out that in the Centro region, more precisely in Aveiro, can be found the largest continuous salt marshes in Europe, with an estimated area of 4.400 ha for the year 2010.

Since 1985 until 2012, Portugal mainland lost about 6% of salt marshes. The greatest losses occurred between 1985 and 2006 mainly due to land conversion to infrastructure development, to industry and to salt explorations (Caetano and Marcelino, 2017). By region and for the period of 1985-2012, Algarve records the largest lost. In accordance to this fact, Airoldi and Beck (2007) had previously reported losses of 70% of salt marshes before 1988. This suggests that salt marshes area in Portugal mainland could be larger than reported by CLC databases, once that there is no information about the period that these losses occurred. These loss rates place this region as the most vulnerable.

Figure 3. Salt marshes geographical location in Portugal Continental

Source: Gonçalves (2016)

In some regions like Norte, Centro and Área Metropolitana de Lisboa (AML), salt marshes' area presents an increasing for the period under consideration. These could be the result of the correct implementation of ecosystems' protection measures within European legislation (such as Ramsar Convention, Rede Natura 2000, Convention on Biological Diversity and Habitats Directive). Even though smaller areas continued to be lost to agriculture and ports infrastructures according to Caetano and Marcelino (2017). Dredging activity has also been cited by Sousa *et al.* (2017) as one of the main threats of these habitats on these days, since it is a support economic activity to maintain the navigability and the harbor activities.

Seagrasses cover estimations of the Portuguese mainland are much more complex to get, since there is no accurate information of their distribution at the country or regional level other than the year 2010. The information included in Table 4 extracted from the first extensive survey about this habitat in Portugal, published by Cunha *et al.* (2014). These authors stressed twenty-one places based on previous records of present or past seagrasses presence. Figure 4 shows the location of those seagrasses areas along the Portuguese mainland coastline. This fact can underestimate seagrasses cover area since there could be other places where seagrasses exists but are not included in this list. Furthermore, even in areas considered by Cunha *et al.* (2014), it was not possible to measure the total seagrasses areas by their hard access.

Despite the previous cited difficulties, as this is the only available data for Portugal about seagrass cover, Cunha *et al.* (2014) inventory was used as data source for the purpose of this dissertation and included in Table 4. They estimated a coverage of 1.574 ha of *Zostera noltii*, 7,5 ha of *Zostera marina* and 109 ha of *Cymodosa nodosa*, totalizing 1.686 ha of seagrass meadows. Algarve region is the largest seagrass area in Portugal mainland holding for 93% of the whole mainland seagrass coverage.

The declining rates of seagrass habitats in Portugal has been dramatic in the last 20 years, with reported losses over 75%, having completely disappeared from some places, registered particularly for *Z. marina* specie, as the cases of Ria de Aveiro, Portinho da Arrábida, Ponta de Adoche, Costa da Galé and Mira River (Cunha *et al.*, 2014).

Even though Cunha *et al.* (2014) have reported some historical records of seagrasses in Portugal, there is no historical data available for all the places studied, hence the historical data of seagrass meadows distribution was not considered in this work. Nevertheless, in this study was found an annual loss about 25ha *per* year in Ria de Aveiro, between 1985-2006, related with traditional activity of collecting "moliço", a mixture of aquatic plants that included mostly *Z. marina* and *Z. noltii*.

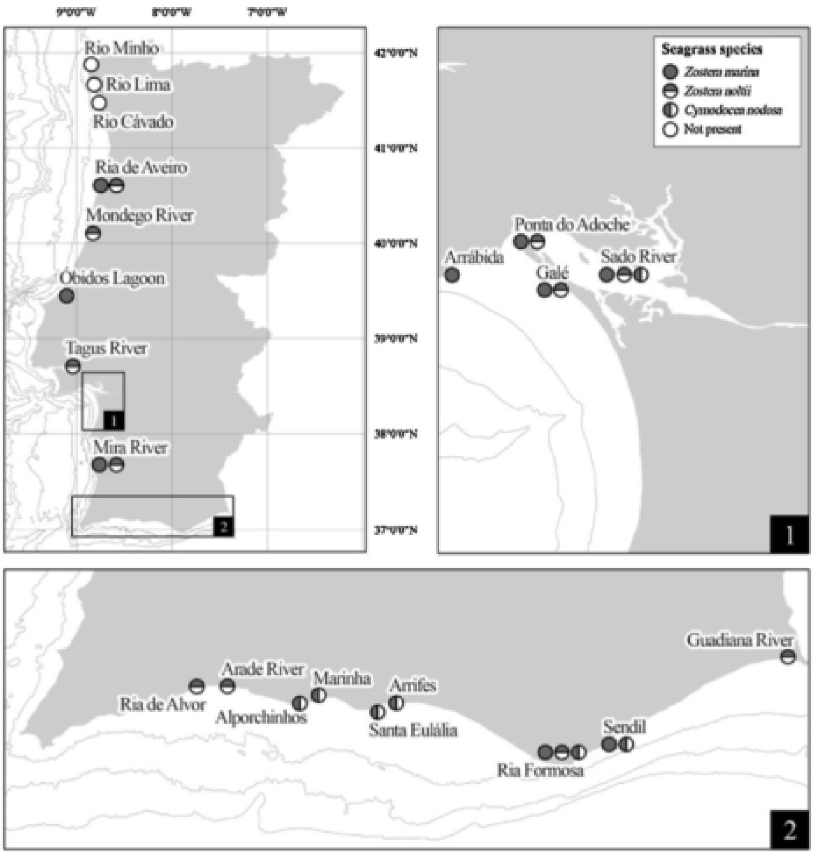

Figure 4. Seagrass distribution in the Portuguese coast

Source: Cunha et al. (2014)

Carbon storage capacity

Data on carbon sequestration and storage capacity from blue carbon ecosystems in Portugal mainland are not easily available and have been collected just for a few locations: Ria de Aveiro (Sousa *et al.*, 2017), Mondego Estuary (Couto *et al.*, 2013; Sousa *et al.*, 2010), Tagus Estuary (Sousa *et al.*, 2010) and Ria Formosa (Nuñez, 2015). Therefore, a series of assumptions have been made to analyze carbon storage capacity of the Portuguese mainland salt marshes and seagrasses.

The analyzes of the carbon storage capacity of the blue carbon ecosystems in Portugal mainland is made taking in account the carbon accumulation rate (CAR) in the sediments of the salt marshes and the seagrass ecosystems. The CAR of each vegetated specie depends of several factors. The main drivers that contribute to different carbon accumulation rates in salt marshes species are primary production on belowground, salt marsh maturity, soil salinity, tidal inundation, temperature, nutrient availability and sediment type (Sousa *et al.*, 2010; Sousa *et al.*, 2017). In seagrass meadows the carbon storage capacity essentially depends on abiotic factors like water turbidity, sheltered and shallow environments; and factors related with anthropogenic pressure, like habitat fragmentation and eutrophication (Mazarrasa *et al.*, 2018).

Salt marshes species varies accordingly with locations. *Spartina maritima* appears to be the most usual specie identified in different Portuguese salt marshes' studies like Caçador and Duarte (2011), Couto *et al.* (2013) and Sousa *et al.* (2017). *Spartina maritima* is a low marsh (tidal marsh zone located below the mean highwater mark) pioneer halophyte, and because of that fact it only represents a part of the total salt marsh area. Besides that fact, for this study, *S. maritima* is considered as being the only specie in whole salt marsh areas, since there is a lack of information about area distribution of other species that compounds this ecosystem.

Table 5. Spartina maritima carbon accumulation rates (gC m^{-2} y^{-1})

Site	CAR (gC m^{-2} y^{-1})	Reference
Ria de Aveiro	120	Sousa *et al.* (2017)
Mondego Estuary	218	Sousa *et al.* (2010)
Tagus Estuary	330	Sousa *et al.* (2010)
	750	Sousa *et al.* (2010)
Ria Formosa	131,8	Nuñez (2015)
CAR Mean value	**309,96**	

Table 6 shows a compilation of the CAR values of *S. maritima* for Portuguese locations included in some studies. Its CAR ranges between a minimum value of 120 gC $m^{-2}y^{-1}$, reported by Sousa *et al.* (2017) and a maximum value of 750 gC m^{-2} y^{-1} cited in Sousa *et al.* (2010). A mean value of 309,96 gC m^{-2} y^{-1}, based on the cited literature for some Portuguese areas in Table 6, is considered to in the calculations to estimate carbon storage capacity of Portuguese salt marshes.

For seagrass meadows, the carbon accumulation rate of *Zostera noltii* in Portugal is estimated by Nuñez (2015) to be near 83,9 gC m^{-2} y^{-1}. This value is considered to estimate carbon storage capacity of this ecosystem, since no comparable data for *Zostera marina* and *Cymodosa nodosa* was found in literature.

In addition, the calculations performed also assume that *S. maritima* being considered the only specie in whole Portuguese salt marshes and in seagrass meadows only exists *Z. noltii* specie. In both ecosystems these species cohabit with others able to sequester and storage carbon, with lower or higher CAR (Sousa *et al.*, 2017). For instance, *Juncus maritimus* has a carbon storage capacity higher than *S. maritima* (Sousa *et al.*, 2017). In the other hand, despite being the most abundant specie in Portuguese seagrass meadows, *Z. noltii* has a lower carbon sequestration and storage capacity than other species like *Z. marina* (Luisetti *et al.*, 2013).

Moreover, seagrass meadows coverage only reflects twenty-one sites where Cunha *et al.* (2014) made their survey. This inventory may not represent the global area of seagrasses in Portugal. Since there is no widely information about seagrasses coverage, it is possible that there is more seagrass meadows area than that reported by Cunha *et al.* (2014).

Thus it is important to note that these assumptions could result in under- or over-estimated carbon storage capacities due the occurrence of other species with lower or higher CAR (Luisetti *et al.*, 2013; Sousa *et al.*, 2017).

The previous estimation values that are considered to develop this study are converted in CO_2 equivalents, resulting in following values: 11,38 tCO_2 ha^{-1} y^{-1} for salt marshes and 3,08 tCO_2 ha^{-1} y^{-1} for seagrasses. The conversion to CO_2 equivalent was made multiplying gC $m^{-2}y^{-1}$ values to the ratio of molar masses: 3,67 or 44/12.

Scenario analysis

Future losses of salt marshes and seagrass meadows are uncertain in Portugal. Recent studies with future trends to our country were not found. Hence the analysis of changes in value of the carbon storage service provided by coastal habitats between 2010-2060 is made under two hypothetical scenario analyses.

Important information about biophysical processes is important to better understand changes in value. Vegetated coastal ecosystem may store carbon in living biomass (above- and below ground) and in the sediments. When changes occur in the habitats the carbon stored in living biomass and surface sediments is released into the atmosphere. Based on Luisetti *et al.* (2013), a 90% carbon loss is assumed in this study, considered simultaneously appropriated for salt marshes and seagrasses, given the sea level rise.

In the *optimist* scenario it is assumed a small future loss of salt marshes: around 4,5% over 20 years extrapolated forward to 2060 (Beaumont *et al.*, 2014; Luisetti *et al.*, 2013). For seagrasses only 10% of current extent was considered as loss.

In the *pessimistic* scenario, for salt marshes, it is assumed a larger total future loss: around 6% over 20 years extrapolated forward to 2060 (Luisetti *et al.*, 2013). For seagrasses, it is supposed a complete loss of this ecosystem for the year 2060.

These two scenarios are constructed bearing in mind that seagrasses is more vulnerable to climate changes effects than salt marshes, as supported by Wong *et al.* (2014). Table 7 resumes the collected data to be used latter on to obtain the monetary value of ecosystems carbon storage capacity loss.

Table 6. Data to obtain monetary value of ecosystems carbon storage capcity loss

Scenario	Area lost (ha y^1)	90% of C storage service ($tCO_2\ y^{-1}$)
Optimistic scenario		
Salt marshes	-44,98	460,70
Seagrasses	-3,37	9,53
Pessimistic scenario		
Salt marshes	-59,98	614,27
Seagrasses	-33,72	-93,47

Economic valuation

Valuing carbon sequestration and storage service of the blue carbon ecosystems is crucial as it allows to better understand the associated benefits of these ecosystems and to better support decisions about different development options for marine and coastal management.

The economic value of the Portuguese blue carbon ecosystems is assessed using the estimated areas distribution of each ecosystem considered and their carbon storage capacity. It is calculated the value of current total carbon stock in Portuguese salt marshes and seagrass meadows as well as the predicted ecosystem changes over a 50-year period (2010-2060) under two scenarios previously described.

A price approach is used in this study, following previous studies published on the topic of economic valuation of carbon sequestration and storage of coastal blue carbon (Beaumont *et al.*, 2014; Jerath *et al.*, 2016; Luisetti *et al.*, 2013), in compliance with this method description at section II.

In order to calculate the economic value of the current carbon stock, it is selected a mean price of the European carbon market. The EU Allowance (EUA) is the tradable unit of the European Union Emissions Trading Scheme (EU ETS) and its value depends on daily auctioning process. According to the European Energy Exchange (EEX), over the last year[3], the prices of EUA have fluctuated from 5,80 to 20,55 €/tCO_2 (EEX, 2018). A mean price of 11,19 €/tCO_2 is considered for the aim of this study.

The blue carbon ecosystems changes are evaluated in this study by two different approaches: the social cost of carbon (SCC) and the marginal abatement cost (MAC).

The first considers the change in the discounted value of economic welfare from an additional unit of CO_2 emissions (Nordhaus, 2017). The application of the SCC assumes that the loss of one hectare of ecosystem will no longer provide the carbon sequestration service leading to an economic damage. The SCC calculation integrates with some uncertainties as the usual models used to get it fails to consider significant risks and costs related to climate change, biodiversity losses, labour productivity impacts, among others (Stiglitz *et al.*, 2017). All these limitations anticipate the final estimates for SCC result being underestimated. In

[3] Last year considers the period of 24-08-2017 to 24-08-2018. For more information about EUA prices.

the literature, it can be found different SCC value, that ranges between \$5 and \$312 /tC. Luisetti *et al.* (2013) have done an exhaustive compilation about the proposed values from the specialized literature. For the purpose of this study it is selected the SCC estimation produced by the European Commission in 2008 and reported by Canu *et al.* (2015), that corresponds to 19 €/tCO_2 for the year 2013.

The marginal abatement cost (MAC) approach reflects the costs to meet a specific reduction target, through the maintaining and/or reducing of carbon emissions or the loss from storage (Jerath *et al.*, 2016). Bearing in mind the temperature achievements under the Paris Agreement, Stiglitz *et al.* (2017) have predicted the time evolution for carbon-prices that are within the interval of US\$40-80/tCO_2 by 2020 and within the interval of US\$50-100/tCO_2 by 2030. These values have been considered to this study for two main reasons: they are based on evidence through industry and policy experience and they are the most recent data available for this topic.

The selected MAC prices are converted to Euros at the conversion rates of US\$ 1 = 0,86 €[4] resulting in ranges between 34,41-68,82 €/tCO_2 by 2020 and in ranges between 43,01-86,03 €/tCO_2 by 2030.

A discounted rate of 3.5% is chosen to get the present value of the carbon storage through the selected period. This rate results from the combination of similar values found in other carbon storage works like in Luisetti *et al.* (2013), and the actual predicted value of the interest rate for a time frame of 50 years.

[4] XE Currency Converter (xe.com) – Last updated 2018-09-06 22:24UTC

IV. Results and Discussion

Blue carbon storage capacity in Portugal

The current area (year of 2012) of the Portuguese mainland salt marshes and seagrasses has a carbon storage capacity of 204.757 tCO_2 and 5.193 tCO_2, respectively. Changes in the distribution area as presented in Table 4 have direct implications on carbon storage capacity. Figure 5 shows the estimations for the changes in carbon storage capacity area across time due to the predicted decrease of blue carbon ecosystems in Portugal mainland under *pessimistic* scenario. It can be seen a decreasing curve for the carbon storage capacity of the two types of ecosystems considered, based on the assumption that these ecosystems are increasingly coming under pressure. For the salt marshes curve there is a strong decreasing for the twenty first years that slows down between 2006 and 2012, presenting a huge decreasing for the following 48 years. For the seagrass, the loss of carbon storage capacity it is only considered between 2012-2060, where a decrease of 5.000 tCO_2 is estimated, since there is no historical data as explained in section III and Table 4.

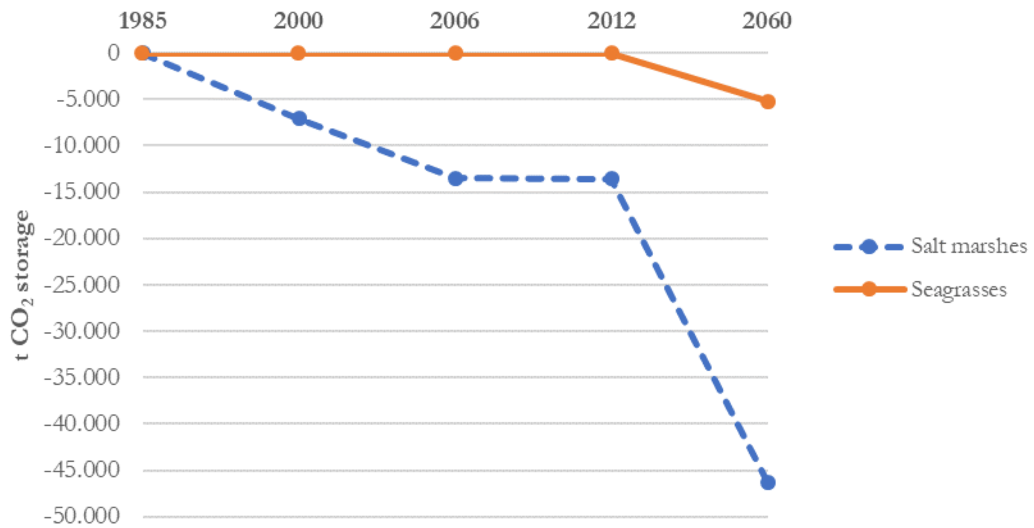

Figure 5. Estimated changes in CO_2 storage capacity provided by Portuguese blue carbon ecosystems for the 1985-2060.

Source: own estimations

The total loss on storage capacity under the *pessimistic* scenario is equivalent to 51.451 tCO_2 between 1985 and 2060. Under the *optimistic* scenario there is a lower loss, around 38.587 tCO_2. These calculations have assumed that the carbon accumulation rate remains constant for the same location over time. Consequently, the real carbon storage capacity of the blue carbon ecosystems in Portugal mainland may be higher than those calculated previously (recall section III about the consequences of the supposition taken).

These major limitations to get these calculations has been the lack of available country and local data. A recommendation should be left to the scientific community and policy institutions responsible for these issues, in order to make an effort in gathering accurate information about coastal vegetated ecosystems.

Economic value of current blue carbon stock

The estimated value of the current blue carbon stock for Portugal mainland regarding EUA prices amounts to 2.349.335€, of which approximately 2.291.228€ are attributed to salt marshes. The major weight attributed to salt marshes carbon stocks can be easily explained. On the other hand, the estimated area of salt marshes in Portugal mainland is larger than the extent of seagrass meadows. In addition, the estimated CAR of salt marshes is also higher than CAR of seagrasses (Nuñez, 2015).

It could be stated the total estimated value of the current Portuguese mainland for blue carbon corresponds to 2%[5] of the European total value estimated by Luisetti *et al.* (2013). Considering only the salt marshes, the blue carbon value of the Portuguese mainland comprises 24% of the European value, while the estimated value of seagrasses blue carbon only corresponds to 0,04% of the value estimated by Luisetti *et al.* (2013) to this ecosystem. This roughly comparison, gives a preliminary idea of Portugal's importance in the European discussion of these affairs. Also, to note that Azores and Madeira Archipelagos, that corresponds to the half of the total Portuguese territory coastline, have not been taking into account in this study.

[5] US$ value from Luisetti *et al.* (2013) was converted to Euros at the same conversion rate used in this study.

The estimated values of the Portuguese mainland blue carbon ecosystems when compared with European study conducted by Luisetti *et al.* (2013) are highly different due to the fact of the estimated value of seagrass blue carbon in Europe includes *Posidonia oceanica* specie and this do not. This specie is the most abundant in Europe as well as it is the most able to capture CO_2 (Luisetti *et al.*, 2013), but it does not exist in Portuguese seagrass meadows (Cunha *et al.*, 2014).

In terms of *per* hectare carbon value, the results of the Portuguese mainland salt marshes and seagrasses are 127 €/ha and 34 €/ha, respectively. Comparing with the Luisetti *et al.* (2013)'s study, the Portuguese values are higher in salt marshes and lower in seagrasses meadows than the European values for the same ecosystems (29 €/ha and 606 €/ha, respectively).

The conclusions of the proposed comparison must be interpreted with some caution. There are major differences in the assumptions underlying carbon estimations from the two studies, mostly related to wide variation on ecological processes and data selection. The difference on prices considered also may influence the carbon value estimations.

In order to do a sensitive analysis of these distinctive prices undertaken for the two studies, an illustration exercise is proposed. If we apply the same EUA price assumed in this study in estimations of carbon storage capacity made by Luisetti *et al.* (2013) for seagrasses ecosystems, contempling only the *Zostera marina* seagrass specie existing in Portugal, the results would be substantially diverse. In this hypothetic case, the statistical weight of the Portuguese mainland under European blue carbon value would decrease to 18% in salt marshes and it would increase to 1,14% in seagrass meadows. The *per* hectare carbon values would result in lower values for both European ecosystems: 38 €/ha to salt marshes and 21 €/ha to *Z. marina* seagrass specie. It is clear that the results of this study would have been lower values if it was considered the same EUA price of Luisetti *et al.* (2013), or if the economic valuation had been based on the international voluntary market prices (recall section II for more detailed information).

Economic value of blue carbon ecosystem changes

The economic value resulting from the loss of the carbon storage service in the Portuguese mainland blue carbon ecosystems is assessed using SCC and MAC prices as described in methodology. The results are summarized in Table 8.

Under the *optimistic scenario* the economic losses in 2060 of the Portuguese mainland coastal blue carbon for the two ecosystems ranges between 218.412€ at SCC prices and 810.180€ at high MAC prices. Under the *pessimistic scenario* the economic loss is than the pessimistic scenario: it ranges between 328.856€ at SCC prices and 1.219.863€ at high MAC prices.

These results reflect the economic losses for society under two scenarios based on expert judgement projections of coastal erosion and sea level rise (Beaumont *et al.*, 2014; Luisetti *et al.*, 2013). Nevertheless, they do not consider other ecosystem losses resulting from land conversion, since Portugal is under European legislation protection. However, the difference in value could be greater if Portugal fails on the protection and conservation action of the coastal blue carbon ecosystems.

Table 7. Economic value (€) of the blue carbon storage loss in the Portuguese mainland in the optimistic and pessimistic scenario over 50-years period (2010-2060), at SCC prices and MAC prices.

Scenario	Ecosystem	Price (€/tCO_2)	Annual loss area (ha/y)	90% of C storage service in tCO_2/y	Blue carbon storage loss value by 2060 (€)
Optimistic scenario	*Salt marshes*	SCC	-44,87	460,70	-214.068,65
		MAC (lower)			-397.034,84
		MAC (higher)			-794.069,68
	Seagrass meadows	SCC	-3,37	9,35	-4.343,10
		MAC (lower)			-8.055,19
		MAC (higher)			-16.110,38
Pessimistic scenario	*Salt marshes*	SCC	-59,98	614,27	-285.424,87
		MAC (lower)			-529.379,79
		MAC (higher)			-1.058759,58
	Seagrass meadows	SCC	-33,72	93,47	-43.431,03
		MAC (lower)			-80.551,88
		MAC (higher)			-161.103,77

Furthermore, these results only include the value of carbon sequestration and storage service as a measure for the economic loss associated with these two ecosystems. However, as described in section I, the coastal blue carbon ecosystems provide an infinite number of life-sustaining services with benefits to the whole society. This means that real economic loss from the reduction area of these ecosystems for the society is much higher that reported by the calculated values.

For instance, Cunha *et al.* (2014) pointed out the decline of seagrasses in the Portuguese coast is promoting the loss of high rates of marine biodiversity, enforcing coastal fisheries depletion, the decrease of coastal water quality and the loss of valuable natural resources such as beach sand.

Both scenarios reflect losses related with SLR and coastal erosion, and it does not account for future changes related with land reclamation to other activities. The estimated values could be greater if protection measures fail.

This study expects to raise awareness on the importance of the blue carbon ecosystems for the Portuguese economy and the nature protection. The Portuguese strategy to face the climate change should pay attention to these particular ecosystems which needs political will and effective actions to stop its damage.

V. Challenges of Blue Carbon Management in Portugal

Blue carbon ecosystems are important natural carbon sinks which are recognized as the most efficient and cost-effective way to counteract GHG emissions, while providing other valuable ecosystem services. Nevertheless, these ecosystems are declining globally.

The state of blue carbon ecosystems in Portugal mainland follows the same trend as evidenced by the results in this study. The Portuguese mainland salt marshes and seagrass meadows have experienced historical area losses and consequently in their carbon sequestration and storage capacity. If Portugal fails to protect these ecosystems there will be future losses due to natural events (such as coastal erosion and sea level rise) as also demonstrated per this study.

Despite the evidence to the carbon sequestration and storage capacity of vegetated coastal ecosystems and their economic value as reported in this study, these ecosystems are still not considered as carbon removals of GHG in the Portuguese National Inventory Report (NIR), that is submitted under the UNFCCC and the Kyoto Protocol, by Portuguese Environmental Agency (APA – *Agência Portuguesa do Ambiente*). Nevertheless, these ecosystems are included in wetland category as carbon emitters (APA, 2018).

The data availability about this issues is stills limited around the world and for this reason coastal ecosystems are not yet included in National GHG Inventories of the majority countries (Villa and Bernal, 2017) as well. So, it could be suggested to the Portuguese and European authorities responsible for these matters the urgency in supporting any initiative coming from private or public institutions that measure and collect information about carbon potential at a national, regional and local level. Also propose them to be the great promoters of this challenge worldwide in future discussions about the topic.

Beyond research, it is crucial the implementation of a legal framework that includes restoration of coastal blue carbon ecosystems and best-management practices to protect them, supporting its preservation and enhancing of blue carbon stocks as a tool to mitigate CO_2 emissions and, hence climate change. This would surely help Portugal to be compliant with the climate international agreements while recovering valuable ecosystems services and key natural resources.

Portugal is committed with European Directives since 1980 that encourages both salt marshes and seagrass species protection. However, as concluded in this study, the declining trend stills happening due mainly to the influence of anthropogenic pressures.

In addition to rebuild natural carbon sinks, there is scientific evidence that reversing decline of blue carbon ecosystems and recovering the lost area would also provide an improvement of the ecological status of coastal environments (Nelleman *et al.*, 2009). Seagrasses play a vital role in this issue and have been recognized as important indicator of a good ecological status under the Water Framework Directive (Cunha *et al.*, 2014).

Both, salt marshes and seagrass meadows restoration are possible, and some actions had already been put in place in Portugal as described in section III. However, restoring seagrass meadows it is pointed out as one of the most expensive procedures due the labour required to insert transplants under the water (Nelleman *et al.*, 2009). Nevertheless, the restoration of blue carbon ecosystem appears to be a viable option and should be used strategically by coastal managers and must be assisted with parallel actions that minimize the pressures that caused the previously loss.

Despite the scientific recognition of blue carbon ecosystems importance there is an imperative need to share this awareness with the society. The general literacy that coastal users, managers, politicians, environmental groups, and the general public have about status and benefits associated to vegetated coastal habitats still scarce (Cunha *et al.*, 2014; Nelleman *et al.*, 2009). An effort of dissemination of the research findings could promote to invert the actual ecosystems' losses.

CONCLUSIONS

The aim of this dissertation is evaluating the Portuguese mainland blue carbon ecosystems in terms of their carbon sequestration and storage capacity and simultaneously identifying some challenges and opportunities for Portuguese coastal habitats. The results stated that services provided by the coastal ecosystems have a great economic as well as natural value in terms of carbon sequestration and storage service. It also highlights that any disturbance in blue carbon ecosystems reduce significantly their carbon storage capacity.

The economic value of the current blue carbon stock was estimated in 2.349.335€. The scenario analysis reveals that if the *pessimistic* scenario occurs Portugal could have an economic loss of at least 328.856€ (SCC prices) but it can reach to 1.219.863€ (3,5% discount rate; high MAC prices), in 2060. If future ecosystem loss is greater than what was predict in this study, the economic loss would increase significantly.

The economic value of carbon storage estimated in this study is just a component of the total economic value of coastal vegetated ecosystems. Besides their role as natural carbon sinks, the blue carbon ecosystems are suppliers of other vital services. Therefore, the values estimated in this study must be used with caution and in reference to appropriate contexts. Thus, the economic and ecological value of these ecosystems should be considered into decision-making for conservation and can be strategically used to help Portugal be compliant with climate agreements.

APPENDIXES

Appendix 1: Estimation of salt marshes area distribution in Portugal

The estimation of salt marshes area distribution was made using the open source software QGIS Desktop 3.03 and land use maps from CORINE land cover projects.

For these study, the CLC maps from the years 1985 (CLC1990), 2000 (CLC2000), 2006 (CLC2006) and 2012 (CLC2012), provided by European Environmental Agency, are used. For the purpose of this study, only salt marshes category is considered (421).

Administrative divisions and borders were introduced according to the CAOP 2017 official cartography and to the NUTS II divisions.

All the calculations of salt marshes area distribution by NUTS II divisions is performed by QGIS Desktop 3.03.

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