UNIVERSIDADE DE LISBOA

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Ecology and dynamics of Mediterranean saltmarshes in a perspective of habitat management and restoration policies: the cases of Alvor and Arade in Portugal

Diana Neves de Almeida

Orientador(es): Professor Doutor Carlos Silva Neto
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Tese especialmente elaborada para obtenção do grau de Doutor em Geografia, especialidade de Geografia Física

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Summary

The main objective of this thesis is to study the impacts of morphology, structure and flora transformations on the former reclaimed saltmarshes of Alvor and Arade (Algarve, Portugal) and to understand how these changes can be connected with ecological restoration.

The methodological approach used to assess the effects of land-use changes on saltmarshes (Chapter II) relied on combining vegetation surveys and the spatial analysis of historic maps (c. 1800) and aerial photographs (1958-2010), which were analysed to map saltmarsh ecosystems and quantify land-use changes. Additionally, vegetation surveys contributed to the identification of saltmarsh typologies: tidally restored saltmarshes (TRS) and enclosed mixed marshes (EMM).

The third Chapter focus on comparing the characteristics of natural saltmarshes with TRS and EMM towards sediment composition and accretion dynamics, linking these with possible consequences for vegetation communities. Accretion rates vary between 0.2 mm/year and 15.53 mm/year. Naturally recovering marshes show lower accretion and higher variability in grain size distribution. TRS present a floristic structure that is similar to natural saltmarshes, while the topographic changes of EMM originate grain size differences but floristic similarities. This is linked with the disturbance gradient that influences floristic diversity.

Aiming to improve the ecosystem services provided by saltmarshes, a coastal defence index was developed to respond to missing institutional-periodical data and demanding software, as well as to be of international application (Chapter IV). Landscape metrics were selected according to shape, complexity, and connectivity parameters, and added to average elevation and distance to the coast, for 1972 and 2010. An equation that measures coastal protection was developed, taking into account the results of PCA and the percentage of explained variation of each component. By using this index, target coastal defence parameters can be outlined, strategies for their conservation designed, and ecological restoration considered.

Floristic composition and diversity developing in secondary marshes result from passive recovery of former reclaimed marshes. To assess the state of these marshes and to evaluate the feasibility of ecological recovery projects, a combined statistical analysis was applied to understand differences in secondary marshes (TRS and EMM) and international examples of managed realignment. A similarity index was calculated to support these comparisons (Chapter V).

Large differences in the floristic composition of Atlantic and Mediterranean saltmarshes hinder the application of managed realignment projects on unmanaged saltmarsh development. Saltmarsh vegetation changes derived from passive recovery develop higher similarity in terms of floristic composition and structure (low, medium and high salt marsh) than active recovery works. Despite the emergence of ubiquitous species, unmanaged saltmarshes are with the local species pool (natural saltmarshes).

Key-words: saltmarsh typologies; land-use changes; floristic composition; ecosystem services; vegetation similarity

Resumo

Os sapais correspondem às formações salgadas do mundo mediterrânico. São compostos por comunidades halófitas e halotolerantes, nanofanefófitos e microfanerófitos, alternando com caméfitos e hemicriptófitos.

A formação de sapais está fortemente relacionada com litorais abrigados da ação direta da ondulação oceânica e das correntes marítimas, contando ao mesmo tempo com a presença de água doce. Este conjunto de fatores permite a deposição de sedimentos finos e taludes suaves.

Os sapais naturais estão entre os ecossistemas mais resilientes e com maiores níveis de produtividade. A sua mono-especificidade e a fraca diversidade florística, confere-lhes uma estabilidade ecológica que se define constante, e consequentemente, capacita-os para dar uma resposta (positiva) a perturbações contínuas. A resiliência dos sapais torna-os bons candidatos a medidas e ações de recuperação e restauro ecológico, visto que estes absorvem rapidamente as perturbações.

Esta dissertação tem como objetivo estudar os impactos das transformações morfológicas ao nível da estrutura e da flora, nas antigas áreas de sapal reclamado, no Alvor e no Arade, e compreender de que forma a sua evolução pode estar relacionada com o restauro ecológico. Pretende-se ainda analisar a composição florística, sedimentar, os serviços de ecossistema e contrapor o restauro ativo (artificial e induzido pelo Homem) com a recuperação passiva, analisando em profundidade as comunidades que se diferenciam nas tipologias de sapal.

A área de estudo corresponde aos sapais que se encontram na Laguna de Alvor, concelhos de Portimão e Lagos e no estuário do Rio Arade e seu afluente, a Ribeira de Boina, concelho de Portimão, localizados na região NUTS II Algarve, em Portugal Continental. Todos estes sapais inserem-se em contextos urbano-turísticos, marcados por uma sazonalidade na procura turística, e simultaneamente com uma área residencial bastante consolidada. A reclamação de terrenos de sapal relaciona-se com a ocupação centenária de ambas as áreas, e consequentemente com as pressões do desenvolvimento agrícola, industrial e urbano. Estas pressões têm marcado o desaparecimento muito significativo de manchas de sapal, e na segunda metade do século XX verificou-se um declínio acentuado na área reclamada, tendo o seu abandono (salinas, tanques e diques) conduzido à formação de diferentes tipologias de sapal.

A evolução da ação humana teve por base a consulta de documentos históricos, nomeadamente cartografia, e também documentos estatísticos que permitissem a sua reconstituição. A avaliação da ação antrópica e da evolução da extensão do sapal baixo e alto, foi elaborada com recurso a cartografia histórica, fotografias aéreas e ortofotomapas de diferentes anos, sobre os quais foram elaboradas operações de geoprocessamento – esta informação foi recolhida no âmbito do Capítulo II.

A reclamação de sapais para a agricultura resultou numa perda de 85% de área de sapal entre 1958 e 2010. Cerca de 41% dos sapais do Arade apresentam uma recuperação para a tipologia "*tidally restored saltmarshes*" (TRS), o que resulta da instalação da maré no interior dos antigos diques, permitindo a recuperação das comunidades pioneiras de sapal baixo. Por outro lado, a Laguna de Alvor apresenta-se mais rica em situações de recuperação do tipo "*enclosed mix marsh*" (EMM), com valores a rondar os 25%. Nas áreas de sapal anteriormente reclamado para a agricultura e onde decorreu efetivamente o arranque da vegetação de sapal, a maré não tem uma entrada direta, mas os sais ascendem por capilaridade, instalando-se um padrão misto de espécies salobras e de água doce. O estudo das comunidades de sapal que

recuperam de forma passiva, vem trazer novas pistas sobre o restauro intertidal, e alimentar possíveis políticas de restauro ecológico, presentes ao longo e toda a dissertação.

O Capítulo III pretendeu justificar os fenómenos de erosão, estabilidade ou acreção dos sapais e a sua relação com a diferenciação florística. Deste modo, foram recolhidas amostras de sedimentos a uma profundidade de 20cm em locais que correspondessem a cada tipologia de sapal. Em laboratório trataram-se 16 cores, onde foi observada a granulometria abaixo dos 63 micra e também a concentração de matéria orgânica. Complementarmente foi analisada a acreção de curto prazo, utilizando-se para o efeito, dois métodos: o pó de tijolo e as varas de erosão. O recurso aos dois métodos referidos teve como objetivo a comparação dos resultados obtidos, avaliando qual seria o mais adequado para os sapais em estudo. Com recurso à análise canónica (CCA) foram analisadas em simultâneo as variáveis bióticas e abióticas das amostras. No final do processo descrito obtiveram-se taxas de acreção que variam entre os 0.2 mm/ano e os 15.53 mm/ano. Os sapais da Ribeira de Boina são aqueles que registam valores de acreção mais baixos, ao contrário dos sapais do Alvor, que apresentam as taxas mais elevadas. Por outro lado, um sapal natural do Arade revelou taxas de erosão preocupantes. Quanto às diferenças tipológicas, os sapais naturais apresentam-se floristica e granulometricamente mais homogéneos, enquanto as alterações topográficas (artificiais), decorrentes de um anterior uso agrícola que está presente nos EMM, originam grandes diferenças nos sedimentos, mas uma maior similaridade na vegetação que coloniza os EMM. No caso dos TRS, verificou-se que estes são mais semelhantes aos sapais naturais. Assim, pode afirmar-se que as diferenças ao nível dos sedimentos favorecem a diferenciação florística nas várias tipologias de sapal.

A par das transformações no uso e ocupação do solo, nomeadamente associadas à reclamação e ao abandono de grandes áreas de sapal, as mudanças ambientais incrementam a exposição dos sapais, contribuindo para o desaparecimento de grandes manchas e consequentemente para a fragmentação de habitats. As várias projeções contemplam ainda, um aumento dos efeitos da subida do nível do mar, em função da magnitude das pressões sobre as áreas costeiras, induzidas pela ação antrópica. Deste modo, o estudo dos serviços de ecossistemas prestado pelos sapais foi objeto de análise no Capítulo IV, com especial enfoque para a defesa costeira, sendo os sapais apontados por vários autores, como tendo a capacidade de se adaptarem a situações de subida do nível do mar, ou de cheias oceânicas. Deste modo, foram aplicadas técnicas de análise da paisagem, através de métricas de paisagem: a dimensão fractal, a forma, o vizinho mais próximo, a dimensão da mancha, tendo sido acrescentadas outras variáveis complementares, como a distância à costa e a elevação. Para uma amostra de sapal de cada área de estudo (Ribeira de Boina, Arade e Alvor) foram analisados dois anos (1972 e 2010). Posteriormente, efetuou-se uma análise de componentes principais (ACP) das métricas de paisagem, o que permitiu desenvolver um índice de defesa costeira, a partir do qual se concluiu: em 1972 os sapais naturais ofereciam uma melhor defesa costeira do que as outras tipologias; já em 2010, os sapais anteriormente reclamados (TRS e EMM), oferecem valores mais elevados de proteção costeira. Assim, pode-se afirmar que as mudanças na forma e na conectividade das manchas de sapal afetaram a performance do índice de defesa costeira de 1972 para 2010. As métricas de paisagem demonstraram constituir elementos válidos para a aferição e avaliação dos serviços de ecossistemas.

Os sapais Atlânticos (EUA e norte da Europa) têm vindo a ser alvo de ações prolongadas de proteção e de restauro ecológico, favorecidas por um conhecimento e domínio dos serviços de ecossistemas prestados por estas áreas. O restauro ativo assume diversas formas, podendo estar associado à compensação ambiental, por perdas ou danos aos habitats de sapal, por destruição de diques e reinstalação artificial do sistema de marés, ou mesmo associado à replantação de espécies pioneiras de sapal. Atualmente, as ações de *managed realignment*, pretendem intervir de forma adaptativa nos ecossistemas costeiros (sapais) de forma a responder às mudanças climáticas (Capítulo V).

No capítulo V pretendeu-se analisar as diferenças na composição florística registadas nos sapais Atlânticos pré e pós intervencionados artificialmente (restauro ativo), com os sapais Mediterrâneos Portugueses, nomeadamente o do Alvor e o do Arade. Para o efeito, estudaram-se os casos internacionais em que os inventários florísticos estivessem disponíveis e que fossem representativos de diferentes medidas de restauro ativo. Quanto aos sapais recuperados passivamente, utilizaram-se os inventários realizados nos sapais do Alvor e do Arade entre 2012 e 2015, tendo sido elaborado um índice de diversidade florística, aplicadas técnicas estatísticas de análise multivariada, incluindo uma análise de componentes principais. Da análise efetuada observou-se uma maior diversidade florística nos sapais EMM, partilhando as espécies ruderais e algumas invasoras com os TRS. Os sapais que recuperaram naturalmente (TRS e EMM), revelaram um índice de similaridade mais elevado do que os sapais recuperados artificialmente (*managed realigment*). Este comportamento deve-se ao facto das espécies pioneiras de sapal, presentes nos sapais recuperados naturalmente, apresentarem similaridades elevadas com o conjunto das espécies locais, enquanto nos sapais recuperados artificialmente, verifica-se a penetração de espécies dos habitats adjacentes (dunas, prados de água doce etc…). Contudo, a recuperação ativa pode desempenhar um papel fundamental como resposta à ocorrência de catástrofes naturais, ou quando as espécies características não conseguem germinar.

A recuperação ecológica sustentável e a gestão dos habitats devem constituir marcos fundamentais na política de planeamento e gestão de sapais, fundamentalmente aplicados a áreas que se encontrem em recuo ou em sapais secundários consolidados, como os TRS ou EMM.

Palavras-chave: tipologias de sapal; alterações do uso do solo; composição florística; serviços de ecossistema; similaridade da vegetação

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List of Abbreviations

A_elev	Average elevation
AL	Alvor
AR	Arade
AWMPFD	Area Weighted Mean Patch Fractal Dimension
BO	Boina
CAP	Class Area Proportion
CCA	Canonical Correspondence Analysis
CDI	Coastal Defense Index
CRT	Controlled Reduced Tide
D	Dunes
D_coast	Distance to Coast
DM	Dry Meadows
DPSIR	Drives Pressures State Impacta and Response
ED	Edge Density
EEC	European Economic Community
EMM	Enclosed Mix Marshes
ES	Ecosystem Services
ES_CoastDef	Ecosystem Service of Coastal Defence (index)
FW	Fresh Water
GPS	Global Positioning System
I	Invasive
ICNF	Instituto de Conservação da Natureza e das Florestas
IPCC	Intergovernmental Panel on Climate Change
MPFD	Mean Patch Fractal Dimension
MPS	Mean Patch Size
MSI	Mean Shape Index
MNN	Mean Nearest Neighbour
MR	Managed Realignment
NS	Natural Saltmarshes
OM	Organic Material
ONG	Organização Não Governamental
PCA	Principal Correspondence Analysis
PCoA	Principal Coordinates Analysis
POOC	Planos de Ordenamento da Orla Costeira
PsCov	Patch Size Coefficient of Variation
R	Ruderal

RTE	Regulated Tidal Exchange
SDI	Shannon´s Diversity Index
SE	Saltmarsh Exclusive
SP	Saltmarsh Preferential
TRS	Tidally Restored Saltmarshes
UPGMA	Unweighted Pair Group Method with Arithmetic Mean

Chapter I - Introduction

1.1 Problem Statment

Saltmarsh ecosystems are one of the most productive ecosystems in the world (M.A., 2005; Deegan et al., 2012), developing in sheltered coastal areas of temperate climate, where wave energy is low and favours the deposition of fine sediments (Moreira, 1992). These ecosystems are characterized by a suite of herbaceous or low shrubby vascular plants that are extremely important to their stability and biological functioning (Flowers & Colmer, 2008). Local conditions — such as climate, hydrology, and geomorphology — result in different sediment supply and marsh elevation, thus contributing to their spatial variability in composition, subsidence, and autocompaction (Allen, 2000). Sediment dynamics in a saltmarsh are influenced by the volume and characteristics of the tidal flow, sediment availability, original topography, and interactions with the vegetation (Allen, 2000; Wilson et al., 2014).

The destruction of saltmarsh ecosystems through erosion is an universal phenomenon (Marani et al., 2011; Francalanci et al., 2013) with major implications to their capacity of delivering ecosystem services, which include sediment accumulation, nutrient cycling, filtering of contaminants, wildlife habitat, flood regulation, and storm protection (Reeve & Karunarathna, 2009; Kim et al., 2011). In addition to natural processes, anthropic activities (i.e. urban, agricultural, industrial, and touristic activities) have contributed in various degrees to saltmarsh vulnerability since the Middle Ages (Currin et al., 2008; Reboreda et al., 2008; Gedan et al., 2009).

These ecosystems are of great ecological and economic value; therefore, major efforts have been made in Europe and elsewhere to halt their loss and degradation and promote their conservation and restoration. In Portugal, until now, the preservation of these important ecosystems (mainly due to their ecosystem service diversity) has been operated almost exclusively by the creation of protected areas; however, restoration, management, and mitigation of impacts driven by environmental changes have been neglected. To maximise the results derived from conservation and restoration efforts, there is an urgent need to better understand passive recovery and the feasibility of active tidal restoration in former reclaimed saltmarshes — subjects that are missing in the national framework.

In the national context, there are vast areas where saltmarshes were reclaimed in the past mainly for agriculture but are now abandoned and therefore could be used to increase the extension of these ecosystems. Portuguese saltmarshes below Mondego are inserted in the Mediterranean Biogeographic region and are able to provide important and unique insights into the few saltmarshes that are Mediterranean in ecological terms but positioned in the Atlantic coast (Costa et al, 2009a) and face enormous pressure from historical human occupation, resource exploitation, agriculture, grazing and rice cultivation, and also urban expansion. In the last decades, the decrease of saltmarsh area has been driven mainly by land use changes, leading to erosion and extinction of large saltmarsh patches. As a consequence of this process, vertical and horizontal saltmarsh migration should occur (Mattheus et al., 2010), although human constructions — especially dykes — are limiting the available space for this potential migration.

Evidences suggest pioneer low marsh communities are developing without artificial intervention, while in former reclaimed saltmarshes pioneer vegetation is also developing, creating a combined mosaic with freshwater communities and invasive species. These evidences were taken as a sign of passive recovery and thus the causes and consequences of these processes need to be addressed. Therefore, the main research question that frames the present thesis investigates what are the impacts of these transformations (in terms of morphology, structure, and flora) in the former reclaimed saltmarshes of Alvor and Arade and understand how these transformations can be connected with ecological restoration.

1.2 Object of Study

Saltmarshes exist in alluvial areas that are frequently flooded and composed of silty or muddy materials. In some cases, they can also be found in sandy soils. Riverine and tidal debris (and sometimes aeolian sand) are often conducted to the mouth (estuary), enriching saltmarsh sedimentary composition (Vasconcellos, 1960). Tidal dynamics are essential for saltmarsh ecology, but it is micro topography that influences the sodium chloride and other salt concentrations, diversifying vegetation typology. Vasconcellos (1960) distinguishes three typologies according to salt concentration and the evolution of the saltmarsh: 1) saline soils, in which salt concentration is very high and their pH is lower than 8.5; these low permeable soils are well structured and the silty texture hampers water absorption from saltmarsh plants, constraining its correct development — plants surviving in saline soils present a green and blue colour (Alvim, 1964); 2) alkaline and saline soils, which can absorb a great percentage of sodium and their pH is approximately 8.5; 3) alkaline non-saline soils, whose pH reaches 10, and where the organic material is in the process of dissolving; this allows organic material to be drawn and deposited at the surface, acquiring a black hue (Vasconcellos, 1960).

Doody (2008) distinguishes six types of environments where saltmarshes can occur, depending on their physical location within a wider geomorphological system: estuary saltmarshes; deltaic saltmarshes; lagoon saltmarshes; barrier island saltmarshes; open coast saltmarshes; loch-head saltmarshes (Figure 1).

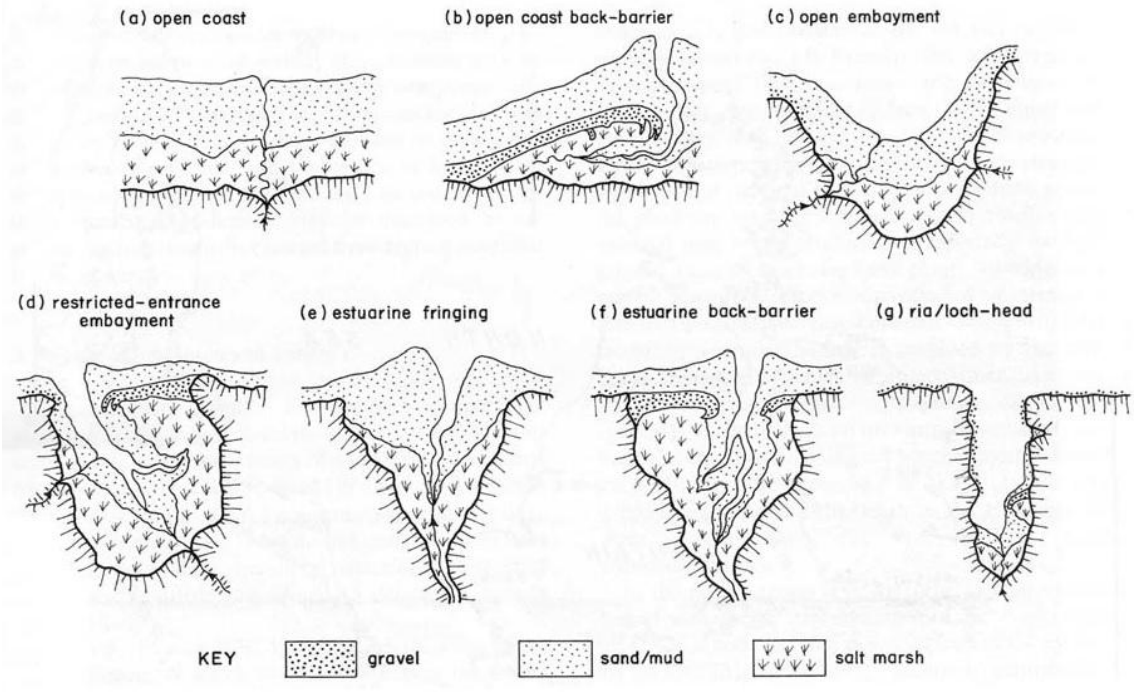


Figure 1. Examples of saltmarsh types accordingly to its position in the wetland system (Allen, 2000; Doddy, 2008).

Soil texture in saltmarshes can vary from clay to sand, depending on the origin of the sediments and the turbulence of the waters (Alvim, 1964). Nevertheless, the majority of Portuguese saltmarshes are of clay origin, located in sedimentation basins defended by seashore dunes (back barrier saltmarshes); frequently, sandy saltmarshes develop in these areas, combining a mix of clay and sand sediments.

Saltmarsh development and formation is related to sheltered shores, depending on the interaction of the land (fresh water input is required) and the sea (tidal inlets) (Doody, 2008). In the bay or estuary shapes, the plant communities of the saltmarshes find their place to grow in the upper shore owing to the deposition of fine sediments (Costa, 2001). Muddy and sandy banks, partially submerged, are the perfect beginning for pioneer vegetation to develop; these plant communities are halophytes and salt-tolerant (Moreira, 1987). According to Costa (2001), saltmarshes correspond to the salt formations of the Mediterranean bioregion, composed of nanophanerophytes and microphanerophytes, alternating with chamaephytes and hemicryptophytes. These amphibious communities manage to survive to great variations of salinity, tide inlets, and temperatures (Neto et al., 2005; Costa et al, 2009a.). The groundwater table plays an important role on saltmarshes, because it balances salt concentrations, allowing the appearance of freshwater species in the less flooded areas of the marsh (Costa, 2001). Braun-Blanquet (1979) defined a classification system that identifies three groups of halophytes: 1) the first group corresponds to those plants that need salts (*Salicornia, Sarcocornia, Arthrocnemum, Limonium, Suaeda, Atriplex, Spartina*…); 2) the second group of halophytes are those requiring the presence of salts (*Juncus maritimun, Salsola vermiculata*…); 3) the third group refers to salt-tolerant communities (*Phragmites australis, Juncus acutus*…). Vegetation is crucial to the sedimentation process, functioning as a retainer of suspended sediments during tide, and simultaneously providing vegetal debris, feeding mudflats to grow and stimulating other plant communities to establish. Therefore, vegetation creates higher frictional force to slow water transition, capturing suspended sediments and favouring their deposition (Allen, 2000; Doddy, 2008). As long as tidal movement and wave action are not acting to erode the settled material, it is possible for accretion to take place, and then new saltmarsh vegetation to develop, increasing the capacity of material retention. Costa (2001) argues that sediment supply influences channel formation, as well as vegetation maturation and elevation. In deltaic saltmarshes, the channel mosaic is usually so dense that ebb waters take different directions, and some are captured into small ponds. These are extremely important to fauna since they provide shelter, food, and nursery in the whole saltmarsh habitat.

The relationship between the tidal regime, the sedimentation process, and vegetation development is what makes saltmarshes such a unique ecosystem, confined to a restrict strip of favourable conditions (Allan, 2000). However, this relationship changes over variable time scales, which can vary from hours, to weeks or even years (considering sediment supply or sea-level change). Nevertheless, storms and waves can transform saltmarsh morphology in a few minutes. Between mean high water spring tides and mean high water neap tides, vegetation succession takes places as the saltmarsh surface upsurges in response to sediment deposition and stabilization (Doddy, 2008).

Saltmarshes are divided into three stages of successional vegetation, which is also related to elevation and inundation: low, medium, and high saltmarsh (Moreira, 1987; Costa, 2001). These distinct but continuous areas of saltmarsh may also be divided into only two types: low and high saltmarshes, whose definitions are mainly used in Anglo-Saxon studies to avoid overstratification. Nevertheless, the present thesis follows Costa (2001) and employs the three different morphologic environments (low, medium, and high saltmarshes). Low saltmarsh corresponds to the muddy vegetated platforms — at an elevation of 2.5m they are daily submerged and colonized by *Spartina maritimae* (Moreira, 1987). Dominant vegetation is hemicryptophytes and chamaephytes (Costa, 2001). The medium saltmarsh designation is used by Costa (2001) and is located up to 3m elevation. Pioneer vegetation developing in medium saltmarsh is followed by annual vegetation in the clearing areas. Inundation plays an important role in medium saltmarsh areas, allowing nanophanerophytes to develop fuelled by the debris brought by the tide. Considering soil properties, clayey saltmarshes rarely surpass 3.40m in terms of elevation, marking the point at which high saltmarshes (frequently sandier) start to appear and bring other species, i.e. *Suaeda vera*, *Limonium algarviense*, *Halimione portucaloides* (Costa, 2001). High saltmarshes are rarely inundated but salts arise by capillarity, allowing salt-tolerant vegetation to develop (i.e. *Arthrocnemum macrostachium, Suaeda albescens, Salsola vermiculata, Frankenia laevis*).

Moreira (1987) states that high saltmarsh is occupied by shrubby vegetation (<1m), daily flooded not more than 10 hours.

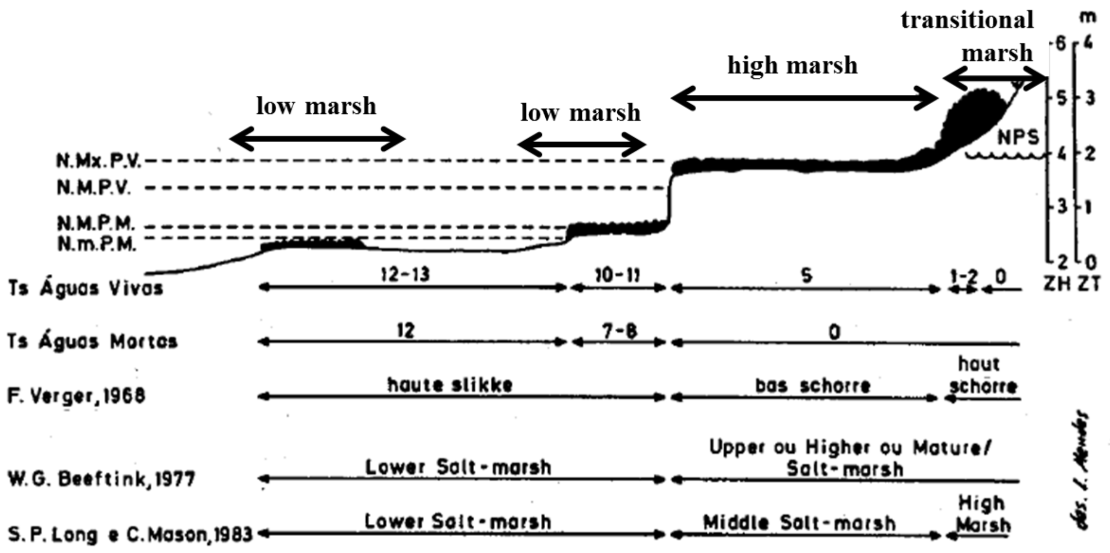


Figure 2. Scheme representing the distribution and classification of saltmarsh stratification (Moreira, 1987).

The last morphologic extract — the saltmarsh ecotone — marks the upper limit of the saltmarsh as an ecosystem. It indicates the soft transition for dunes, saltpans, or *montado* (Moreira, 1987), occupying areas above 3.75m. The saltmarsh ecotone is not visited by tides, but groundwater table plays a crucial role in the development of vegetation. In floristic terms, it is composed mainly of shrubby and arboreal vegetation, i.e. *Atriplex halimus,* but also of herbaceous communities (Costa, 2001).

In this context Caçador (2007a) introduces the concept of chronosequence to describe the plant distribution pattern in a saltmarsh ecosystem. Saltmarshes are frequently pointed as classic examples of succession, where the formation and vertical growth of the ecosystem closely depends on the sedimentation process. Different successional states are related to pioneer vegetation colonization, such as *Spartinetum* and *Salicornetum*, which are represented in the very first saltmarsh band — this first band is where inundation is more frequent and severe; it is followed by *Atriplex*, *Halimione* and *Sarcocornia*, which are part of the same succession.

As mentioned before, tidal cycles can introduce differences in salinity and inundation gradients, delivering topographic variations to the saltmarshes. Vegetation responds to these variations by re-positioning according to its tolerance gradient (Costa, 2001; Caçador, 2007c), and to the physical and chemical characteristics of the sediments (Champman, 1970). The saltmarsh geomorphological system and tidal influence coordinate sediment input (Sousa, 2006), influencing the elevation of shallow tide. Vertical accretion contributes to vegetation development and maturation, which in turn leads to the horizontal expansion of plant communities of saltmarsh pioneer vegetation — this is referred to as zonation (Caçador, 2007c).

Horizontal succession and zonation specify the competition among saltmarsh plants and their capacity to manage physical stress (Caçador, 2007c). This relation is established based on each plant's own characteristics that make it more or less competitive; based on the physical and chemical components of the habitat; and the balance and maturity of the plant community as a whole. The notion of Potential Natural Vegetation (PNV), initially introduced by Tüxen in 1956, represents the climax or the highest stage of vegetation development in a given habitat (Rivas-Martinez & Loidi, 1999). It is defined by the ecological stability achieved by plant communities; these communities depend on their internal organization, which is the most effective response to the continuity of matter and energy delivery. As these conditions do not change, the plant community maintains its structure and composition, becoming representative as potential natural vegetation (Neto et al., 2008). In the cases of perennial mono-layered potential vegetation of coastlines and salt steppes (in which saltmarshes are included), vegetation is studied on the basis of its geopermaseries (geopermasigmeta). The group formed by plant communities that occur in different marshes (low, medium, high) is linked by gradients of environmental variability that culminate by contacting with terrestrial ecosystems — saltmarsh ecotone (Rivas-Martinez, 2005: 141):

> Geopermaseries or geopermasigmetum is the catenal expression used to describe a group of contiguous permasigmeta, delimited by different topographic or edaphic situations. It is influenced by variable climatic, microtopographic and edaphic situations, which give rise to many adjacent ecological situations, populated by permanent perennial communities (contiguous permaseries) at the equilibrium.

Considering stress management, Caçador (2007a) identifies two large areas among saltmarshes: low saltmarshes where physical and chemical factors are extreme and the potential redox is lower — *Spartina* species are the less competitive but develop in physically constraining environments, sustaining long periods of inundation; high saltmarshes offer more favourable conditions, presenting a higher potential redox that is associated with short inundation periods. Caçador (2007a) highlights that the distribution of saltmarsh plants is explained by the tolerance to physical stress (low marsh vegetation) and by the species competition capacity along the gradient of better conditions (medium and high marshes). Changes in this gradient may influence the competition between species and ultimately modify the distribution of saltmarsh vegetation. This might be observed in former reclaimed saltmarshes in agricultural lands, where physical and chemical stress is more favourable to the development of brackish communities because salt delivery is made by capillary ascension — this would probably be a fully closed system, registering higher elevation towards a natural saltmarsh and therefore competition conditions change as alien species find place to settle and develop (i.e. *Cotula coronopifolia*).

Other stressors are pointed by Caçador (2007b) as influencing changes in the competition conditions found in saltmarshes: proximity to urban centres or anthropic activities will possibly increase nitrogen enrichment. A eutrophic saltmarsh may present plants that are more tolerant to nitrogen concentrations, and exclude the presence of others. Anthropic nitrogen influences the distribution patterns of saltmarsh plants, even in terms of their abundance, since primary production is also nitrogen related. Frequency, speed, and tidal cycle duration, as well as the origin and texture of transported materials contribute to differences in the nitrogen concentration gradient, which also depends on saltmarsh maturity. Higher nitrogen concentration in less competitive plants might improve their dominance over others (Caçador, 2007b); belowground competition fails in being determinant as plants start to compete for light instead of competing for nutrients. Aerial competition stimulates the reception of light and the production of leaves (development), leading to another level of competition: inter-specific competition.

1.3 Relevance of the Research

Coastal wetlands occupy about 6% of the Earth's surface and enclose some of the most productive, biodiversity-enriched, and dynamic ecosystems in the world (Allen, 2000; Silva et al., 2007). Nearly a quarter of the world´s population lives in coastal areas in a 100-km belt and within 100m above sea level (IPCC, 2007). Coastal ecosystems are of great importance in terms of ecosystem functioning: they host several biotic and abiotic exchanges; they provide refuge to a large number of sea animals and birds; they ensure resource exploitation, navigation, and the establishment of seaports and cities.

One of the most common coastal systems is estuaries, which enclose other water-dependent ecosystems (wetlands, beaches, mudflats, and saltmarshes) that have been essential for the development of Mankind throughout the centuries. Estuaries are relatively young and volatile systems, evolving from the Flandrian transgression, which followed the last glaciation dating back to 17,000 years ago. For that reason, estuaries are structures of rapid sediment filling, which makes them quite asymmetric over time and space, being influenced by several environmental and anthropic factors (Dürr et al. 2011).

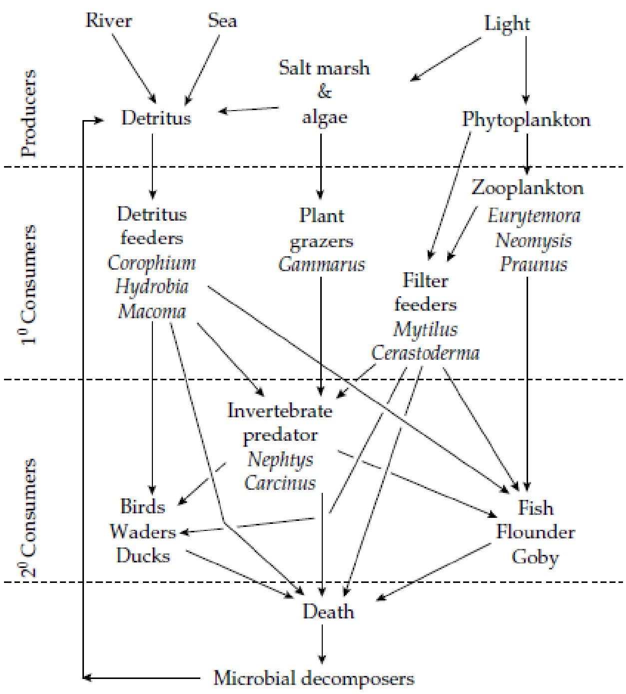


Figure 3. Estuarine food web, positioning saltmarshes as top producers in the whole estuarine ecosystem (McLuscky and Elliot, 2004)

Estuaries are semi-closed systems that maintain a direct connection between land and sea, where tidal water plays a crucial role in the overall coastal ecosystems associated with the estuaries. Salt water is a critical input to these systems, as well as freshwater from continental drainage (McLusky & Elliot, 2004), creating an essential balance for amphibious ecosystems, such as saltmarshes. The classification of estuaries can be grounded on topography, salt gradient, stratification, synchrony, and water flux (Elliot & McLusky, 2002). Additionally, estuaries reflect the combination of marine and riparian sediments, allowing the formation of sand banks or mudflats that shelter various forms of animal and vegetal life.

On the other hand, estuaries are also the stage for important urban, industrial and touristic pressures, as well as resource exploitation and reclamation processes. Saltmarshes located in the fringing areas of estuaries are one of the coastal ecosystems where land-use and land-cover changes have more impact. Set as primary producers, saltmarshes feed estuaries and various sources of consumers in a sophisticated trophic chain (Figure 3).

Saltmarsh habitats are protected by European Environmental Law (Law 140/99 of 24th April – Annex B-1; Directive 92/43/CEE – Annex I). Natura Network 2000 classifies several saltmarsh habitats within the ecosystem itself, regarding the priority and classified saltmarsh habitats (PSRN2000, 2006; Interpretation Manual of European Union Habitats - EUR27, 2007):

- 1310 - *Salicornia* and other annuals that colonise mud and sand: includes annuals that colonise the coast on the boundary between sea and land. In tidal areas, they colonise the periodically inundated muds and sands of marine or interior salt marshes and can be found in areas that are prone to temporary inundation.
- 1320 - *Spartina* swards *(Spartinion maritimae):* colonises a wide range of substrates, from very soft muds to shingle in areas that are sheltered from strong wave action. This occurs on the seaward fringes of saltmarshes and creek-sides and may colonise old pans in the upper saltmarsh.
- 1410 - Mediterranean salt meadows *(Juncetalia maritimi):* salt meadows are frequently found in estuaries and coastal lagoons — this is halophytic vegetation periodically inundated by saline or brackish water and composed of the following: tall rush saltmarshes dominated by *Juncus maritimus* and/or *J. acutus*; short rush, sedge and clover saltmarshes and humid meadows behind the littoral, rich in annual plant species; halophilous marshes along the coast and the coastal lagoons;
- 1420 – Mediterranean and thermo-Atlantic halophilous scrubs *(Sarcocornetea fruticosi):* a perennial vegetation of marine saline muds (schorre) mainly composed of scrub, essentially with a Mediterranean-Atlantic distribution (*Salicornia*, *Limonium vulgare*, *Suaeda* and *Atriplex* communities) and belonging to the *Sarcocornetea pruinosae* class.
- 1430 - Halo-nitrophilous scrubs *(Pegano-Salsoletea):* halo-nitrophilous scrubs belonging to the *Pegano-Salsoletea* class, typical of dry soils under arid climates, sometimes including taller, denser bushes.
- 1510 - Mediterranean salt steppes (*Limonietalia*): associations rich in perennial rosette-shaped species (*Limonium ssp.*) that occupy the Mediterranean coasts and the fringes of Iberian salt basins — soils that are temporarily permeated (though not inundated) by saline water and subject to extreme summer drying, with the formation of salt efflorescence.

Besides the function of these habitats, saltmarshes provide refuge, nursery, and food for many birds, crustacean and fish species, biodiversity preservation, carbon burial, and ensure the good functioning of the biogeochemical cycle. Regarding ecosystem services, saltmarshes deliver provisioning services, such as chemical and drug industries for cosmetics (Gendan et al., 2009) and grazing areas (Milotić et al., 2010); regulating and supporting services, i.e. processing the nutrient cycle and carbon sequestration (Caçador et al., 2007b; Reboreda et al., 2008; Mattheus et al., 2010), as well as coping with rising sea levels, mitigating the consequences of coastal flooding (Simas et al., 2001; Kim & Bartholdy, 2011, Möller, 2006), and providing cultural services mainly connected with sports (Currin et al., 2008).

The richness of saltmarsh habitats and their vulnerability to sea level rise and other environmental changes implied the need for common legislation on priority habitat and conservation. Several authors (Castillo et al., 2000; Bertness et al., 2002; Berteness & Silliman, 2008; Bertness et al., 2008; Banna & Frihy, 2009; Mattheus et al., 2010) indicate that a large part of original saltmarsh areas has been reclaimed for anthropic development since the Middle Ages, despite their importance for the ecology and geology given that they provide protection to shorelines. Saltmarsh erosion and destruction is a worldwide phenomenon. For this reason, many authors have been studying saltmarsh ecosystems in order to understand the causes and consequences of such erosive processes and the disappearance of large saltmarsh patches. Sectorial studies have been produced on the following topics:

- **Vegetation and morphology:** Lousã, 1986; Caçador, 2007a; Civco et al., 1986; Moreira, 1987; Catarino & Caçador, 1991; Costa & Lousã, 1989; Woerner & Hackney, 1997; Turner et al., 2000; Sánchez et al., 2001; Costa, 2001; Costa et al., 1996, 2001, 2009b , 2011, 2012; Esselink et al., 2002; Henning et al., 2002; Kleyer et al., 2003; Neto et al, 2005, 2008; van de Koppel et al., 2005; Caçador et al., 2007c; Kaligaric&Skornic, 2007; Castillo et al., 2008; Feagin, 2008, Mateos-Naranjo et al., 2008, Almeida, 2009; Bertness el at., 2009; Feagin et al., 2009; Jiang et al., 2009; Watson&Byrne, 2009; Yeager et al., 2009, Belzen, 2010; Milotić et al., 2010; Deegan et al., 2012; Wollstonecroft et al., 2011; Kleyer et al., 2012; Sawtschuk&Bioret, 2012; van der Wal&Herman, 2012; de la Fuente et al., 2013; Cortinhas et al., 2014
- **Grain size and sedimentation**: Shen et al., 2008; and its relation with **sea level rise** Stevenson et al., 1986; Allen, 2000; Lefeuvre et al, 2000; Psuty & Moreira, 2001; Van Wijnen & Bakker, 2001; Enssign et al., 2014; Cahoon, 2015; Butzeck et al., 2015; Carey et al., 2015
- **The role of vegetation in the sedimentation process:** Boorman et al., 2001; Carvalho, 2003; Salgueiro & Caçador, 2007; Möller, 2006; Silva et al., 2009
- **Saltmarsh capacity of heavy-metal retention:** Valiela & Cole, 2001; Camacho, 2004; Sousa, 2006; Reboreda et al., 2008; Dürr et al., 2011
- **Saltmarsh erosion processes** : Moreira 1986; 1992; Castillo et al., 2001; Thanh et al 2004; Gu et al 2007; Currin et al 2008; Kim et al 2011b; Deegan et al., 2012
- **Climate change:** Simas et al, 2001; Ferreira et al., 2008; Gedan & Bertness., 2010; Kemp et al., 2010; Kirwan et al. 2010; Kim et al., 2011a

- **Saltmarshes mitigating sea level rise effects:** Cahoon et al., 1995; Turner et al., 2000; Möller et al., 2001; Miller et al. 2001; Bakker et al. 2003; Kemp et al., 2010
- **Saltmarsh reclamation**: Robinson et al., 2004; Bonis et al., 2005; Wolters et al. 2005; Garbutt & Wolters, 2008; Barkowski et al., 2009; MacDonal et al., 2010
- **Human activities** and its reflections directly and indirectly on saltmarsh ecosystems: Moreira, 1986; Castillo et al., 2000; Thanh et al., 2001; Bertness et al., 2002, 2008; Bertness & Silliman, 2008; Banna & Frihy, 2009; Gedan et al., 2009, 2011; Mattheus et al., 2010; Erlandson, 2010;
- **Saltmarsh ecosystem services:** King & Lester, 1995; Costanza et al., 1997, 2008; Möller et al. 2001; Crépin, 2005; Möller, 2006; Currin et al 2008; Reboreda et al 2008; Craft et al., 2009; Loomis & Craft, 2010; Feagin et al., 2010a; Barbier et al., 2011; Spencer & Harvey, 2012; Liquete et al., 2013; van Loon-Steensma & Vellinga,2013; Roebeling et al., 2013; Barbier, 2015.
- **Saltmarsh restoration and recovery:** Flynn et al., 1999; Portnoy, 1999; Verbeek & Storm, 2001; Bakker et al., 2003; Boorman & Boorman, 2002; Strange et al, 2002; Hofstede, 2003; Blackwell et al., 2004; Crépin, 2005; Ledoux et al., 2005; Wolters, 2005; Andrews et al., 2006; Morris, 2013; Plater & Kirby, 2006; Byers & Chmura, 2007; Elliot et al., 2007; Rolo, 2007; Turner et al., 2007; Zedler & West, 2008; Currin et al., 2008; Spencer et al., 2008; Wolters et al., 2008; English & Peterson, 2009; Jacobs et al., 2009; Mattsson et al., 2009; Martín et al., 2010; Mazik et al., 2010; Stagg & Mendelssohn, 2010; Chapman & Underwood, 2011; Davy et al., 2011; Llewellyn & Peyre, 2011; Van den Bruwarne et al., 2011; Spearman, 2011; Mossman et al., 2012b; Duarte et al., 2013; Doody, 2013; Foster et al., 2013; Morris, 2013; Esteves, 2013, 2014; Friess el al., 2014; Pontee, 2014; Greenwood et al., 2015; van Loon-Steensm et al., 2015; Anisfeld et al., 2016.

Decreasing the rhythm to which saltmarshes are eroding is crucial, mainly in the most exposed areas (such as the Alvor estuary), but also to develop effective recovery measures, based on the results of international interventions (Mossman et al., 2012a; Mitchell & Uncles, 2013; Martínez-López et al., 2014). Currently, the de-embankment of historically reclaimed saltmarshes has become a widespread option to re-create saltmarshes (Esteves, 2014).

In Portugal, there is no planning or management strategies that allow for the de-embankment of reclaimed saltmarshes and their progression inland, just as there are no ecological recovery policies. Responses appear locally, usually to deal with the destruction of old dikes by lagoon seich, which also implies some damage to human activities, such as agriculture. Nevertheless, these casuistic measures fall into the coastal defence categories and do not promote saltmarsh ecological recovery.

The policy guidelines focused on estuarine and lagoon ecosystem restoration and watershed management have arisen from the European policy in the scope of the Water Framework Directive (Directive 2000/60/EC), which is, according to Plater & Kirby (2006), one of the most ambitious parts of environmental legislation at the European level. The same authors confirm the need to look at saltmarshes as part of an eco-hydrological tool applied to river and estuarine management. In addition, the creation of perimarine wetlands enhances the response capacity of saltmarshes to the rising sea level and therefore ecological restoration achieves the objectives launched by the Water Framework Directive.

Portuguese saltmarshes have not been given the required importance in the national planning context, although a large percentage of the Portuguese saltmarshes is protected by national laws of environmental protection and also by the European environmental policy (Natura 2000 Network). Historical documents provided remarkable insights into human occupation, land use and cover, and their evolution over centuries. Both estuaries had been occupied even before the Romans, but the Romans were the ones that started to fuel the economic life of these old settlements, mainly based on commercial trades (Loureiro, 1904). Seaports and river side towns started to develop, becoming producers and exporters of valuable merchandise all over the empire, developing and economy in which agriculture and fisheries provided the background of trades (Vieira, 1911). The centenary occupation has significantly influenced land use and land occupation, in which changes are an ongoing process. Additionally, the 1755 earthquake and subsequent tsunami caused profound transformations along the Algarve coast, with direct consequences to the position and morphology of saltmarshes (Pereira et al., 2014). Laguna de Alvor and Rio Arade have been facing major transformations since the beginning of 17^{th} century due to the siltation resulting from the construction of dykes. Saltmarsh embankments were carried out to provide agricultural land and afterwards for urban expansion (Loureiro, 1904; Vieria, 1911).

Moreover, reclamation processes extended to salt wetlands in the 19^{th} century (Pullam, 1988; Batty, 1997). In the first half of the 20^{th} century, more saltmarshes were reclaimed; their main uses started to be associated with salt and rice production, and their pans were later transformed into aquaculture units and adjacent abandoned agricultural fields. Responsibilities are attributable to the Planos de Fomento Agricola (1953-1964). This set of Portuguese Agricultural Plans consisted in a large investment in developing agriculture, and to achieve that several hectares of saltmarshes were reclaimed due to their extremely nutrient-rich estuarine muds.

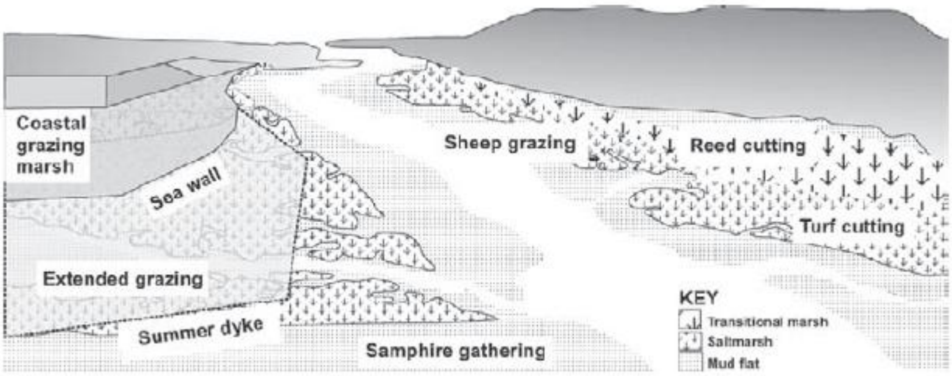


Figure 4. Anthropic modification in saltmarshes with respect for changing and in some cases destroying natural values (Doody, 2008)

This reclamation consisted in embankments, construction of dykes and drainage channels to dry the tidal water from the saltmarsh and also to remove all pioneer vegetation (Alvim, 1964). These types of works encompassed great transformations for saltmarsh land use and occupation, changing also the overall wetland landscape. However, this reclamation process was particularly unsuccessful in the Algarve, because the dry season in Mediterranean climates is marked by the absence of rain, disabling runoff and rainwater accumulation, which are essential to saltmarsh ecosystems. Additionally, high temperatures increase the salt capillarity ascension, thus preventing agriculture to develop. The entrance of Portugal into the former European Economic Community (EEC) and the subsequent obligations related with the Common Agricultural Policy marked the end of targeted investments in agricultural exploitation. Nevertheless, these former saltmarshes found an opportunity to develop a secondary halophytic vegetation (Pullam, 1988), also favoured by the existence of fresh water (proximity of steams) and salt intrusion (estuary areas). These areas have the particularity of combining saltmarsh microtopography and anthropic microtopography.

Hereafter, non-institutional care started to prevent the permanent loss of saltmarsh ecosystems. However, few measures have actually been taken to counter the trend of saltmarsh shrinkage and fragmentation, and to understand the causes and consequences of this process. Spatial and temporal dynamics and patterns of land-use and land-cover changes have poorly been studied in these areas. Furthermore, the full track of saltmarsh natural recovery is also missing from the national framework. The absence of large time series of high-resolution spatial information that could provide insights into land-use and land-cover changes, vegetation cover, and the evolution of the coastline is compromising an effective planning.

Thus, the contribution of international action lines is vital for the establishment of a viable land-use planning tool that allows the understanding of saltmarsh spatial-temporal dynamics, diagnosing their condition (erosion, accretion, or stability), and occasionally intervening in damaged areas in order to stimulate self-recovery. This recovery would probably be a slow transformation from a reclaimed saltmarsh by directly or indirectly installing tidal dynamics, which would boost the hydro-geomorphology system and enhance the installation of pioneer vegetation, thus creating different patterns. On the other hand, international studies point to more targeted measures as compensatory ecological restoration, or even actions incorporating managed realignment. Beyond these methodologies employed to deal with saltmarsh ecosystem management, the objectives of a saltmarsh planning policy are to strengthen the coastal defence capabilities of saltmarshes to cope with sea level rise and protect human settlements and activities in cases of environmental change (ocean storms or coastal flooding).

1.4 Study Area

The main motivation for selecting these study areas is the absence of studies on the Alvor and Arade saltmarshes, particularly those combining a mix analysis of human occupation and land-use and land-cover changes, vegetation, sedimentation, ecosystem services, and recovery. Additionally, the location and exposure of these saltmarshes, as well as the evolution of human occupation in these areas and its influence on these particular saltmarshes were also relevant factors.

Laguna de Alvor´s geomorphological and sediment characteristics have caused several discussions among authors who disagree on its designation. According to Rolo (2007) Laguna de Alvor as a system-barrier bay, while Dias (1993) considered it to be a small coastal lagoon (the lagoon estuarine complex type); Batty (1997) affirmed that it is a mesotidal estuary. In fact, the Laguna de Alvor corresponds to a semi-enclosed water body, with a single, narrow and shallow entrance channel that allows the communication with the ocean. Generally, this definition assumes the Laguna de Alvor as a coastal lagoon system (Dias, 2004b; Rolo, 2007). Laguna de Alvor is classified as a Site of Community Importance by the Natura 2000 network. Topographically, it has a U-shape, resulting from the confluence of four smaller rivers (tributaries): Arão and Odiáxere in the West, and Farelo and Torre in the East, which together form the Alvor River (Rolo, 2007). These four streams come from the Monchique Mountain and cross agriculture alluvial plains (Azerêdo, 1981). The streams regime is the torrential Mediterranean type, with a drying period during the summer (Cabral et al., 1989). The maximum tidal range in the Laguna de Alvor area is 3.30 m (Pereira et al., 1994), with a mesoidal tidal range, between 2-4 m in height. However, due to the strong tidal flow, the water from the outermost zone of Laguna de Alvor is practically renewed in each tidal cycle. This estuarine system presents relatively low depths (0-10 m) but the inputs and outputs are constant (Batty, 1997).

In the coastal zone of the Laguna de Alvor, the dominant winds usually blow from SW and NW, and the average wave height is 1 m. The periods of storm occur when coming from SE (Gibraltar Strait), characterized by 2-2.5 m waves, rarely exceeding 3 m for a few days (Cabral et al., 1989; Pereira & Trindade, 2011). The southwest sea corresponds to storm events, with a wave height of 2-3 m, often reaching 3-4 m high (Cabral et al., 1989; Batty, 1997). The Laguna de Alvor is protected from the ocean by two sandbanks: Meia-Praia on the right-hand side and Alvor on the left (Rolo, 2007).

The Arade River is ranked as a Wetland of International Importance in almost its entire length, except for the river mouth (Camacho, 2004). The Arade river basin has an area of 966 km^2 and a length of 73 km. The Arade estuary is 8 km long and has an average width of less than 1 km. The average depth of the estuary is about 6 m and the maximum does not exceed 10 m near the city of Portimão and also between the jetties protecting the estuary entrance (IPS, 2003). The tributaries of greater significance are the Ribeira da Boina (object of study), 85 km^2 and 23 km in length, and the Ribeira de Odelouca, 514 km^2 and 103 km in length, both tributaries on the right bank (IPS, 2003). The mountains of Monchique and Caldeirão are located 20 and 30 km from the coast, respectively, and constitute the two main cores for the basin´s precipitation, thus generating runoff (IPS, 2003). According to data from the IPS (2003), climate classification of the area under study was performed using the method of Köppen, based on the Praia da Rocha weather station, and corresponds to a climate type Csa. Camacho (2004) shows that in the period from 1951 to 1980, the highest values of relative humidity were recorded between November and February (average value of about 87.5%) and the lowest values between May and September (average value of about 71.6%). Regarding the wind regime, a system of breezes is characterized by very light winds from the north at night and light winds from the south or southeast during the late morning. Throughout the afternoon, the wind wheels to southwest, increasing its intensity and blowing from the west in the late afternoon with moderate intensity. Separating these two systems, breezes occur during periods of calm early evenings and early mornings (IPS, 2003).

In the driest period of the year — July, August and September — the majority of the streams dry up even in years of average annual runoff above average. However, the Arade River registers a permanent flow. Indeed, the Odelouca and Arade rivers have the most regular flow throughout the year, with the highest average flow in the entire Algarve region, because they drain from the mountains of Monchique and Caldeirão (IPS, 2003). The current hydrologic regime is influenced by the operation of the Odelouca and Funcho dams; which hinder the precise knowledge of the flow rates (Camacho, 2004).

1.4.1 Alvor Estuary

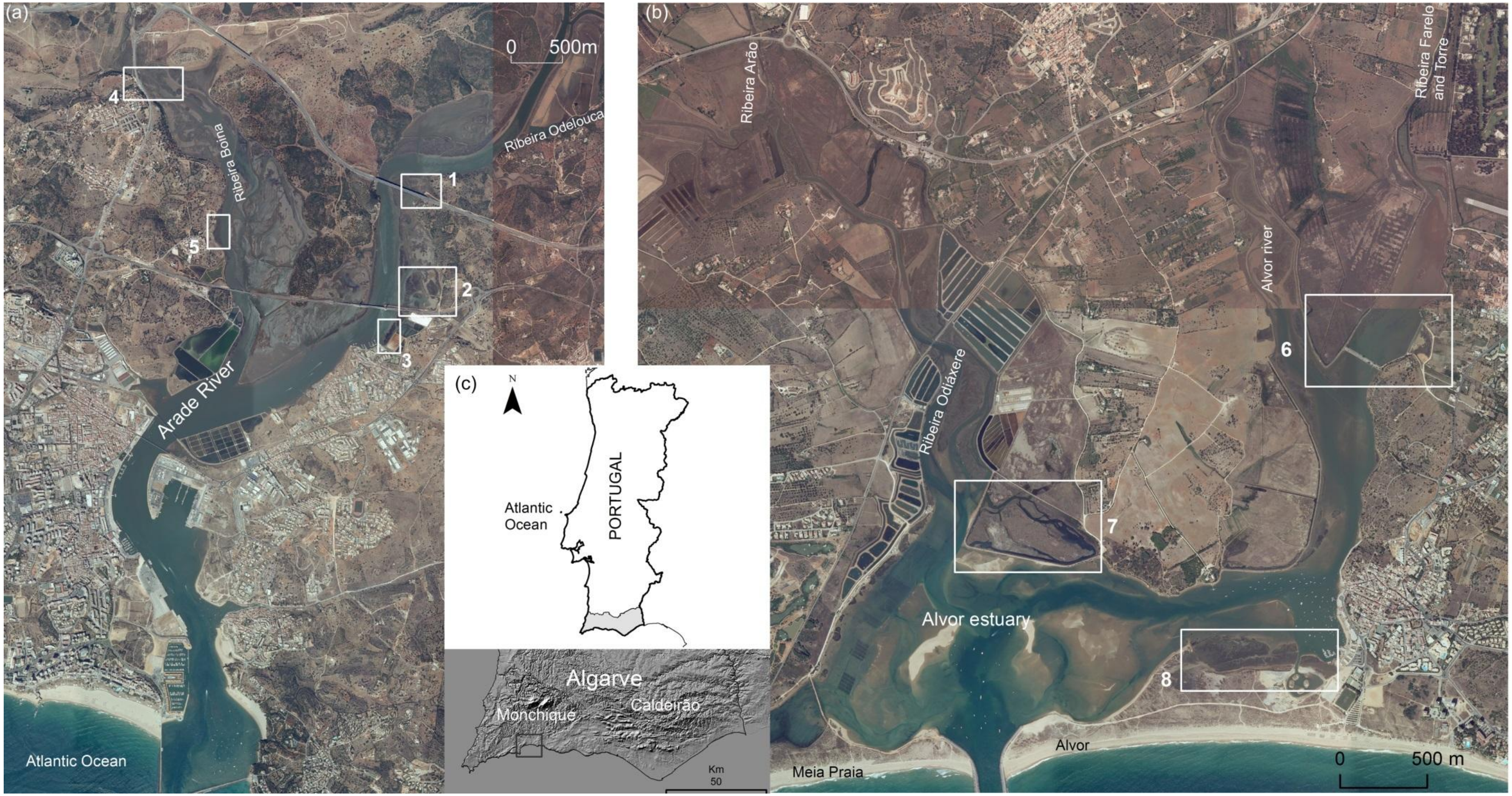
This is a lagoon system, protected by two sand spits, originating a greater dominance of sandy sediments. There is a predominance of back barrier saltmarshes. Here, the urban and touristic pressure is clearly present in the daily life of the estuary and its outskirts. The Alvor estuary has been stage to profound transformations, not only in its shape (more open to the sea or more sheltered by the dune spit), but also regarding land uses. Several embankments, reclamation, and real estate interests have changed the Alvor landscape: upstream there used to be active agricultural exploitation and grazing areas, and some segments of the front shore used to be occupied by saltpans. Large former reclaimed saltmarsh areas found an opportunity to recover through the abandonment of the first sector activities. Sandy saltmarshes evolving with a different morphologic profile needed to be framed in a context of ecological recovery.

1.4.2 Arade River and Estuary

The estuarine saltmarshes of Arade are protected from direct wave erosion and count on a significant fresh water input from Arade and other tributaries — Funcho dam influences sediment availability in the Arade estuary. Several land-use and land-cover changes have occurred in Arade, which reflected in the saltmarsh morphology and configuration. Its proximity to a great urban centre (Portimão) gave the possibility of several embankments and reclamation (i.e. wall building, riprap, flood gates building, water mills, and other tidal structures), which have remarkably modified the saltmarsh succession. Former dykeland started to evolve with the establishment of the tidal regime, allowing *Spartina* communities to grow.

1.4.3 Boina stream

Boina stream is a small water surface in a sheltered position, which has contributed to the development of mature and stable natural saltmarsh in the last decades (Loch-head saltmarshes accordingly to Doody, 2008). Significant farming and grazing areas were obtained from saltmarsh reclamation. A complex network of dykes and other tidal protection systems (flood gates) favoured saltmarsh dredging. As in Arade and Alvor, the dykes' destruction or simply the abandonment of agricultural practices have led to a small recovery of saltmarsh inside former dykeland. This replicates passive recovery in saltmarshes through the creation of low energy environments allowing saltmarsh vegetation to develop sheltered from wave action. Nevertheless, this stage reflects several transformations in land use and land cover, as well as alterations in saltmarsh plant communities, which need to be understood.



Legend: (Arade and Boina) 1 - Estuarine fluvial salt marshes; 2 - Estuarine fringing salt marshes and estuarine transitional salt marshes; 3 - Estuarine fluvial salt marshes; 4 - Estuarine transitional salt marshes; (Alvor) 5 - Estuarine fluvial salt marshes and fringing salt marshes; 6 - Deltaic salt marshes; 7 - Back barrier salt marshes (based on Allen, 2000).

Figure 5. Geographical inbound of the study area, with the location of the main saltmarshes in focus.

1.5 Objectives

Few studies have been developed that integrate both the human and physical aspects of geography applied to the study of the anthropic and natural factors that contribute to the understanding of saltmarsh dynamics. Saltmarshes are complex ecosystems whose resilience and carrying capacity have been put to the test since the early days of human occupation. Many transformations have occurred in saltmarsh lands, which mostly resulted in a reduction of the area occupied by saltmarshes, endangering the ecological sustainability of these ecosystems.

This work intends to study the processes and dynamics underlying the Alvor and Arade saltmarsh evolution, thus contributing to the knowledge of habitat management and ecological restoration.

In order to address this topic, a comparative analysis of the saltmarshes of the Alvor estuary and the Arade River (plus Ribeira de Boina) was performed. This analysis was based on the consequences of land-use changes deriving from anthropic activities in the floristic composition, morphology, and structure of these saltmarshes, as well as the impacts on ecosystem services. Firstly, the contributions of human occupation to the Alvor and Arade saltmarshes' dynamics were identified; then, early land-use changes and their consequences to the retreat of saltmarsh areas were studied; eroding and accreting saltmarsh patches were identified, addressing the conditions to replicate saltmarsh recovery processes; an inventory of the flora and vegetation of these saltmarshes was made and the consequences of land-use changes to floristic differences between these marshes were identified; finally, the origin and the possible contribution of former reclaimed saltmarshes to ecological recovery options was studied.

The scientific questions underlying this study are:

- What are the consequences of past and present anthropic activities in saltmarsh structure and composition?
- How are saltmarshes evolving and what is the extent of their retreat?
- Which impacts are predicted to local populations and their activities?
- How may the abandonment or the continuous use of saltmarsh reclamation influence the habitat recovery rate?
- What differences can be found in passive recovered saltmarshes towards natural saltmarshes?
- What is the role of saltmarshes in the mitigation of extreme events associated with environmental changes?
- What are the consequences of passive recovery for the floristic composition of the saltmarshes?

- How can saltmarsh recovery be framed in a Mediterranean context?

This thesis is expected to advance in the knowledge of saltmarsh passive recovery and inform the planning system about the past and future evolution of saltmarsh structure and dynamic in floristic and sedimentation processes. This work aims to build the basis for options in habitat management and saltmarsh ecosystem recovery, bridging a gap of knowledge in the national planning context, and also promoting an international understanding of the role of land-use changes in distinguishing different typologies (profiles) of saltmarsh recoveries in the absence of targeted policies.

1.6 Hypothesis

The main hypothesis that guides the whole research focuses on *the abandonment resulting from the evolution of former reclaimed saltmarshes that allowed the emergence of saltmarsh typologies. These typologies are the reflex of different evolution patterns, as well as the way tides have influenced a passive recovery regarding flora and vegetation composition.*

Secondary hypotheses are addressed within each chapter of the thesis:

- There are consequences of past anthropic activities in the saltmarsh evolution and composition (Chapter II);
- The main consequence of saltmarsh reclamation is the major decrease of saltmarsh patches and the reduction of the area occupied by natural saltmarshes (Chapter II);
- Differences between saltmarshes are mostly in terms of vegetation and plant composition (Chapter II);
- Reclaimed saltmarsh dynamics and evolution are similar to natural saltmarshes, but there are differences in floristic composition and grain size (Chapter III);
- There are differences among accretion and erosion rates, and those differences are related with grain size characteristics, floristic structure, and the saltmarsh typology (Chapter III);
- Landscape metrics can provide information on saltmarsh shape, complexity and connectivity and be a tool to assess ecosystem services (Chapter IV);
- Former reclaimed saltmarshes are able to provide complementary coastal defence services in a context of passive recovery (Chapter IV);
- Abandoned reclamation areas and passive tidal reactivation are the answer to passive saltmarsh recovery (Chapter V);

- The combination of a Mediterranean climate and soft de-embankments support floristic and sediment recovery in former reclaimed saltmarshes, which can evolve to become natural saltmarshes (Chapter V);
- The saltmarshes of Alvor and Arade combine several characteristics that make them good candidates for ecological recovery (Chapter V).

1.7 Materials and Methods (approach)

1.7.1 Human activities and historical evolution

This work was developed based on the consultation of both historical and recent records and documents that led to tracing the anthropic landscape evolution, namely the installation of dykes, saltmarsh reclamation for agriculture and saline extraction, fishermen's settlements and the evolution of their activities (collection of shellfish and bait), aquaculture and its history, and the evolution of local touristic activities (see Appendix A).

1.7.2 Salmarsh temporal evolution

Low and high saltmarsh areas and patches were mapped for different years from 1958 to 2010 using the ArcGis 10.2 software. There is also some work on historical cartography, from the late 1790s. Additionally land-use and land-cover changes will be identified using the same aerial photographs and ortophotomaps in order to understand the spatial and temporal evolution of the Alvor and Arade saltmarshes. Erosion and accretion differences among years will be recorded and used to address the policy options in terms of recovery and habitat management (see Appendix A).

1.7.3 Saltmarsh evolution: floristic communities structure

The preparation of floristic surveys is in accordance with the Zurich-Montpelier methodology in order to address floristic differences among marshes. Floristic surveys were conducted in following the abundance/dominance scores method (Braun-Blanquet, 1979) by using sampling quadrats of 2 m^2 along selected transects. A total of 368 floristic surveys were conducted in randomly selected saltmarsh areas: 167 were in Alvor saltmarshes and 201 in Arade River and Boina creek. These inventories were applied to the saltmarsh typologies upon which this thesis is structured during several time periods in order to avoid seasonal variability of species flowering. The botanical nomenclature followed the works of Castroviejo et al., (1986, 2007), Franco (1971, 1984); Franco & Rocha Afonso (1994, 1998, and 2003) and Rivas-Martínez et al., (2005). The degree of presence (Braun-Blanquet, 1979) was calculated to evaluate the differences in species' richness between saltmarsh areas. The presence was estimated in percentages of a species and classified according to a chosen scale into a set of 'classes of presence’. The classes were defined as follows (Costa et al., 2009b): *r* (< 6%); + (6-10%); *I* (11-20%); *II* (21-40%); *III* (41-60%); *IV* (61-80%); *V* (> 81%). Surveyed areas were selected from pre-defined situations at each study site and different saltmarsh typology. This analysis allowed evaluating specific losses, community transformations or changes in saltmarsh relative abundance. Data were analysed using CANOCO and SYNTAXA software.

1.7.4 Soil´s characteristics and accretion/erosion dynamics

Sediment characterization and marsh dynamics: collected materials were studied to determine pH, texture, mineralogy, organic and calcium carbonate content, elemental and isotopic composition. Texture relies on grain size analysis of the fraction over 63 µ (conventional sieving methods) and under 63 µ (laser diffraction); organic matter content (total, labile and refractory) will be assessed by loss on ignition and calcium carbonate content by the gasometric method using a calcimeter; fine fraction mineralogy will be determined by X-ray diffractometry. The suspended sediments concentration is related with sediment supply, vital to vegetation-sedimentation cycle, despite of other important interactions, like the sea level (hydroperiod). It is used to describe the dry mass of sediment that is suspended in a defined volume of water, during a defined period. This process contributed to the understanding of the sediment deposition onto the marsh surface (see Appendix B).

1.7.5 Ecosystem services and landscape metrics

Landscape metrics at patch, class, and landscape levels were calculated to assess habitat fragmentation. The software FRAGSTATS 4.1, ArcGIS 10.2 extension Patch Analyst was used. Landscape metrics were tested using previous work on saltmarsh differentiation and land-use changes in order to evaluate ecosystem service delivery. The materials used are aerial photographs and ortophotomaps on the study area and manual classification of saltmarsh morphology and typology (see Appendix C).

1.7.6 Recover viability

The viability of recovery was explored starting from an analysis of the vegetation composition of stabilized salt marshes and comparing them with the vegetation development of nearby restored areas. International examples of managed realignment sites (MR), representative of active recovery, are expected to provide the background to assess the progress of vegetation development. On the other hand, passive recovery observed in Western Algarve salt marshes (Portugal) was compared with natural salt marshes based on the similarity of vegetation cover and degree of presence (species-by-species). This would provide valuable comparison of vegetation succession both in Atlantic and Mediterranean salt marshes in different contexts of recovery (active or passive), trying to provide further insights into unmanaged salt marsh development.

1.7.7 Discussion and conclusion

An overall discussion of pressures and challenges is provided to frame the results and discussions stirred by the previous chapters. Saltmarsh management found its preliminary suggestions on international examples, and on a DPSIR for the study area.

1.8 Organization

This thesis is organized in seven chapters. The core Chapters II to V correspond to published, accepted or submitted papers in peer reviewed international journals indexed by *ISI Web of Knowledge*.

Firstly, in the Introduction (Chapter I), the problem is stated, and the object of study, the relevance of the research, the study areas, and the reasons for the selection of these areas are presented. The objectives are outlined, and the hypotheses underlying this thesis are discussed. Additionally, the internal organization of the thesis is presented in a table that highlights the guidelines that provide the whole research with a consistent common thread: saltmarsh typologies.

The identification of saltmarsh typologies was possible by fulfilling the main objectives underlying this research (topics A, B, and C). Here the human-induced land-use changes are discussed in terms of loss and degradation of intertidal habitats, where Portuguese saltmarsh dynamics are greatly influenced by historic uses and consequent habitat degradation. The first paper — 'Impacts of land use changes and ecosystem recovery on saltmarshes in Portugal' — was published in 2014 in *Ocean and Coastal Management* and corresponds to Chapter 2. It uses an original approach that combines vegetation surveys with the spatial analysis of historic maps and aerial photographs to assess the effects of land-use changes on saltmarshes in two areas in the Algarve, Southern Portugal. The study area includes Laguna de Alvor and Rio Arade, distinguishing two saltmarsh typologies: (1) enclosed mix marshes, formed by patches of brackish, freshwater and some invasive species developing due to saline intrusion in areas where dykes have not been breached; and (2) tidally-restored saltmarshes, formed in areas where dyke breaching allows the incursion of tides and the development of a vegetation structure that is similar to natural saltmarshes.

Chapter III discusses the sedimentation characteristics of naturally-recovering marshes within reclaimed areas, and compares the characteristics of natural saltmarshes with two typologies of saltmarshes that are recovering naturally in previously reclaimed land where agricultural practices have been abandoned for 30-40 years in the Algarve (Southern Portugal). Short-term sedimentation rates were measured using brick dust and erosion pins in 16 cores collected in the Alvor estuary and the Arade River. Grain size distribution and organic matter concentration were also analysed to understand the interactions between sediment processes and floristic composition. Having a better understanding of the characteristics of saltmarshes that are recovering naturally can provide useful insights to assess whether active intervention may be the best way forward in all locations. This paper addresses the concerns arisen in topic D.

The fourth Chapter covers landscape metrics applied to former reclaimed saltmarshes as a tool to evaluate ecosystem services. It corresponds to the approach on topics E and F. The analyses of saltmarsh ecosystem services have been particularly targeted at the capacity to mitigate climate change effects, coping with rising sea levels, and dealing with flood management. Landscape metrics were selected according to shape, complexity, and connectivity parameters, and added to average elevation and distance to the coast, for two years: 1972 and 2010. An equation measuring the coastal protection was developed, aiming to outline target coastal defence parameters, design conservation strategies, and consider ecological restoration.

The fifth Chapter addresses the discussion underlying the recovery processes and restoration projects: 'Vegetation diversity and patterns in realigned and naturally recovered saltmarshes'. The hypothesis guiding this part of the research stated that saltmarsh vegetation changes derived from passive recovery are more successful than active recovery works (i.e. managed realignment, restoration). The vegetation composition of stabilized saltmarshes was analysed and compared with the vegetation developed in nearby restored areas resorting to international examples of managed realignment sites (active recovery). This chapter aims to set a reference to assess the progress of the development of vegetation due to passive recovery observed in Alvor and Arade saltmarshes, when compared with natural saltmarshes. This chapter concerns topics C and F.

The floristic catalogue is presented in Chapter VI. Discussion and Conclusions close up the last chapter of this thesis (Chapter VII), and provide recommendations on habitat management and restoration policies. This chapter discusses the results of the previous chapters and their adequacy to test the underling hypothesis. Recommendations for the Alvor and Arade saltmarshes are put together in a guiding framework with the aim of contributing to the evaluation of the best restoration tools to achieve habitat management goals. Compensatory measures can ensure the replacement of some lost ecosystem services by others with the same biological function, favouring soft de-embankment. This should result on proposals for mitigating the direct anthropic impacts on saltmarshes and define restoration measures, as well as discussing active and passive recovery. Finally, the hypotheses were discussed and summarised in a table. Final remarks and further work to be developed are also addressed in this chapter. The following scheme summarizes the organization of the chapters (Table 1):

Table 1. Schematic summary of thesis organization

CHAPTER I - Introduction
Problem statement; object of study; relevance of the research; study area; objectives; hypothesis; approach; organization
CHAPTER II – Impacts of land use changes and ecosystems recovery in Portugal
Human occupation in Alvor and Arade saltmarshes; Land use/cover changes from 1958 to 2010 Floristic surveys; Naturally recovering marshes **Saltmarsh typologies**
CHAPTER III – Sedimentation characteristics of naturally-recovering marsh within reclaimed areas
Short term accretion (9months) - accretion and erosion rates Grain size study; Floristic surveys Differentiation of saltmarsh typologies in grain size, accretion dynamics and vegetation communities Guidelines for recovery options within **saltmarsh typologies**
CHAPTER IV – Landscape metrics applied to reclaimed saltmarshes: a tool to evaluate ecosystems services?
Saltmarsh ecosystem services; Landscape metrics analysis applied to saltmarsh typologies Saltmarshes evolution in landscape patterns from 1972 to 2010 Coastal defence index (1972-2010) Relation between metrics, **saltmarsh typologies** and the service of coastal defence provided by different saltmarshes
CHAPTER V –Vegetation diversity and patterns in realigned and naturally recovered saltmarshes
Assess the progress vegetation recovery occurring from passive recovery observed in Alvor and Arade **saltmarsh typologies**, comparing to international examples of restored saltmarshes in terms of species composition. Establish recover trajectories based on similarity of vegetation cover and degree of presence (specie-by-specie).
CHAPTER VI – Floristic cast
Flora and vegetation aspects are compiled in a floristic cast. Information on the floristic surveys are provided.
CHAPTER VII – Discussion and Conclusions
Pressures; Challenges; DPSIR Management options, Recommendations for Alvor and Arade according to saltmarsh typologies; Hypothesis revisiting and limitations to the research; Further work to be developed

Chapter II – The impactos of land-use changes on the recovery of saltmarshes in Portugal

THE IMPACTS OF LAND-USE CHANGES ON THE RECOVERY OF SALTMARSHES IN PORTUGAL

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2.1 Abstract

Human-induced land-use changes have resulted in loss and degradation of intertidal environments worldwide. Saltmarsh ecosystem dynamics in Portugal are greatly influenced by historic uses and consequent habitat degradation. This study uses an original approach combining vegetation surveys and spatial analysis of historic maps and aerial photographs to assess the effects of land use changes on saltmarshes in two areas in the Algarve, southern Portugal. Historical maps from c. 1800 and aerial photographs from 1958 to 2010 were analysed to map saltmarsh ecosystems and quantify land-use changes in the Alvor estuary and Arade River. Between c. 1800 and 2010 more than half of saltmarshes were lost due to dyke building and saltmarsh reclamation for agriculture. In mid-1960s, the abandonment of reclaimed agricultural areas resulted in the recolonization of saltmarsh vegetation, which developed physically separated from natural marshes. In the study area, these saltmarshes naturally evolved into two distinct typologies: (1) enclosed mix marshes, formed by patches of brackish, freshwater and some invasive species developing due to saline intrusion in areas where dykes have not been breached; and (2) tidally-restored saltmarshes, formed in areas where dyke breaching allows incursion of tides and development of a vegetation structure similar to natural saltmarshes. In Europe, passive (without human intervention) and active (artificially planned) saltmarsh restoration are important mechanisms for voluntary or statutory re-creation of intertidal habitats. Improved understanding of the factors influencing the development of distinct saltmarsh typologies through passive ecosystem recovery can provide new insights to support decision-making concerning intertidal habitat restoration[1].

Key-words: de-embankment; saltmarsh loss; tidally-restored saltmarshes; enclosed mix marshes; habitat recovery

[1] For further data and supplementry information on this chapter, see Appendix A.

2.2 Introduction

Human-induced saltmarsh loss is widespread along the world's temperate coasts (Moreira, 1986, 1992, Gu et al., 2007, Currin et al., 2008, Neto et al., 2010). Many studies have assessed the causes and consequences of saltmarsh loss (Elliot et al., 1999, Miller et al., 2001, Wu et al., 2002, Cahoon et al., 2006, Marfai and King, 2008), including in Portugal (Moreira, 1986; Ferreira et al., 2008). Degradation and loss of saltmarshes have major implications to their capacity of delivering ecosystem services, including: sediment accumulation; nutrients cycling; filtering of contaminants; wildlife habitat; flood regulation and storm protection (Simas et al., 2001; Currin et al., 2008; Feagin et al., 2008; Gedan & Bertness, 2010; Kim et al., 2011). The extension, exposure and orientation of the saltmarsh in relation to the coast, and the vegetation cover and maturity level of the plant communities influence the capacity to provide ecosystem services.

Environmental changes driven by natural processes, such as sea-level rise, have greatly affected saltmarshes (Simas et al., 2001; Morris et al., 2002; Gedan & Bertness, 2010; Mensah & FitzGibbon, 2012). However, human occupation in coastal areas has affected saltmarhes natural dynamics (Reeve & Karunarathna, 2009). In Europe, since the Middle Age, artificial manipulation of these ecosystems often enhanced erosion rates and resulted in substantial transformations in saltmarsh composition, distribution and functions (Castillo, 2000; Mattheus et al., 2010, Gedan et al., 2011). These transformations increased saltmarshes vulnerability to environmental changes, reducing their natural adaptive capacity (French and Burningham, 2003; Gedan, 2009). The urban, touristic, industrial and agricultural uses along the Portuguese coast have severely prevented the chances of saltmarsh recovery (Moreira, 1992; Rolo, 2007).

Natural saltmarsh evolution and diversity have been widely studied (Moreira, 1987; Costa & Lousã, 1989; Leveuvre et al., 2000; Caçador et al., 2007c; Deegan et al., 2012). More recently, studies have focused on vegetation cover and species richness in embanked areas (Kleyer et al., 2003; Robinson et al., 2004; Bonis et al., 2005) and as a result of de-embankment (Garbutt & Wolters, 2008; Barkowski et al., 2009). Such studies describe the effects of human intervention on the characteristics and recovery of saltmarshes and are important to inform practices and policy concerning intertidal habitat creation (e.g. Habitat Directive objectives and/or to compensate for loss of protected areas) and their ability to deliver ecosystem services (Mossman et al., 2012b; Spencer & Harvey, 2012). In southern Europe (e.g. Portugal, Spain and France), economic drivers led to the abandonment of agricultural practices in some areas, including coastal reclaimed land, creating opportunities for saltmarsh colonization (Boorman et al, 2002; Vélez-Martín et al., 2010).

This article assesses the effects of land use change on the evolution of secondary saltmarshes and describes the emerging distinct typologies using selected areas in the Algarve

(Southern Portugal) as case study. A novel approach combining vegetation surveys and mapping of land-use changes based on analyses of historic maps and aerial photography is used here to assess how historic land uses (from 1800´s to 2010) contribute to the development of different saltmarsh typologies. The article first identifies the effects of human activities on saltmarsh dynamics along the Alvor estuary and the Arade river mouth. Then results from spatial analysis of land use changes and vegetation surveys are used to identify the main typologies of recovered saltmarshes. The discussion focuses on the drivers influencing unmanaged saltmarsh recovery under a Mediterranean climate and highlights wider applications concerning the restoration of saltmarsh habitats. The article ends with a summary of the main conclusions concerning the effects of land-use changes on the typology and development of secondary saltmarshes.

2.2 Study area

The Alvor estuary (Laguna de Alvor) and the marshes of the Arade river (and the tributary Ribeira de Boina) are located in the Portimão municipality (Algarve, southern Portugal, see Fig.5 in Chapter I). The area has temperate Mediterranean climate, showing mean annual temperature around 17ºC (at Faro Airport, IGP, 2005) and annual mean precipitation at the coast between 400 mm and 500 mm, with the wet season occurring from October to April (Ribeiro et al., 1997). Alvor is a tidal lagoon dominated by sandy sediments of maritime origin set in a mesotidal environment with little freshwater input. The Arade River mouth is a drowned river valley subjected to tidal influence from the mouth to 13 km upstream.

Human occupation in the study area began in the Neolithic (2,000 – 1,600 BC). The Carthaginians and the Romans had introduced fish salting and salt exploitation as regular economic activities. In 715 BC the Arab occupation enhanced the economic importance of Portimão and Alvor, which were important seaports exporting products to the rest of the Arab empire (Vieira, 1911). The economic activities were based on activities undertaken at high marsh areas, including: fishing, bivalve gathering, sun-drying fish and fruit and extraction of salt in salt pans (Loureiro, 1904; Vieira, 1911).

The earthquake of 1755 and the associated major tsunami reconfigured the Portuguese coast (Dias, 2004b). Records suggest that the tsunami wave entered 667 m inland in the area of Alvor altering the ebb delta morphology preventing tall ships to enter the estuary; saltmarshes were one of the most affected ecosystems (Loureiro, 1904). The tsunami resulted in a greater mobility of the Alvor estuary inlet, which threatened adjacent property and triggered the construction of dykes and embankments leading to saltmarsh reclamation (Pullam, 1988). The tsunami impact at the coast triggered a series of floods in the Arade River, in which the sea entered 14 m inland, causing siltation of the river mouth (Loureiro, 1904).

2.3 Materials and methods

Qualitative data sources were analysed to understand the socio-economic and environmental factors influencing saltmarsh evolution in the study area. The main sources of qualitative information included: parish records of population surveys and land registry; the genealogy of Algarve's landlords and noble families, in order to access the land use changes and acquisitions; and port authorities' records about damage caused by the earthquake. Historical documents allowed the reconstruction of the human occupation and the economic activities that have taken place in marsh areas, including land reclamation.

The methodology applied here is an original approach specifically designed for this study. A combination of vegetation surveys and spatial analysis of historic maps and aerial photography is used to assess the effects of land-use changes on saltmarsh typologies in the study areas. The methods of the spatial analysis and vegetation surveys are detailed below.

2.3.1 Spatial analysis

Historical cartography was essential in the spatial analysis of changes in saltmarsh boundaries and extent. Spatial analysis of historic data was possible only for the Arade River due to lack of data sources with adequate quality for the Alvor estuary. Although a map from 1909 shows the area of the Alvor, it does not show features that allow georeferencing. Sources of spatial information used in this study are listed in Table 2. The hydrography chart covering the area of the Arade river (named then Villa Nova de Portimão) dated from the period 1789-1800 is used here as a historic baseline for the photointerpretation and geoprocessing operations. The historic chart was georeferenced using ArcGIS 10 resulting in a root mean square error of 8.90 m. This historic chart allowed clear identification of land cover types (Figure 6a), such as: high and low marshes (indicated on the original map by the common name of the dominant species); salt pans were easily identified by their shape, name, and location in the edge of the high marsh; other agriculture activities showing in the chart were not object of analysis. Geoprocessing analysis resulted in a land cover map and a table showing the extent of saltmarsh areas (Table 3).

Analysis of aerial photos and ortophotomaps were also used to map the human occupation in the study area. The Portuguese Cartographic Institute has obtained aerial photographs covering areas of the Portuguese coast at intervals of approximately ten years.

Aerial photographs covering the study area were obtained in 1958, 1972 and 1987. All maps and photographs were georeferenced using the Portuguese national coordinate system ETRS_1989_Portugal_TM06. Spatial analysis in ArcGIS 10 was undertaken to estimate areas of saltmarsh accretion, erosion and stability. Validation of photo-interpretation was undertaken during fieldwork, conducted at the same dates of the vegetation surveys described below.

Table 2. Cartographic data sources used in this study.

Date	**Chart type**	**Source**	**Information provided**
1789-1800	Segundo Plano Hidrográfico do Rio de Villa Nova de Portimão	Baltazar de Azevedo Coutinho, Eng.º. Instituto Geográfico do Exército - IGEOe	high and low saltmarshes, mudflats, saltpans, agriculture
1884	Carta de Portugal 1:100000, No.36 (Military Chart)	Direcção Geral dos Trabalhos Geodésicos do Reino,	coastal configuration and saltmarshes position
1909	Planta da Bahia do Porto de Lagos	Loureiro, Adolfo (1904), *Os Portos Marítimos de Portugal*, vol.5 (1904-1909)	coastal configuration and saltmarshes position
1922 1923 1930	Carta de Portugal 1:50000, sheets 49D, 52ª; 49C (Land Use Chart)	Direcção Geral dos Trabalhos Geodésicos e Topográficos	Wetlands, saltmarshes and irrigation schemes
1950 1951	Carta Agricola e Florestal 1:25000, sheets 594, 595, 603	Secretariado Geral da Agricultura	agriculture uses, land occupation and transformation of saltmarshes
1967 1970	Carta de Portugal 1:100000, sheets 52, 49 (Hydrographic Chart)	Instituto Geográfico e Cadastral	alterations in irrigation schemes, reservoirs, mudflats, marshes and salt pans.
1958 1972 1987	Aerial photos	Centro de Estudos Geográficos (1958) Instituto Geográfico Português (IGP)	Land cover
1995 2005 2010	Ortophotomaps	Instituto Geográfico Português (IGP)	Land cover

2.3.2 Vegetation data

Floristic surveys were conducted in the following dates: 15-18 October 2012 (Alvor); 7-10 February 2013 (Arade); 23-25 March 2013 and 20-26 April 2013 (Alvor and Arade). Floristic surveys were conducted following the abundance/dominance scores method (Braun-Blanquet, 1979) by using sampling quadrats of 2 m^2 along selected transects. A total of 112 floristic surveys, amounting to 209 quadrats, were conducted in 15 randomly selected saltmarsh areas (Table 4). The botanical nomenclature followed the works of Castroviejo et al. (1986, 2007), Franco (1971, 1984); Franco and Rocha Afonso (1994, 1998, 2003) and Rivas-Martínez et al. (2002). The degree of presence (Braun-Blanquet, 1979) was calculated to evaluate the differences in species richness between saltmarsh areas (Table 4). The presence is estimated in percentages of a species and classified according to a chosen scale into a set of 'classes of presence'. The classes were defined as follows (Costa et al., 2009b): *r* (< 6%); + (6-10%); *I* (11-20%); *II* (21-40%); *III* (41-60%); *IV* (61-80%); *V* (> 81%).

2.4 Results

The analysis of land cover c. 1800 (Figure 6a) indicates that wetlands have been progressively urbanized in the area of Portimão (Figure 6b). Geoprocessing of historic maps revealed that saltmarshes occupied an area of 242 ha c. 1800. Analyses of aerial photography indicate that in 2010 the total area of saltmarshes was 118 ha. Therefore, 51% of saltmarshes in the Arade River mouth were lost between 1800 and 2010. Further analyses taking into account other areas along the Arade River indicate that in total 65% of the saltmarshes were lost between 1800 and 2010. Reclaimed areas along the river Arade correspond to excavated old marshes, filled with estuarine mud and protected from tidal influence, with the purpose to create land for agriculture. Estuarine muds, nutrient-rich, were used to fill the reclaimed saltmarsh and build dykes structures upstream the Boina and along the left shore of the Arade River (Vieira, 1911).

Figure 6. a) Arade River and surroundings in c. 1800 (Baltazar de Azevedo Coutinho, Geographic Military Institute-IGEOe), b) saltmarshes of Arade River in c. 1800 over ortophotomap of 2010

2.4.1 Land-use changes between 1958 and 2010

Land-use changes in the period 1958 to 2010 highlight major transformations in the wetlands (Figure 7), especially concerning saltmarsh reclamation for diverse economic activities (Table 3) and natural ecosystem distribution and composition. Table 3 shows the distribution in area of the saltmarsh sub-types and land uses in the Alvor estuary and Arade River identified though geoprocessing and photointerpretation. Dykeland refers to areas enclosed by dykes and embankments, built to prevent tidal incursion.

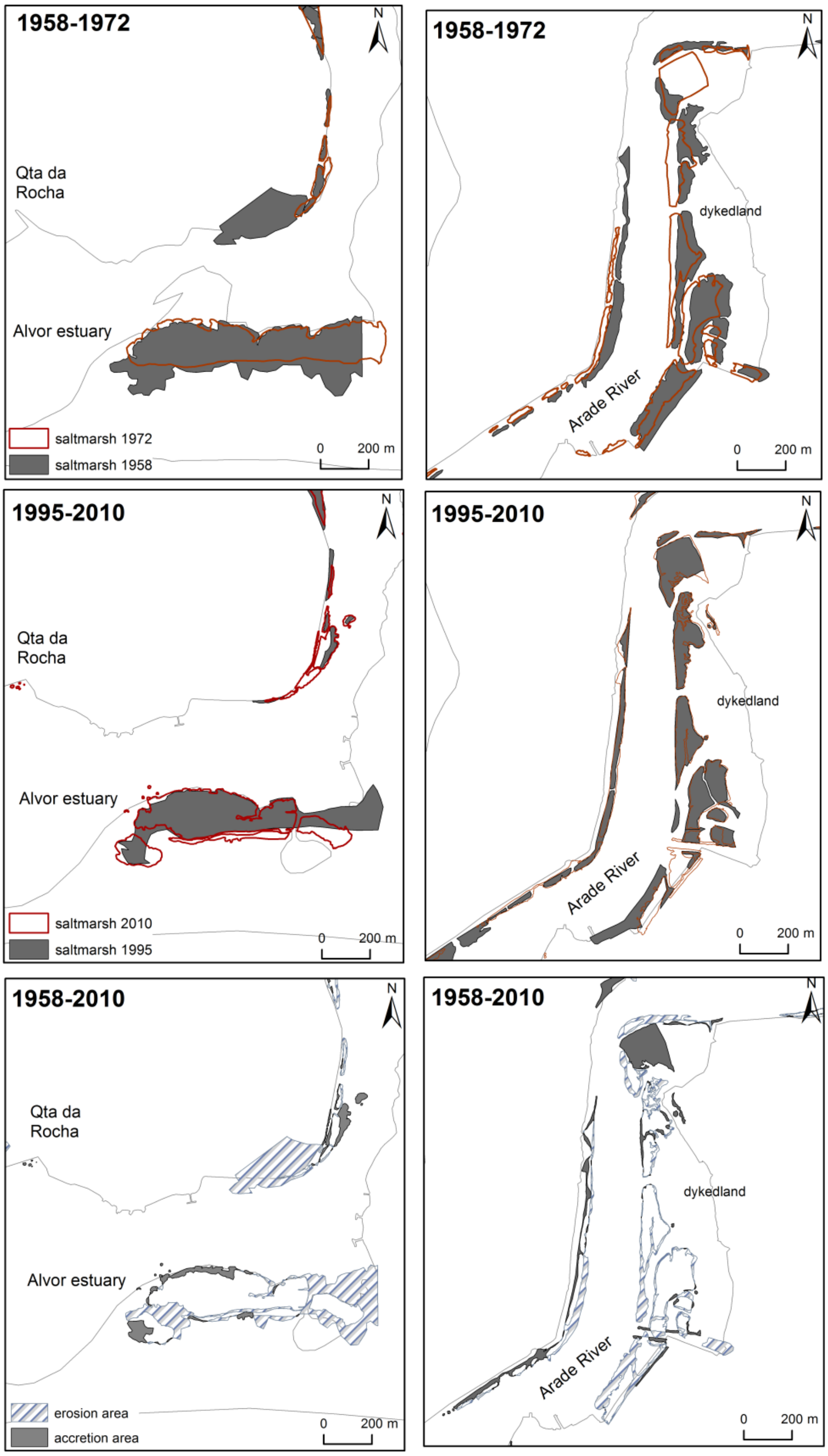

Figure 7. Examples of saltmarsh changes in two of the studied periods (1958-1972 top and 1995-2010 middle) in a section of the Alvor estuary (left) and Arade river(right). At the bottom, results from spatial analysis show saltmarsh erosion and accretion areas in 1958-2010.

Table 3. Land-cover/land-use changes over 1958 to 2010 in Alvor and Arade

Covered area (ha)

Land cover	**1958**		**1972**		**1987**		**1995**		**2005**		**2010**	
	Alvor	**Arade**	**Alvor**	**Arade**	**Alvor**	**Arade**	**Alvor**	**Arade**	**Alvor**	**Arade**	**Alvor**	**Arade**
saltpans (working)	39	34	48	38	43	40	30	36	30	12	29	12
saltpans (abandoned)	0	0	0	0	24	0	44	0	25	22	36	4
aquaculture	0	0	0	0	0	0	9	11	21	29	20	48
saltmarsh early reclamation	44	120	0	0	0	0	0	0	0	0	0	0
saltmarsh in process of reclamation	41	0	6	70	0	67	0	0	0	0	0	0
salrmarsh (reclaimed)	84	120	161	70	45	67	24	13	55	27	47	49
saltmarsh (natural)	182	196	73	129	69	111	55	109	65	85	72	118
saltmarsh (dyked)	0	48	35	66	5	5	10	17	5	17	24	25
saltmash (recovered)	0	0	0	0	0	0	1	24	1	1	16	14
saltmash (eroded)	0	3	1	0	3	2	1	0	0	0	0	13
dykeland	134	211	128	153	125	151	2	36	2	70	65	30
dykeland (agriculture)	0	0	0	0	117	0	123	0	14	0	0	17
dykeland (abandoned)	0	0	0	0	24	0	243	121	237	139	188	167

One of the major land cover transformations was the saltpans abandonment. These complexes of salt production generally occupied large areas. The abandonment of the salt production occurred as the traditional method for food preservation became less important, decreasing significantly the commercialization and export of salt (Rau, 1951). As a result saltpans were transformed into aquaculture production units starting around the mid-1980s in the Alvor and in the early 2000s in the Arade. By 2010, about 59% of saltpans were abandoned in the Alvor estuary and 85% in Arade river.

In 1958 natural saltmarshes occupied a slightly larger area in Arade River (196 ha) than in the Alvor (182 ha), this relative proportion was maintained throughout the studied period. The Agriculture Development Plan was responsible for a 90% of the saltmarsh reclamation in both areas. Despite the decrease on natural saltmarshes between 1958 and 1995, a slight increase was observed in the subsequent years (Figure 7). The recovery dynamics started to be noticed in 1995 with an average of 9 ha/year of new saltmarshes in the Alvor and 13 ha/year in the Arade during 1995-2010. There was a great overall loss of saltmarsh areas in the period 1958-2010 (61% in the Alvor and 40% in the Arade). However, the loss would have been even greater without the recovery of saltmarsh areas observed between 1995 and 2010, resulting in the creation of 18 ha of new saltmarshes in the Alvor and 39 ha in the Arade.

Along the Arade River 48 ha of saltmarshes were already dyked in 1958. In 1987 agriculture within dykelands grew the most causing a great decrease in saltmarsh extent within embanked land. Until 1987, the loss of dyked saltmarsh represented the total saltmarsh area that had been destroyed. Over the period 1958-2010, 47% of saltmarshes were dyked along the Arade River. Only 19% of saltmarshes were diked in the Alvor estuary reflecting a more effective process of reclamation in Alvor.

Saltmarshes dynamics is not independent of other environmental and anthropic factors. Most changes in saltmarsh are associated with small local activities or development. For instance, the 13 ha of eroded saltmarshes quantified between 1995 and 2010 in the Arade River was caused by the construction of a touristic marina. Between 1958 and 2010, there was a net saltmarsh loss of 73 ha in the Alvor estuary, resulting from 98 ha of eroded saltmarshes and 25 ha of restored saltmarshes. For the same period, the Arade River showed a net saltmarsh gain of 12 ha, resulting from 14 ha of eroded saltmarshes and 26 ha of restored saltmarshes.

2.4.2 Saltmarshes typologies

Land-use changes created a process of transformation in which natural marshes gave place to new sub-types (i.e. altered saltmarshes) and/or eventually were completely lost. Figure 8 schematically represents the progressive transformation of saltmarshes caused by reclamation and restoration process, using as example an area along the left shore of Ribeira de Boina. Several marshes have disappeared through this process, at the same time that many other typologies emerged.

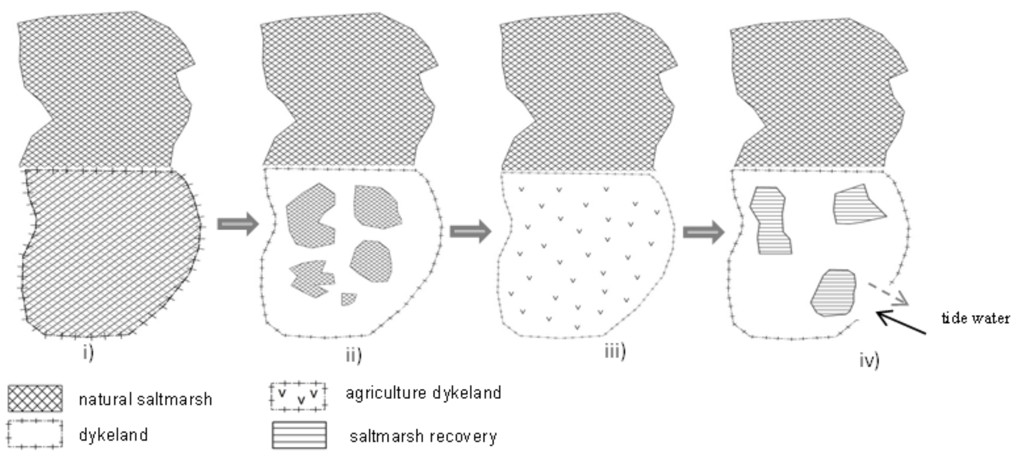

Figure 8. Schematic diagram representing the evolution of the saltmarsh in dykelands: i) the dyke was built in a natural saltmarsh; ii) the saltmarsh started to defragment and progressively disappears; iii) after the drainage and the exsiccation, agricultural dykeland is ready to be at work; iv) dyke wall broke due to abandonment of the agriculture activity, the tide water comes in, providing a low energy environment, allowing the marsh to evolve: an ecosystem recovery takes place.

Based on photointerpretation of images dating from 1958, 1972, 1987, 1995, 2005 and 2010 and quarterly reports that accompanied the land reclamation process until 1987, six stages of transformation or saltmarsh sub-types were identified:

1. *natural saltmarshes:* all saltmarshes that were not dyked, destroyed or damaged;
2. *early reclamation*: the saltmarsh was dyked and started to fragment, usually observed in the period 1958-1972, when the Agriculture Development Plan was being implemented;
3. *fragmented saltmarsh:* at the final stages of the reclamation process, the saltmarsh is clearly fragmented and occupying a smaller area than in the *early reclamation* stage. The fragmented stage was observed until 1987; therefore, saltmarsh transformation from ‘early reclamation’ to ‘fragmented’ in the study area occurred in a time-frame of around 15 years.
4. *dyked saltmarshes:* saltmarshes are enclosed within embankments/dykes and completed isolated from tidal influence;
5. *recovered saltmarshes* corresponding to patches that were affected by land reclamation but were naturally recovering outside dyked areas; and
6. *eroded saltmarshes*, these refer to saltmarshes that were permanently lost.

In the study area, *dyked saltmarshes* naturally evolved into two distinct typologies based on their floristic structure: (1) *enclosed mix marshes*, formed by patches of brackish, freshwater and some invasive species developing due to saline intrusion in areas where dykes have not been breached; and (2) *tidally-restored saltmarshes*, formed in areas where dyke breaching allows incursion of tides and development of a floristic structure similar to natural saltmarshes.

2.4.3 Floristic characterization of different saltmarshes

Table 4 lists the 15 locations where floristic surveys were conducted and indicate the dominant saltmarsh sub-type (or stage of transformation) present in each area. The surveys were conducted in randomly selected areas of 14.9 m^2.

Table 4. Floristic surveys places by stage of saltmarsh transformation, location description and coordinates.

Stage of saltmarsh transformation	**River / Estuary**	**Name of the place (village or site)**	**Coordinates (WGS 84)**
(1) natural saltmarshes	*Arade*	Marine of *Mexelhoeira da Carregação*	N 37º08.910´ W 008º30.311´
(1) natural saltmarshes		*Companheira* site	N 37º09.586´ W 008º31.376´
(1) natural saltmarshes		Arade tributary: *Ribeira de Boina*	N 37º10.280´ W 008º31.861´
(3) fragmented saltmarsh	*Alvor*	Right side of *Odiáxere* creek	N 37º07.991´ W 008º37.440´
(2) early reclamation		West of the Alvor village	N 37º07.596´ W 008º35.871´
(4) dyked saltmarshes	*Arade*	Dyked marshes at the left side of Arade River	N 37º09.272´ W 008º30.042´
(4) dyked saltmarshes		*Morgado de Arge* (Boina creek)	N 37º10.335´ W 008º31.985´
(4) dyked saltmarshes		*Companheira* dykedland	N 37º09.615´ W 008º31.372´
(4) dyked saltmarshes	*Alvor*	*Quinta da Rocha*	N 37º08.056´ W 008º36.939´
(6) eroded		Southwest Alvor estuary	N 37º07.325´ W

saltmarshes			008º38.091´
(4) dyked saltmarshes	*Arade*	*Companheira* inland	N 37º09.656´ W 008º31.382´
(5) recovered saltmarshes		Upstream left side Arade	N 37º09.696´ W 008º29.992´
(5) recovered saltmarshes	*Alvor*	Former agriculture dykeland upstream Alvor River	N 37º08.537´ W 008º35.841´
(5) recovered saltmarshes		*Maria Pires*	N 37º08.607´ W 008º35.464´
(6) eroded saltmarshes		Northeast *Quinta da Rocha*	N 37º08.185´ W 008º37.119´

The degree of presence calculated in each of the 112 floristic surveys is shown in Table 5 for *natural saltmarshes* and the two typologies of *diked saltmarshes* identified in this study (i.e. *tidally-restored* and *enclosed mix*). Surveys allowed differentiating the floristic structure and composition of a natural saltmarsh from the other typologies. The floristic composition of a *natural saltmarsh* indicates low, medium and high marshes clearly structured in relation to the tidal levels. The floristic composition of *tidally-restored saltmarshes* varies depending on the elevation in relation to the tidal range. As the tidal flow is confined by the breach, the volume of tidal flow and the relative distance from the dike breach determines whether certain areas are subjected to inundation. *Bolboschoenus maritimus* var. *compactus, Halimione portulacoides* and *Sarcocornia perennis* ssp. *perennis* appear near the dyke breach (low marsh); *Sarcocornia alpini, Sarcocornia pruinosa, Limonium algarvense* are present at the medium marsh; *Limoniastrum monopetalium, Suaeda vera, Inula critmoides* (high marsh), occur near the paths or the upper dykes. *Enclosed mix marshes* are not subjected to tidal inundation.

Table 5 indicates a loss of biodiversity at *tidally-restored saltmarshes* in comparison with *natural saltmarshes*, as can be seen by the reduction in the number of *taxa* and species observed in the surveys. *Natural saltmarshes* have higher number of *taxa* in all low, medium and high marsh areas (in relation to the tidal range) than *tidally-restored saltmarshes* (Table 5). In both typologies, low marshes show considerably lower number of *taxa* than medium and high marshes (Table 5). *Enclosed mix marshes* show a distinguished composition and structure influenced by saltwater intrusion. These marshes are marked by the dominance of freshwater over brackish species (as freshwater input from rainfall is sufficient to last throughout the winter) and the presence of invasive (*Cotula coronopifolia, Carpobrotus edulis, Oxalis pes-caprae*) and terrestrial species (*Hypochaeris radicata, Polypogon maritimus, Lotus creticus, Sedum sediforme).*

Table 5. Degree of vegetation presence in each marsh typology according to the scale: r (< 6%); + (6-10%); I (11-20%); II (21-40%); III (41-60%); IV (61-80%); V (> 81%)

	Habitat						
Total surveys by saltmarsh typology	**Natural saltmarsh** (46)			**Tidally-restored saltmarsh** (32)			**Enclosed mix marshes** (34)
No. Surveys in relation to tidal level	low 16	medium 15	high 15	low 10	medium 12	high 10	-
Species Number of *taxa* present	10	17	27	9	14	14	35
Spartina maritima	**III**						
Bolboschoenus maritimus var. *maritimus*	**I**						**r**
Atriplex prostrata	+						**I**
Suaeda albescens	**I**	**I**	**II**				
Sarcocornia perennis ssp. *perennis*	**III**	**I**		**VI**	**VI**		**VI**
Sarcocornia perennis ssp. *alpini*	**I**	**II**		**VI**	**VI**	**III**	+
Sarcocornia pruinosa	**I**	**III**	**II**	**I**	**III**		**I**
Cistanche phelypaea	**II**	**II**	**III**		**II**	+	**II**
Halimione portulacoides	**V**	**V**	**II**	**VI**	**V**	**VI**	+
Puccinellia iberica		+	+	**II**	**I**	+	+
Puccinellia maritima		**I**	+	+			
Arthrocnemum macrostachyum		+	**II**		**II**	**III**	**r**
Suaeda vera		**I**	**IV**				
Spergularia media		**I**	**I**				
Limonium algarvense		**I**	**II**				**r**
Spergularia bocconei		**I**	**I**				
Limonium lanceolatum		+	**I**		+		
Juncus maritimus		**II**		+	**I**		+
Limonium vulgare		+					
Limoniastrum monopetalum			**III**			**II**	
Inula crithmoides			**II**		**I**	**III**	+
Aster tripolium ssp. *pannonicus*			**I**				**r**
Oxalis pes-caprae			**I**				**I**
Salsola vermiculata			**I**				**III**
Myriolimion diffusum			**I**				
Atriplex halimus			**I**				
Calendula arvensis			+				**r**

	Habitat						
Total surveys by saltmarsh typology	**Natural saltmarsh** (46)			**Tidally-restored saltmarsh** (32)			**Enclosed mix marshes** (34)
No. Surveys in relation to tidal level	low 16	medium 15	high 15	low 10	medium 12	high 10	-
Species Number of *taxa* present	10	17	27	9	14	14	35
Carpobrotus edulis			+				**r**
Elymus farctus			+				**r**
Emex spinosa			+				+
Ferula tingitana			+			+	**VI**
Medicago polymorpha			+				**r**
Sonchus maritimus			+			+	
Frankenia laevis			+	**r**	**II**		**I**
Bolboschoenus maritimus var. *compactus*				+			**r**
Aster tripolium ssp. *pannonicus*					**II**		
Scorpiurus vermiculatus					**I**	**III**	**r**
Elytrigia elongata					+	**I**	
Artemisia gallica						+	**II**
Juncus acutus						+	**r**
Salicornia ramosissima						+	**II**
Artemisia crithmifolia							**r**
Atriplex hastata							**r**
Cotula coronopifolia							**VI**
Hypochaeris radicata							**I**
Lotus creticus							+
Melilotus segetalis							**r**
Polypogon maritimus							**II**
Salsola sola							**I**
Sedum sediforme							**II**

2.5 Discussion

Land-use changes commonly result in erosion and degradation of marshes (Bertness & Sillimanm, 2008). Natural saltmarshes are resilient to various disturbances in their ecology or ecosystems function (Roebeling et al., 2013) and they can respond positively to sea-level rise (Moreira, 1987). However, human occupation in coastal areas and the presence of hard engineering structures prevent inland migration of the saltmarsh ecosystems. Economic interests have been more important than the value or services offered by saltmarshes. Historically, coastal wetlands were seen as land available at low price with a privileged location.

The Alvor dykes were built in the early 17th century with the objective of supporting agricultural practices, e.g. by allowing freshwater farm irrigation from the Alvor tributaries to the Mexelhoeira Grande and Arão (Mariano, 2010). In Alvor, it was very common to artificially control tides along the tributary Ribeira de Odiáxere, in order to reclaim wetlands for agriculture. Two saltpans existed in Portimão; one at west of the urban center and the other in the north riverside, which progressively expanded towards Ribeira de Boina (Arade tributary).

Saltmarsh reclamation has occurred over a long period in the study area, but the greatest impacts arose by the implementation of the Portuguese Agriculture Development Plans (1953-1964), which targeted investments to stimulate economy. These Plans were implemented in two phases. The first phase (1953-1958) prioritized saltmarsh reclamation to create agricultural land through the construction of small embankments, clearing the original vegetation and irrigation with freshwater. These activities provided the leaching needed to avoid saltmarshes communities to grow. The second phase (1959-1964) focused on the exsiccation of dyked saltmarshes by trenching. Land reclamation in study area ceased mainly due to the failure of the Portuguese Agriculture Development Plans.

In many locations, the practices introduced by the Portuguese Agriculture Development Plans resulted in the destruction of marshlands beyond chances of restoration. Land reclamation for agricultural purposes was very successful on upstream saltmarshes, where freshwater streams feeding the Alvor estuary (i.e. Odiáxere, Arão, Penina) were able to reduced saltwater intrusion. Currently, marsh vegetation is not seen in these agricultural fields (agriculture dykeland). Recovery of saltmarshes are highly hindered in areas where the abandonment of commercial salt production has been replacement by aquaculture and in areas developed for urban-tourist uses (Gedan et al., 2009).

In other locations, the great exsiccation works had limited success in creating agricultural areas. In the Alvor estuary, the influence of saltwater intrusion allowed *enclosed mix marshes* to develop inside the dykelands. An important factor for the emergence of *enclosed mix marshes* is the Mediterranean climate dry conditions, which favor salt intrusion by capillarity, as observed in the areas of Maria Pires and Quinta da Rocha in the Alvor estuary. In the Arade River, the failure of agricultural dykelands is related to the degradation of the embankments or the destruction of floodgates, after economic practices (e.g. grazing, agriculture or salt production) were abandoned. In these areas, the natural breaching of abandoned embankments restored tidal flow into the dykelands providing opportunities for *tidally restored* saltmarsh communities to develop. The restoration of saltmarsh communities was favored especially in areas of high sedimentation rates.

Surveys show significant differences in floristic composition between the primary (*natural marsh*) and the *enclosed mix marshes* that grow inside dykelands. *Enclosed mix marshes* are formed by patches of brackish species (*Haliminone portulacoides*, *Sarcocornia pruinosa, Arthrocnemun macrostachium*), within freshwater communities (*Puccinellia iberica, Juncus accutus, Bolboschoenus maritimus* var. *compactus*) presenting some invasive species (*Cotula coronopifolia, Carpobrotus edulis, Cistanche phelypaea, Oxalis pes-caprae* (e.g. observed at Maria Pires saltmarsh in the Alvor). Mossman et al. (2012b) suggest that "*it is unlikely that restored marshes with very different vegetation will be functionally equivalent*" and discusses how floristic differences might affect different functions. Spencer & Harvey (2012) indicate that saltmarshes restored through active management (e.g. managed realignment) "*may be significantly impaired"* in their capacity to deliver "*ecosystem services including biodiversity, climate regulation and waste processing*".

Despite being impaired in their potential functionality when compared to natural saltmarshes, secondary saltmarshes (e.g. *tidally-restored* or *enclosed mix marshes*) are still able to offer some ecosystem services. It is therefore required that further studies are able to quantify how functionally impaired restored saltmarshes might be and whether there are any ecosystem services they might not be able to provide. Nevertheless, especially for saltmarshes restored unmanaged, it might be more pertinent to compare their capacity to provide ecosystem services in relation to the land-use/land-cover type they have replaced (e.g. abandoned agricultural land), rather than their equivalence to natural habitats. For example, *enclosed mix marshes* tend to be sheltered, providing conditions for birds and other animals to feed and refuge from predators; additionally, the saltmarshes enhance the ability to offer storm protection. These services are better provided by the restored saltmarshes than the previous land cover/use. Further understanding the evolution and functionality of restored saltmarshes and the ecosystem services they are able to provide is required to inform coastal managers on the benefits gained (and lost) from this unmanaged land use/cover change.

Regardless of the great land use transformations described here, *natural saltmarshes* continue to exist in both the Alvor and Arade study areas. Facing this great diversity of saltmarshes typologies, dealing with habitat management is truly a challenge. *Enclosed mix marshes* and *tidally-restores saltmarshes* have different recovery rhythms mainly influenced by the previous land cover (economic activity) or by the stage of transformation at the time of abandonment. The saltmarshes typologies differentiation revealed extremely useful to categorize vegetation succession, which former studies fail to address. The analysis of the effects of past anthropic occupation revealed the resilience of saltmarshes and their ability to recover without artificial intervention.

2.7 Conclusion

Persistent human occupation and land-use changes in Portugal, rather than sea-level rise or other environmental changes, have caused the greatest impacts in coastal ecosystems in the last two centuries. This study quantified losses and gains in saltmarsh areas and identified typologies resulting from responses to land-use changes from 1800 to 2010 in the Arade River and Alvor estuary (Algarve, southern Portugal). A custom methodology involving a combination of spatial analysis (based on historic charts and aerial photographs) and vegetation surveys was used to quantify changes and identify saltmarsh typologies.

Land reclamation for agriculture was promoted by national policies in the 1950s and 1960s resulting in an overall loss of 763 ha (85%) of saltmarshes in the study area between 1958 and 2010. Six stages of transformation were identified associated with the response of natural saltmarshes to land reclamation. In the mid-1960s, economic drivers led to the abandonment of agricultural practices allowing saltmarsh recovery in reclaimed areas. Local particularities resulted in the (unmanaged) development of two distinguishable saltmarsh typologies: (1) *enclosed mix marshes* formed by patches of brackish, freshwater and some invasive species developed due to saline intrusion in areas where dykes have not been breached; and (2) *tidally-restored saltmarshes* formed in areas where dyke breaching allowed incursion of tides and the development of a floristic structure similar to natural saltmarshes.

Over the 1958-2010 period, saltmarsh recovery in the Arade River is dominated by *tidally-restored saltmarshes*, while the Alvor estuary *enclosed mix marshes* dominate. Identifying the different typologies of secondary saltmarshes helps inform management practices aiming to enhance environmental benefits. For example, it is necessary to identify and quantify the capacity of different typologies to deliver ecosystems services in order to inform management efforts targeting enhancement of specific ecosystem services (e.g. carbon sequestration, flood risk management etc.). In this regard, for example, studies focusing on sediment dynamics in areas of *tidally-restored saltmarshes* and saltwater intrusion affecting *enclosed mix marshes* would contribute to further understand the processes involved in the restoration of ecological functions in the study area and elsewhere.

The methodology used in this study proved appropriate to assess the response of saltmarshes to land-use changes. The six transformation stages and the two typologies of secondary saltmarshes identified in this study are likely to be found in other areas of Mediterranean climate (e.g. in Portugal, Spain and France). This methodology can be easily applicable to other locations worldwide providing land-use changes have been recorded through time (e.g. in historic charts).

Chapter III – Sediment characteristics contributing to the development of naturally recovering marsh within reclaimed areas

SEDIMENT CHARACTERISTICS CONTRIBUTING TO THE DEVELOPMENT OF NATURALLY RECOVERING MARSH WITHIN RECLAIMED AREAS

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3.1 Abstract

Sea level rise and human activities pose increasing pressure on coastal habitats, leading to loss and degradation of ecosystems, such as saltmarshes. Growing efforts are observed in Europe and elsewhere to restore or recreate saltmarshes. This study compares the characteristics of natural saltmarshes with two typologies of saltmarshes which are recovering naturally in previously reclaimed land where agricultural practices have been abandoned for 30-40 years in the Algarve (South Portugal). Short-term sedimentation rates were measured using marker horizon and erosion stakes in 16 cores collected in the Laguna de Alvor and the Rio Arade. Grain size distribution and organic matter concentration were analysed to understand the interactions between sediment processes and floristic composition. Accretion dominated in the study areas with rates varying between 0.2 mm/year and 15.53 mm/year. Compared with local natural marshes, the recovering marshes tend to show lower accretion rates, a wider variation in grain size distribution and differences in floristic composition. Natural marshes were either dominantly sandy or clayey, while restored marshes had more variable percentages of sand, silt and clay. Enclosed mix marshes show greater floristic diversity, contrasting with natural marshes, which show species composition typical of either low or high saltmarsh communities. Results indicate that differences in tidal flow and the associated changes in sediment characteristics (which are also influenced by localized effects) are the main drivers leading to the floristic composition variations between the three saltmarsh typologies. Better understanding of the characteristics of saltmarshes that are recovering naturally can provide useful insights to assess whether active intervention may be the best way forward at all locations[2].

Key-words: saltmarsh, recovery, accretion rates, grain size, floristic composition

[2] For further data and supplementry information on this chapter, see Appendix B.

3.2 Introduction

Saltmarshes are one of the most productive ecosystems in the world (McLusky & Elliot, 2004; Millenium Ecosystems Assessment, 2005; Deegan et al., 2012). The ecological and socio-economic importance of saltmarshes is widely recognised (Millenium Ecosystems Assessment, 2005; UK National Ecosystem Assessment, 2011). However, despite a range of national and international efforts to protect and restore this habitat and their ability to provide ecosystem services, saltmarsh losses continue worldwide (e.g. Foster et al., 2013). In Europe, managed realignment techniques are increasingly implemented to actively re-create saltmarshes both as a mean to provide more sustainable coastal flooding and erosion risk management and to offset habitat loss required by environmental legislation (Esteves, 2014). There is a growing debate on whether recreated habitats can be considered equivalent to natural habitats (e.g. Garbutt & Wolters, 2008; Mossman et al., 2012a; Petillon et al., 2014).

Saltmarshes thrive in sheltered coastal areas where wave energy is low and favours the deposition of fine sediments. Sedimentation is a decisive process for marsh growth, interacting in a positive feedback mechanism. Sedimentation stimulates vegetation development (Temmerman et al., 2005); the presence of vegetation reduces wave and tidal flow energy, which in turn promotes further sedimentation (Möller, 2006). Therefore, a two-way dependence exists between sediment availability and saltmarshes vertical and horizontal accretion (Caçador et al., 2007c; Shen et al., 2008; Reeve & Karunarathna, 2009).

Sediment dynamics within a saltmarsh is influenced by tidal flow volume, sediment availability, original topography and interactions with the vegetation (Allen, 2000; van der Wal & Herman, 2012; Butzeck et al., 2015). Plant morphology influences mean flow speed, turbulence and the vertical speed profile, allowing differences in deposition of suspended sediments within the same saltmarsh system (Allen, 2000). Plant species respond differently to changes in the tidal regimes, affecting the vegetation cover on the marsh surface (Silva et al., 2009; Watson & Byrne, 2009) and contributing to changes in erosion and deposition patterns of different grain size material (De Groot et al., 2011a). The importance of saltmarsh vegetation to the sedimentation process has been studied, among others, by Watson (2004), Salgueiro & Caçador (2007) and Silva et al. (2009).

Local conditions, such as climate, hydrology and geomorphology, influence sediment supply and its temporal and spatial variability, which in turn contribute to differences in marsh elevation (Cahoon et al., 2006). In organogenic saltmarshes, accretion rates are dependent on plant production, while minerogenic saltmarshes are more reliant on sediment trapping. Biomass depends on tidal inundation frequency and duration and salinity. Sedimentation dynamics depends on concentration of suspended sediments, the supply into the marsh and the vegetation structure to promote deposition (De Groot et al., 2011b).

Saltmarsh responses to climate change have attracted increasing research interest (e.g. Kirwan & Temmerman, 2009), especially concerning their capacity to cope with rising sea levels (e.g. Tuner et al., 2000; Kim et al., 2011; Suchrow et al., 2012). Studies have focused both on sedimentation processes to evaluate saltmarshes response to sea-level rise (Cahoon et al., 1995; van Wijnen & Bakker, 2001; Kemp et al., 2010; Cahoon, 2015). In addition to natural processes, anthropic activities contribute in various degrees to the vulnerability of saltmarshes (Gedan & Bertness, 2010).

Improvements of coastal wetland restoration depend on the application of the best available knowledge (Weinstein, 2014). To inform future habitat recreation efforts, it is relevant to better understand the factors contributing to the natural restoration of saltmarshes. The natural recovery of saltmarshes can occur in abandoned reclaimed areas, where saltwater flows are restored through accidental breaching of coastal defences during storms (e.g. Garbutt & Wolters, 2008) or through groundwater saline intrusion due to rising sea levels (e.g. Almeida et al., 2014). Both cases are observed in the Algarve (South Portugal), where agricultural practices in reclaimed land were abandoned in the last quarter of the 20th century and natural restoration of tidal flows resulted in two distinct typologies of saltmarshes as described by Almeida et al. (2014):

(1) Enclosed mix marshes have developed due to groundwater saline intrusion in areas where embankments built to protect the reclaimed land have not been breached. The enclosed mix marshes are formed by patches of saltwater species (*Haliminone portulacoides*, *Sarcocornia pruinosa*, *Arthrocnemum macrostachyum, Cistanche phelypaea*, *Puccinellia iberica*), brackish species (i.e. *Juncus maritimus*), freshwater species (*Juncus acutus*, *Bolboschoenus maritimus* var. *maritimus*) and invasive species (*Carpobrotus edulis*, *Oxalis pes-caprae*, *Cotula coronopifolia*).

(2) Tidally-restored saltmarshes have formed in areas where accidental breaching of embankments, restored tidal flow into formerly protected reclaimed land. The incursion of tides into this sheltered environment facilitates the development of a vegetation structure similar to natural saltmarshes, where *Spartina maritima*, *Halimione portulacoides* and *Sarcocornia perennis* ssp. *perennis* characterise the lower marsh closer to the breach; *Sarcocornia alpini* and *Sarcocornia pruinosa* occur in the medium marsh; *Limoniastrum monopetalum*, *Suaeda vera* and *Inula crithmoides* are observed in higher elevations; and *Bolboschoenus maritimus* var. *maritimus* is observed in areas where freshwater accumulates in the winter (Costa & Lousã, 1986; Costa et al.,1996, 2001, 2012).

This paper advances the understanding of the interactions between sedimentation and the passive recovery of saltmarshes, with focus on the two typologies identified by Almeida et al. (2014) in two locations in the Algarve (the Rio Arade and Laguna de Alvor). A comparative analysis of sediment characteristics and short-term accretion rates allowed identification of factors influencing the differences in vegetation composition between the two typologies and natural marshes. Although this study focuses on naturally occurring saltmarsh recovery, it is anticipated that the evidence provided here will be valuable to inform decision-making concerning artificial saltmarsh restoration.

3.3 Study site description

The Algarve is the only location exposed to Atlantic conditions, where saltmarshes show Mediterranean floristic composition (Costa et al., 2009a). The only variation is the presence of *Sarcocornia pruinosa* instead of *Sarcocornia pruinosa*, which is only found along the Mediterranean Sea as recently demonstrated by Fuente et al. (2013). This study focuses on saltmarshes of two sites in the Algarve, the Rio Arade (river mouth) and Laguna de Alvor (estuary) (see Figure 5, Chapter I).

A detailed description of the two study sites is provided in Almeida et al. (2014). Seven types of saltmarshes (as described by Allen, 2000) are found in the study area (see Table 6 and Figure 9).

Table 6. Correspondence of marsh typology found in study sites according to Allen (2000).

Study site	Sampling area	Marsh Typology	
		Allen (2000)	**Almeida et. al (2014)**
Rio Arade	1	Estuarine fluvial saltmarshes	enclosed mix marsh
	2	Estuarine fringing saltmarshes	natural and tidally restored saltmarshes
	2	Estuarine transitional saltmarshes	natural saltmarshes
	3	Estuarine fluvial saltmarshes	natural saltmarshes
	4	Estuarine transitional saltmarshes	tidally restored and enclosed mix marshes
Laguna de Alvor	5	Estuarine fluvial saltmarshes	enclosed mix marshes
	5	Fringing saltmarshes	tidally restored saltmarshes
	6	Deltaic saltmarshes	tidally restored saltmarshes
	7	Back barrier saltmarshes	natural saltmarshes

3.4 Materials and methods

Terms used throughout this paper follows the work of Nolte et al. (2013): sedimentation process refers to the dry mass of sediment deposited in the marsh surface (measured in g/m^2); accretion is the vertical increase in surface elevation, expressed per unit of time (e.g. mm/year), in relation to a fixed benchmark of known elevation in the strata. Therefore, accretion results from the combination of sedimentation, accumulation of dead biomass, auto-compaction and erosion. Erosion refers to the loss of sediment and the associated reduction in surface elevation, also expressed per unit of time (e.g. mm/year).

3.4.1 Sediment accretion

Survey areas and sampling locations were selected to obtain data from the three different saltmarsh typologies described by Almeida et al. (2014). Hereafter samples collected in areas of these typologies are coded with the letters E, T and N, respectively for enclosed mix marshes, tidally restored saltmarshes and natural saltmarshes, followed by the initials of the study area (AR for the Rio Arade; AL for the Alvor and BO for Ribeira de Boina) as shown in Table 7 and Figure 9. Access to sites was a constraint and prevented data collection of enclosed mix marshes in the Rio Arade. Therefore, accretion rates could not be estimated for this typology and location.

Two methods were used to measure sediment accretion over a period of 203 days between late April and December 2013: marker horizons and erosion stakes. Nolte et al. (2013) describe marker horizons and erosion stakes as methods of good precision and easy implementation to evaluate short-term accretion (months to decades) in saltmarshes. Both methods have been used simultaneously in order to validate accretion rates data (Saynor et al., 1994 *in* Nolte et al., 2013). In this study, erosion stakes were used at sites of known higher energy, where erosion would prevent the use of horizon markers.

Thomas & Ridd (2004) demonstrated the utility and the simplicity of using brick dust as a horizon marker to measure short-term changes in sedimentation rates. Brick dust was used as marker horizon to measure vertical accretion in the study sites. A 2 mm-thick layer of orange brick dust was applied over an area of 0.5m^2 at 9 sampling sites (see Figure 9). Accretion rates were estimated based on the sediment thickness above the brick dust layer sampled in sediment cores taken at the end of the period. The marker horizon method was applied at sites of low energy, where erosion is unlikely to occur. Disadvantages of marker horizons include the possibility of losing material during sampling, the possibility of material redistribution (Moreira, 1992; Nolte et al., 2013) and autocompaction (Van Wijnen & Bakker, 2001).

Stainless steel stakes of 1-m long were used to validate the accretion rates obtained from the marker horizons method, and to ensure data collection at more exposed sites. The stakes (erosion stakes) were installed at 7 sites where tidal dynamics result in overtopping of the saltmarsh and the risk of erosion prevents the use of the marker horizons method. The erosion stakes are visible to people and prone to deliberate removal by passers-by. This disadvantage prevented data collection with erosion stakes in four locations, leading to unequal sampling number in relation to the marker horizon method.

3.4.2 Grain size analysis and organic matter content

A total of 16 cores were collected at the end of December 2013 using a 20-cm long and 4 cm in diameter hand probe. Figure 2 shows the sampling locations. Sediment samples were taken from the cores at 20 cm from the top for grain size analysis. Sample preparation and grain size analysis of the coarse fraction followed the standard sieving process (e.g. Friedman & Sanders, 1978; Tanner, 1995; Dias, 2004a). Sieving allowed to estimate the proportion of sand (0.063 mm to 2 mm), and mud (silt 0.002 mm to 0.063 mm and clay <0.002 mm) in each sample. Detailed analysis of grain size distribution was conducted for the silt fraction using the laser diffraction particle size analyser Malvern 2000, which allowed obtaining the proportion of silt and clay. As samples were mainly gravel-free muddy sediments, the percentages of sand, silt and clay were then displayed in a ternary diagram to obtain the textural classification according to Flemming (2000). The mean particle size value was used to identify differences across saltmarsh typologies and study sites. The percentage of organic matter in each sample was obtained by the loss-on-ignition method, in which an aliquot of 2 g of dried sediment was heated at 500 ºC for about 2 hours. Results were used to classify the samples following the classes proposed by Botelho da Costa (1991).

Figure 9. Sampling locations for marker horizons, erosion stakes, sediment cores and floristic surveys.

3.4.3 Vegetation data

Floristic surveys (Figure 9) were conducted during the local peak standing crop biomass (23-25 March, 20-26 April and 15-18 August 2013), following the method of Braun-Blanquet (1979) by using sampling quadrats of 2 m^2 along selected transects (details are provided in Almeida et al., 2014). Sampling was undertaken at the different time periods to reduce the effects of seasonal variability. A total of 36 *taxa* was recorded (see Table 7) in 16 transects. The locations of the floristic surveys are presented in Figure 9.

The botanical nomenclature used in this study followed the works of Castroviejo et al. (1986-2007), Costa et al. (2012) and Fuente et al. (2013). Following the work of Lepš & Šmilauer (2003) and Portela-Pereira (2013), the abundance-dominance index calculated in this study is a conversion of the Braun-Blanquet (1979) classification scale and uses the values of mean vegetation cover as follows: (5) 87.5%; (4) 62.5%; (3) 37.5%; (2) 15%; (1) 3%; (+) 0.5%; (r) 0.1%.

3.4.4 Statistical methods

Two-way ANOVA test was used to assess whether accretion rates obtained by the marker horizon and the erosion stakes methods are statistically different. To account for the assumption of inherited spatial variability we tested whether each method (independent variable) have a significant effect on accretion/erosion rates (dependent variable). Additionally, the Pearson Correlation Coefficient is used to test whether accretion rates are dependent on elevation, methods (marker horizon and erosion stakes) and saltmarsh typology.

The Canonical Correspondence Analysis (CCA), using the CANOCO4.5 software, was applied to analyse the influence of environmental factors in the floristic composition. Biotic (vegetation) and abiotic data (grain size composition, organic material concentration and accretion rates) were analysed for each saltmarsh typology. The Simpson Index of Diversity (SID) was calculated to assist the characterisation of the vegetation composition in each marsh typology, using the following formula (Capelo, 2003):

$$\text{SID} = 1 - \left(\frac{\sum_i n_i (n_i - 1)}{N (N - 1)} \right)$$

3.5 Results

3.5.1 Short-term accretion

Annual mean accretion rates (mm/year) were estimated through extrapolation of the accumulation data obtained for the 203-day period from April to December 2013. The observation period comprises both the dry and the raining seasons; therefore, it is assumed to be representative of mean annual local conditions.

Table 7. Synthesis of floristic surveys, sediment results and short-term accretion methods, by code area.

	E_AL1	E_AL2	E_AR3	E_AR4	E_BO5	N_AL6	N_AR7	N_AR8	N_AL9	N_BO10	N_BO11	T_AL12	T_AL13	T_AL14	T_AR15	T_BO16
% sand	32.7	17.51	29.43	27.94	10.27	80.52	6.96	3.37	85.21	4.45	3.26	16.75	32.72	5.25	49.98	5.51
% silt	59.66	68.01	59.92	59.54	76.56	17.24	81.03	81.88	13.15	86.96	85.15	68.43	59.49	77.89	43.09	79.33
% clay	7.64	14.48	10.65	12.52	13.17	2.24	12.01	14.75	1.64	8.59	11.59	14.82	7.79	16.85	6.93	15.16
% organic matter	10.24	6.93	4.83	5.9	7.71	3.01	11.56	6.32	2.78	7.05	5.75	16.96	21.21	7.39	3.05	6.04
Elevation (m)	5.1	6.5	5.7	4.4	3.3	8.3	4.5	4.1	9.2	2.7	1.7	5.8	10.6	9.5	3.9	3.9
short-term accretion method	erosion stakes				marker horizon	erosion stakes	marker horizon	erosion stakes		erosion stakes		marker horizon	erosion stakes		marker horizon	marker horizon
extra data on short-term accretion, code and elevation (elev.m)	Erosion stakes: N_AR557 (elev 3.9) ; N_BO519 (elev.3.0); T_AR403 (elev. 9.4); T_AR404 (elev. 9.7) Marker horizon: N_AR435 (elev. 4.3); N_BO506 (elev. 4.1); N_AL495 (elev.10.0); N_AL499 (elev.9.2); T_BO536 (elev. 1.9); T_BO551 (elev. 4.0)															
	I	***II***	***III***	***IV***	***V***	***VI***	***VII***	***VIII***	***IX***	***X***	***XI***	***XII***	***XIII***	***XIV***	***XV***	***XVI***
Spartina maritima (sptma)	.	.	.	.	.	.	.	.	.	.	.	.	.	.	87.5	15
Sarcocornia perennis ssp. *perennis* (sarpp)	.	.	.	.	37.5	37.5	62.5	.	.	.	.	.	.	.	37.5	62.5
Sarcocornia perennis spp *alpine* (sarap)	.	62.5	.	.	15	3	.	37.5	37.5	37.5	.	.	3	.	.	.
Sarcocornia pruinosa (sarpu)	15	37.5	62.5	62.5	.	.	.	3	15	.	.	.	.	.	.	.
Salicornia ramosissima (salrm)	.	.	.	.	0.1	.	.	15	15	.	.	.	.	.	.	.
Halimione portulacoides (halpt)	5	62.5	62.5	37.5	15	62.5	37.5	0.5	.	.	.	.	3	.	.	0.1
Inula crithmoides (incri)	.	37.5	3	0.1	0.1	.	.	.	.	.	.	0.5	.	.	.	.
Arthrocnemum macrostachyum (atmac)	3	.	37.5	.	.	.	.	0.5	.	.	.	3	2	.	.	.
Puccinellia iberica (puci)	0.5	.	37.5	.	.	.	.	.	15	0.5	.	15	.	.	.	.
Limonium algarvense (lmal)	.	.	0.5	.	0.5	.	.	.	.	.	.	.	3	.	.	.
Limonium vulgare (lmvg)	.	.	.	.	.	.	.	.	.	.	.	15	.	.	.	.
Limonium lanceolatum (lmlan)	.	.	.	.	.	.	.	.	.	.	.	.	37.5	.	.	.
Limoniastrum monopetalum (limp)	.	.	.	.	.	.	.	.	.	.	.	15	.	62.5	.	.
Suaeda vera (suver)	.	.	.	.	.	.	.	.	.	3	5	.	3	.	.	.
Suaeda maritima (sumar)	.	.	.	.	.	.	.	.	.	.	.	.	0.5	.	.	.
Artemisia gallica (artga)	.	.	.	.	0.5	.	.	.	.	.	.	.	.	.	.	.

Table 7. Synthesis of floristic surveys, sediment results and short-term accretion methods, by code area.

	E_AL1	E_AL2	E_AR3	E_AR4	E_BO5	N_AL6	N_AR7	N_AR8	N_AL9	N_BO10	N_BO11	T_AL12	T_AL13	T_AL14	T_AR15	T_BO16
% sand	32.7	17.51	29.43	27.94	10.27	80.52	6.96	3.37	85.21	4.45	3.26	16.75	32.72	5.25	49.98	5.51
% silt	59.66	68.01	59.92	59.54	76.56	17.24	81.03	81.88	13.15	86.96	85.15	68.43	59.49	77.89	43.09	79.33
% clay	7.64	14.48	10.65	12.52	13.17	2.24	12.01	14.75	1.64	8.59	11.59	14.82	7.79	16.85	6.93	15.16
% organic matter	10.24	6.93	4.83	5.9	7.71	3.01	11.56	6.32	2.78	7.05	5.75	16.96	21.21	7.39	3.05	6.04
Elevation (m)	5.1	6.5	5.7	4.4	3.3	8.3	4.5	4.1	9.2	2.7	1.7	5.8	10.6	9.5	3.9	3.9
short-term accretion method	erosion stakes				marker horizon	erosion stakes	marker horizon	erosion stakes		erosion stakes		marker horizon	erosion stakes		marker horizon	marker horizon
extra data on short-term accretion, code and elevation (elev.m)	Erosion stakes: N_AR557 (elev 3.9) ; N_BO519 (elev.3.0); T_AR403 (elev. 9.4); T_AR404 (elev. 9.7) Marker horizon: N_AR435 (elev. 4.3); N_BO506 (elev. 4.1); N_AL495 (elev.10.0); N_AL499 (elev.9.2); T_BO536 (elev. 1.9); T_BO551 (elev. 4.0)															
Atriplex halimus (atxha)	.	.	.	.	.	.	.	.	.	.	.	0.5	.	.	.	.
Salsola soda (sals)	.	.	.	.	.	.	.	.	.	.	.	0.5	.	.	.	.
Salsola vermiculata (salvm)	.	.	.	.	.	.	.	.	.	.	.	.	0.5	3	.	.
Chistanche phelypaea (cisph)	.	3	.	.	.	.	.	.	.	62.5	3	.	0.1	.	.	.
Juncus acutus (juac)	.	.	.	.	.	.	.	.	.	.	.	0.5	.	.	.	.
Sonchus maritimus (sonma)	.	.	.	.	.	.	.	.	0.1	.	.	.	.	.	.	.
Cotula coronopifolia (ctcor)	0.1	.	.	.	.	.	.	0.1	0.1	.	3	.	.	.	.	.
Oxalis pes-caprae (oxpes)	.	.	.	.	.	.	.	.	.	.	3	.	.	.	.	.
Carpobrotus edulis (cped)	.	.	.	.	.	.	.	.	.	.	3	.	.	.	.	.
Polypogon maritimus (polm)	.	.	.	.	.	.	.	0.1	0.1	.	3	.	.	.	.	.
Medicago polymorpha (mdpl)	.	.	.	.	.	.	.	.	.	.	3	.	.	0.1	.	.
Scorpius vermiculata (scver)	.	.	.	.	.	.	.	.	.	.	3	.	.	0.1	.	.
Mellilotus segetalis (mlsg)	.	.	.	.	.	.	.	.	.	.	3	.	.	0.1	.	.
Bromus lanceolatus (brlan)	.	.	.	.	.	.	.	.	.	.	.	.	15	.	.	.
Plantago coronopus (plco)	.	.	.	.	.	.	.	.	.	.	0.1	.	.	.	.	.
Hypochaeris radicata (hyrd)	.	.	.	.	.	.	.	.	.	.	3	.	.	.	.	.
Emex spinosa (exsp)	.	.	.	.	.	.	.	.	.	.	0.1	.	.	0.1	.	.
Elytrigia elongata (elyeg)	.	.	.	.	.	.	.	.	.	.	.	15	0.5	.	.	.

Table 7. Synthesis of floristic surveys, sediment results and short-term accretion methods, by code area.

	E_AL1	**E_AL2**	**E_AR3**	**E_AR4**	**E_BO5**	**N_AL6**	**N_AR7**	**N_AR8**	**N_AL9**	**N_BO10**	**N_BO11**	**T_AL12**	**T_AL13**	**T_AL14**	**T_AR15**	**T_BO16**
% sand	32.7	17.51	29.43	27.94	10.27	80.52	6.96	3.37	85.21	4.45	3.26	16.75	32.72	5.25	49.98	5.51
% silt	59.66	68.01	59.92	59.54	76.56	17.24	81.03	81.88	13.15	86.96	85.15	68.43	59.49	77.89	43.09	79.33
% clay	7.64	14.48	10.65	12.52	13.17	2.24	12.01	14.75	1.64	8.59	11.59	14.82	7.79	16.85	6.93	15.16
% organic matter	10.24	6.93	4.83	5.9	7.71	3.01	11.56	6.32	2.78	7.05	5.75	16.96	21.21	7.39	3.05	6.04
Elevation (m)	5.1	6.5	5.7	4.4	3.3	8.3	4.5	4.1	9.2	2.7	1.7	5.8	10.6	9.5	3.9	3.9
short-term accretion method	erosion stakes				marker horizon	erosion stakes	marker horizon	erosion stakes		erosion stakes		marker horizon	erosion stakes		marker horizon	marker horizon
extra data on short-term accretion, code and elevation (elev.m)	Erosion stakes: N_AR557 (elev 3.9) ; N_BO519 (elev.3.0); T_AR403 (elev. 9.4); T_AR404 (elev. 9.7) Marker horizon: N_AR435 (elev. 4.3); N_BO506 (elev. 4.1); N_AL495 (elev.10.0); N_AL499 (elev.9.2); T_BO536 (elev. 1.9); T_BO551 (elev. 4.0)															
Elytrigia juncea spp *borealo-atlantica* (elyjun)	.	.	.	.	0.1	.	.	.	.	.	.	15	.	.	.	.
Elytrigia elongata (elteg)	.	.	.	.	.	.	.	.	.	.	.	15	.	.	.	.

The mean accretion value obtained from the marker horizon method was significantly higher than the mean value obtained from erosion stakes method ($p= 0.653$). There is a higher variability within values of accretion obtained by erosion stakes method than by marker horizon. Variance homogeneity test (Sig 0.071) reveals that there are no significant differences between results obtained from marker horizon and erosion stakes methods (Table 8). Also, the ANOVA test indicates that the variance of values within the methods is statistically significant, which does not happen between methods. This means that measurements within the same method, i.e. marker horizon, are statistically different from each other (Table 8) and therefore there is a great spatial variability in the accretion rates regardless the method used.

Table 8. ANOVA test between (BG) and within groups (WG) of methods (markers horizon and erosion stakes) used in evaluation of sediment accretion.

	Sum of squares	df	Mean square	F	*Sig.*
BG	3.133	1	3.13	0.056	0.071
WG	833.97	15	55.59		
Total	837.10	16			

The highest mean accretion rates were measured in Laguna de Alvor for all typologies (Table 9). Accretion rates in the Rio Arade tend to be the lowest recorded, except for tidally restored marshes, which show lowest accretion rates in Ribeira de Boina. It is noticeable that natural marshes are eroding in the Rio Arade (indicated by the negative mean accretion value), while accretion is observed in all typologies in the other areas. There is a relative high variability in the accretion rates between the typologies. Tidally restored saltmarshes tend to show the lowest variation in the mean rates of accretion ($\sigma 1.03$), while the enclosed mix marshes show the largest variability in mean accretion rates (σ 5.24), with rates varying from 0.0 mm/year in the Rio Arade to 15.53 mm/year in the Laguna de Alvor (Table 9).

Table 9. Mean and standard deviation values of accretion rates per saltmarsh typology and location (mm/year).

Mean values	Ribeira de Boina	Rio Arade	Laguna de Alvor	Overall Mean per typology	***Standard deviation per typology***
Natural saltmarshes	1.33	-8.56	5.19	-0.68	***5.79***
Tidally restored saltmarshes	0.43	0.69	2.72	1.28	***1.03***
Enclosed mix marshes	0.20	0.00	15.53	5.24	***7.27***
Overall mean per location	0.65	-2.62	7.81		
Standard deviation per location	***0.49***	***4.21***	***5.55***		

A positive correlation is found between accretion and elevation, indicating that the higher the elevation, the higher the accretion rates (Table 10). The strongest correlation is found between accretion and saltmarsh typology (0.554) and it is important to highlight that no correlation was found between accretion and selected method, or saltmarsh type and selected method. Through this test, it is proved that the method (marker horizon or erosion stakes) to measure accretion rates do not have a statistical influence on the results.

Table 10. Pearson correlation coefficient (R) and covariance (Cov) calculated using accretion as the dependent variable and elevation, method and saltmarsh typology as the independent variables.

		Elevation	Accretion	Method	Saltmarsh typology
Elevation	*R*	1	0.517*	-0.004	0.044
	Cov	1.505	0.452	-0.005	0.055
Accretion	*R*	0.517*	1	-0.195	0.554*
	Cov	0.452	0.509	-0.139	0.396
Method	*R*	-0.004	-0.195	1	-0.193
	Cov	-0.005	-0.139	1.000	-0.193
Saltmarsh typology	*R*	0.044	0.554*	-0.193	1
	Cov	0.055	0.396	-0.193	1.000

p value 0.816
*inidcates significant correlations

3.5.2 Particle size, organic matter and vegetation

Sandy samples show the widest distribution of grain sizes (poor sorting in comparison with muddy samples), as indicated by the relatively high standard deviation (26.17). Samples from the Arade are richer in clay than those in Alvor, which are richer in sand. The ANOVA test (Table 11) shows a statistically significant difference between the percentage of sand and silt (Sig.< 0.05), but indicating that there are no significant differences among sample sites attributable to the percentage of clay and organic material (sig. > 0.05). The percentage of silt varies largely between sample sites but less so across the different sampling sites. Clay content is better distributed between sample sites and there is a greater homogeneity within clay samples; organic material percentage follows a similar pattern, but variability within sample sites is higher. Considering the F value, it is considerably higher in silt than for clay or organic material, but F value for sand (16.56) demonstrates that the mean share some variability (for ternary diagram see Appendix B).

Table 11. ANOVA test between (BG) and within groups (WG) of grain size and organic material.

		Mean square	F	Sig.
Sand	*BG*	1219.27	16.56	.001
	WG	73.61		
Silt	*BG*	909.17	29.65	.000
	WG	30.66		
Clay	*BG*	28.58	2.54	.118
	WG	11.25		
Organic material	*BG*	32.11	1.86	.214
	WG	17.27		

Correlation between species and environmental variables is higher in axe 1 (0.89). Vegetation explains 10.3% of variance, while environmental variables explain 19.6%. The first axe explains 52% of environmental variables variability. CCA graphic indicates associations between accretion and sand variability, as well as an inverse relation of organic material (O.M.) and enclosed mix marshes (Figure 10).

Natural saltmarshes (N_AL6, N_AL9 and N_AR8) are influenced by variability in silt and clay composition. Species exclusively occurring in natural saltmarshes within the study areas are gathered around this axe, which are species typical of low-medium saltmarshes: *Sarcocornia perennis* ssp.*alpini*, *Sarcocornia pruinosa*, *Puccinellia iberica*, *Arthrocnemum macrostachyum*, *Chistanche phelypaea*. Within the natural saltmarshes group, also T_AR15 shows a linear relation with variations of sand and silt; it is also influenced by low-medium saltmarsh exclusive species, as well as the saltmarsh preferential *Sonchus maritimus*. Canonical analyses showed a direct relation between the variability of accretion and samples N_AL6, N_AL9 and T_AR15, while accretion is indirectly proportional in relation to N_AR8. Concerning the other natural saltmarshes (N_AR7 and N_BO11) sand variability is inversely proportional, allowing the appearance of the low marsh pioneer *Spartina maritima*. Diversity index (SDI) for natural saltmarshes is the lowest of the three typologies (0.8437), while for tidally restored saltmarshes is 0.866.

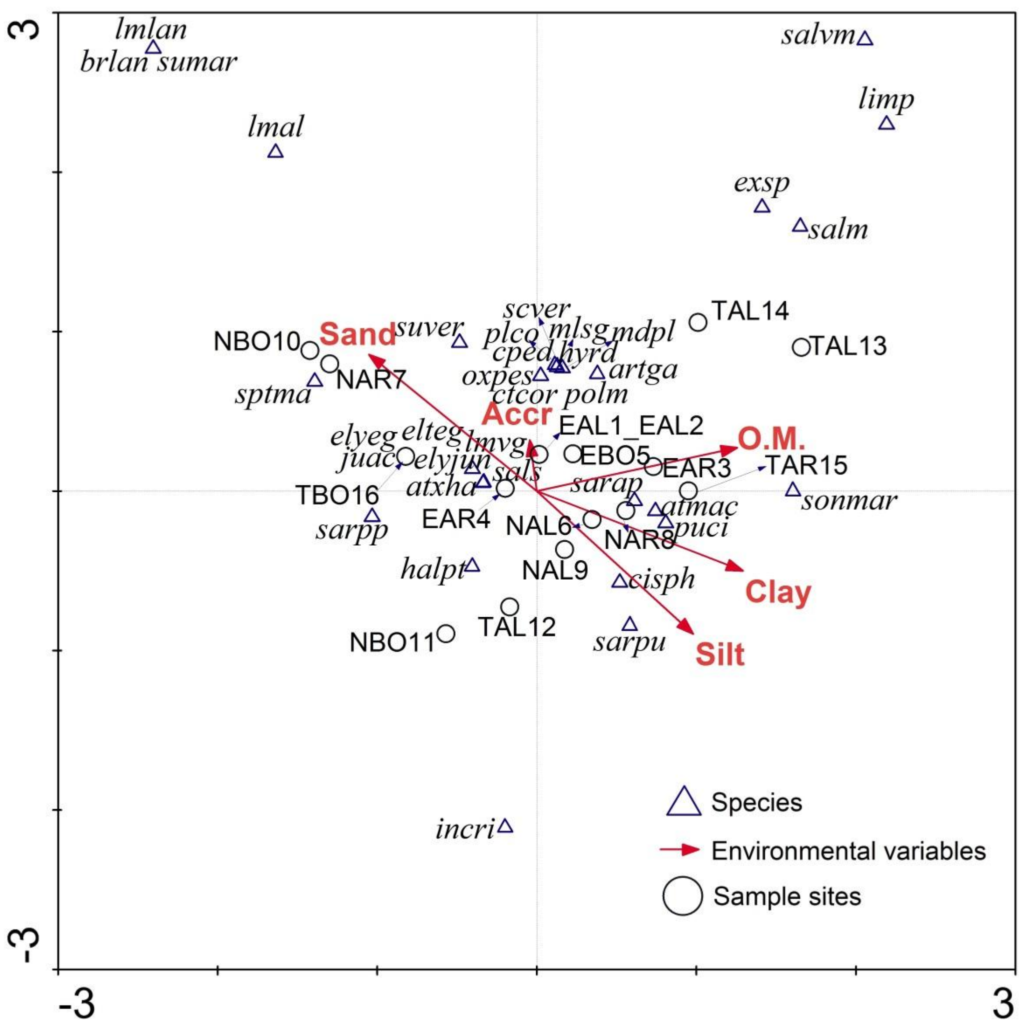

Figure 10. Canonical Correspondence Analysis.

Note: *taxa* correspondence provided in Table 7.

Tidally restored saltmarshes (T_AL13 and T_AL14) show the highest percentages in organic material; floristic composition is typical of upper medium saltmarsh (*Salicornia ramosissima, Salsosa vermiculata*, *Limoniastrum monopetalum*) counting with the presence of the ruderal *Emex spinosa*. The other TRS (T_AL12) is gathered with N_BO11, to which O.M. and clay variations are directly proportional. Vegetation belongs to saltmarsh exclusive species (*Sarcocornia perennis* ssp. *perennis* and *Halimione portulacoides*), but some ruderal species can be found in T_BO16, in which sand is inversely proportional.

Ubiquitous species evolves indifferently to the environment in which they are found; for that reason ruderal species (*Medicago polymorpha*, *Scorpius vermiculata*, *Mellilotus segetalis*, *Plantago coronopus, Emex spinosa*, *Salsosa vermiculata*) are projected into the central axes of the CCA graphic, mostly part of E_AR4. Around these, enclosed mix marshes are plotted together, highlighting the presence of invasive *Cotula coronopifolia*, *Carpobrotus edulis* and

Oxalis pes-caprae frequently develop on areas with anthropic disturbance. Additionally, it is important to highlight the presence of freshwater communities (*Hypochaeris radicata* and *Elytrigia enlongata)* associated with EMM, contributing to a diversity index of 0.886.

Saltmarsh preferential species which develop under a varied range of conditions (*Elytrigia juncea* spp *mariti-atlantica, Polypogon maritimus, Elytrigia enlongata, Artemisia gallica, Juncus acutus, Limonum vulgare* and the exclusive *Atriplex halimus*) are also plotted within the central left side of the graphic, revealing their reduced sensitivity to changes in the environmental variables. *Bromus lanceolatus*, *Suaeda maritima*, *Limonium lanceolatum* are gathered in the upper left axe, but apart from *Limonium algarvense* (endemism of southeast Iberian Peninsula, Costa el al., 2012) and separated from the rest of saltmarsh communities – relation with silt and clay is invasively proportional. Also *Inula crithmoides* is represented far away from sample sites, indicating an inverse variability within accretion.

3.6 Discussion

3.6.1 Tidal flow influence on sedimentation and plant diversity

The statistical tests indicate that the measured rates of short-term sedimentation were not influenced by the type of measurement used, demonstrating that both marker horizon and erosion stake are valuable methods to assess accretion rates. Differences are mainly associated with elevation (and other local conditions not discussed within this scope) and saltmarsh typology. Enclosed mix marshes are the typology showing the highest accreting rates, as shown in Table 4. In general, Ribeira de Boina's saltmarshes show low accretion rates; the lowest rates were found on tidally restored marshes, as these are in elevations relatively high in the tidal frame (3-4 m above mean sea level) and consequently have less tidal influx. However, the higher elevations do not prevent species characteristics of lower natural saltmarshes to occur, as demonstrated by floristic surveys and canonical analysis.

The grain size composition of Arade´s natural saltmarsh samples are similar to other saltmarshes in Portugal, such as the ones described by Moreira (1992), Camacho (2004), Sousa (2006), Caçador et al. (2007a;c), Delgado et al. (2011). However there are considerable differences in the reported short-term accretion for different Portuguese natural saltmarshes: 2.1 mm/year on average at the low marshes of the Sado's estuary (Moreira, 1992; Psuty and Moreira, 2000); 4.6 mm/y to 39.7 mm/year along the Tagus estuary (Salgueiro & Caçador, 2007); and results from this study indicate accretion of 7.81 mm/year at the Alvor, 0.62 mm/year in Riberia de Boina, and a loss of -2.62 mm/year in Arade. Based on these measurements, natural saltmarshes are not accreting in the same proportion of other typologies. Analysis based on an overall mean value to reflect the sedimentation rates of natural saltmarshes in the study area does not give a fair representation of the conditions across all sites. However, modelling results (Teles et al., 2013) show an intensification of tidal flow speed (0.25 to 0.50 m/s) during ebb tide in the Arade (sampling area 2) due to the presence of a bridge. The intensification of the tidal flow is likely to cause erosion of the marsh scarp (loss of 22.55 mm/year) and the resulting loss in vegetation cover, as it has been reported by several authors (Moreira, 1992; Allen, 2000; Marani et al., 2011; Francalanci et al., 2013). Despite local characteristics of saltmarshes, marker horizon and erosion stake methods can be used to evaluate accretion in diverse saltmarsh typologies and for this reason can be applied to other case studies elsewhere.

Different grain size distribution observed in this study drives significant changes in floristic composition between the three saltmarsh typologies (Costa & Lousã, 1989; Costa 2001; Costa et al., 2009a), allowing the identification of bio-indicator species (*Sacocornia perennis* ssp. *perennis*, *Salicornia ramosissima*, *Sarcocornia pruinosa*, *Halimione portulacoides* and the species of *Limonium* genus). Ecological significance of these species are addressed by Costa et al. (2012) as characteristic species of the taxonomic family of *Sarcocornietae pruinosae* which represent perennial saltmarsh communities of moistly soils, rich in sodium salts and flooded daily by salt waters.

Due to its estuarine position, saltmarshes in the Alvor receive a greater input of sand both from marine origin and aeolian transport from the dune system. These considerable sand inputs are spread along the estuary during the incoming tide, reaching mainly the low saltmarshes, that´s why high sand content is found in natural saltmarshes (and also in other typologies) of Alvor. However, this does not compromise the vegetation succession of a natural saltmarsh (low, medium and high saltmarshes species) of Alvor (i.e. back barrier saltmarshes).

Vegetation is assembled around the variance of sand and silt on one hand, and clay and organic matter, on the other. Also, ubiquitous species seams to pose less influence on accretion, while saltmarsh characteristics play a moderate influence on accretion. Floristic diversity was accounted higher for all surveys (from all typologies and locations), but differences arise in enclosed mix marshes registering higher diversity than in tidally restored saltmarshes. Conditions present at enclosed mix marshes favour the encroachment of species from terrestrial ecosystems surrounding the marsh, resulting in a diverse floristic composition that evidences an environment disturbed by anthropic activities (McDonald et al., 2010). Annual species are found in these areas, especially far away from tidal influence, where a mix pattern of brackish saltmarsh communities and ruderal species find place to grow (Robinson et al., 2004). Tidally restored saltmarshes share similarities in floristic composition and diversity with natural saltmarshes, related to resemble in grain size composition. High abundance of typical species of medium natural saltmarshes (*Sarcocornia perennis* ssp. *alpini*, *Salicornia ramosissima*, *Sarcocornia perennis* ssp. *perennis, Spartina maritima, Halimione portulacoides*) can be found in TRS. Additionally, the presence of *Puccinellia iberica* indicates a reduction in salinity due to less frequent inundation influenced by higher land elevation in former reclaimed saltmarshes (Almeida et al., 2014). Species as *Sonchus maritimus* and *Limoniastrum monopetalum* are frequently found in high natural saltmarshes, especially in saltmarsh edges. These areas are marked by a high floristic diversity, which heavily contrasts with typical monospecific low or high saltmarsh communities (Moreira, 1987; Costa, 2001).

3.6.2 Future evolution of naturally restored saltmarshes

Differences in grain size and vegetation are driven by local characteristics. For example, in Laguna de Alvor saltmarshes are greatly influenced by the input of sand facilitated by the proximity of the dune system and the tidal delta (De Groot et al., 2011b). In the Rio Arade, saltmarshes occur in a more silty-clayey soil, in which saltmarsh pioneer vegetation finds best conditions to develop. Due to greater influence of tidal exchange, tidally restored saltmarshes show sediment characteristics and floristic structure more similar to natural saltmarshes than to enclosed mix marshes. The variation of micro-habitats within enclosed mix marshes favours the development of a combination of freshwater communities, several ruderal species and typical saltmarsh species, which contribute to the observed differences.

Coastal barrier marshes (e.g. in the Alvor) are more likely to be affected by rising sea levels than marshes further inland along an estuary (e.g. the Arade marshes); consequently, they are more vulnerable to erosion (van der Wal et al., 2008; Vandenbruwaene et al., 2011). Results from this study indicate that natural marshes may have a greater chance to avoid submergence than the other two marsh typologies, if sediment input brought by tidal inflow is sufficient to promote accretion rates that exceed the rate of sea level rise. Other factors influencing whether marshes will be able to cope with sea-level rise include land elevation (in relation to tidal levels) and the future changes in sedimentation patterns driven by climate change.

Another potential impact of sea level rise concerns changes in vegetation composition towards a more salt tolerant community (van Wijnen and Bakker, 2001). Although the dry Mediterranean climate may favour the dominance of salt tolerant species, especially in conditions of rising sea levels, this study indicates that changes in sediment grain size will also contribute to specific changes in vegetation composition. The grain size distribution is relatively more important in tidally restored and enclosed mix marshes, where tidal input is more restricted and likely to show gradual change through time. Other studies (e.g. Salgueiro & Caçador, 2007; Silva et al., 2009 and Butzeck et al., 2015) also suggest that sediment characteristics influence the vegetation distribution gradient.

Tidally restored and natural saltmarshes show important differences related to variations in topography (e.g. Brooks et al., 2015). The absence of micro topographic changes in tidally restored saltmarshes (mainly previously agricultural land) is essentially due to soil compaction or presence of internal barriers limiting the development of an adequate drainage network. Topography variation on natural saltmarshes facilitates development of a more efficient drainage network, sediments accumulation and consequent vertical and horizontal accretion (van de Koppel et al., 2005; Vandenbruwaene et al., 2011).

Topographic variability within marshes is fundamental to enhance biodiversity by facilitating the development of a typical succession of low, medium and high saltmarshes, with different tolerances to inundation and salinity (Möller & Spencer, 2002; van de Koppel et al., 2005; Garbutt & Wolters, 2008; Wolters et al., 2008; Vandenbruwaene et al., 2011). In tidally restored saltmarshes, constraints of tidal flow result in low rates of deposition and/or erosion preventing the 'natural' development of topographic variations, even when a drainage network is artificially created (Brooks et al., 2015). Therefore, it is advocated that more intense management practices are required to create the range of elevations required for the development of the full range of marsh communities (Brooks et al., 2015).

In Western Europe, there has been a growing interest in the creation or restoration of saltmarshes to offset loss of designated conservation areas and to promote the delivery of ecosystem services (e.g. Esteves, 2014). Better understanding of the characteristics of saltmarshes that are recovering naturally can provide useful comparison to assess whether active intervention may be the best way forward at all locations. In southern Portugal, 30-40 years after abandonment of agricultural practices, saltmarshes have developed but differ from local natural marshes, mainly as a consequence of restricted tidal flow and low sediment input. Increased flow may occur naturally due to sea level rise and/or degradation of the dykes (which would transform enclosed mix marshes into tidally restored marshes) or artificially due to active intervention (e.g. managed realignment). Studies have shown that restoration of saltmarshes is a very long process and active intervention may not result in ability of the vegetation composition to provide ecosystem services equivalent to natural saltmarshes (Byers & Chmura, 2007; Garbutt & Wolters, 2008; Wolters et al., 2008; Mossman et al., 2012b). New evidence is required to support any claim that active intervention may be justified in sites (such as in the Algarve) where natural processes are leading to passive saltmarsh restoration.

3.7 Conclusions

Results from this study contribute to improve the understanding of the differences in the sedimentation processes occurring in three saltmarsh typologies found in southern Portugal. About 30-40 years after abandonment of agricultural practices in reclaimed land, natural recovery has led to the development of two typologies of naturally recovering saltmarsh, which are characterised and compared with adjacent natural saltmarshes. Although grain size characteristics and rates of vertical accretion were variable across the study sites, differences between the three typologies were identified. In comparison with local natural marshes, the recovering marshes tend to show lower accretion rates and a wider grain size variation. Natural marshes were dominantly sandy or muddy, while restored marshes had more variable percentages of sand, silt and clay. The variations in grain size reflect the restricted tidal flow into the recovering marshes and the influence of sediment input from freshwater or marine sources, which contribute to differences in floristic composition.

The recovering marshes have low accretion rates due to the low sediment input resulting from the shelter offered by the presence of dykes. On the other hand, this shelter effect protects the sites from wave action reducing the risk of erosion, which is observed in natural marshes at more exposed locations. This study indicates a disturbance gradient of tidal flow and salinity contributing to the differences in floristic diversity between typologies. In an enclosed marsh, the saltwater influx is irregular (depending on balances between groundwater infiltration and freshwater input) creating inter-annual and intra-seasonal effects on plant diversity. In years with highest rainfall, the freshwater input may favour germination and plant growth of freshwater species creating competition with brackish communities, and ruderal and invasive species. Topographic variations within enclosed mix marshes create conditions for impoundment of freshwater in places and the influence of saline intrusion in others, promoting the emergence of diverse micro-habitats.

Management practices aiming to increase topographic variations within naturally recovering marshes may be beneficial to create communities with composition and functions more similar to natural marshes. However, improvements on biodiversity and ecological functions achieved in artificially recreated saltmarshes are debated in the literature. Although it is difficult to ascertain based on current data, it is possible that sea level rise, through enhanced saltwater intrusion and/or tidal flow exchange, may facilitate that the two typologies of restored saltmarshes may become more similar to natural saltmarshes with time. In southern Portugal, conditions are favourable to the natural restoration of saltmarshes. It is therefore difficult at present to sustain any argument for investments in active intervention to restore saltmarshes in the study sites and other areas of similar conditions.

Chapter IV – Landscape metrics applied to former reclaimed saltmarshes: a tool to evaluate ecosystem services?

LANDSCAPE METRICS APPLIED TO FORMER RECLAIMED SALTMARSHES: A TOOL TO EVALUATE ECOSYSTEMS SERVICES?

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4.1 Abstract

Analyses on saltmarsh ecosystem services have been particularly focused on the capacity of mitigating climate change effects to cope with rising sea levels and concerning flood management. Nevertheless, ecosystem stability is vital for accurate service delivery, but land-use changes and coastal erosion are affecting saltmarshes. This provides the background for one of the primary arguments for protecting saltmarshes.

Landscape metrics were selected according to shape, complexity, and connectivity parameters, and added to average elevation and distance to the coast, for two years – 1972 and 2010. We developed an equation that measures coastal protection, taking into account the results of PCA and the percentage of explained variation of each component (coastal defence index: ES_CoastDef). Three saltmarshes located in the Algarve region, Portugal, were selected to apply the coastal defence index. Individual patches were analysed according to saltmarsh typologies.

Results revealed that every saltmarsh decreased its coastal defence from 1972 to 2010; changes in shape and connectivity metrics affect mostly the index performance. In 1972, natural saltmarshes offered a better coastal defence than the other typologies, but in 2010 formerly reclaimed saltmarshes comprised higher values of coastal defence. Positive evolutions in terms of reclaimed saltmarshes have enabled them to provide coastal defence ecosystem services. Thus, through this index it is possible to outline target coastal defence parameters and design strategies for their conservation and consider ecological restoration.

Key-words: ecosystem services; landscape metrics; saltmarsh typologies; PCA; coastal defence index

4.2 Introduction

Ecosystem services have been broadly discussed since pioneers (Costanza et al., 1997; de Groot et al., 2002) first introduced the concept of biodiversity economics into the discourse of resource protection and sustainability. When considering giving nature a price, opinions are divided: some authors are for an economic approach (Costanza et al., 2008; Barbiers et al., 2011), while others alert for the need to combine different knowledge bases to understand ecosystem services (Wallace, 2007; Fisher & Turner, 2008). Nevertheless, Costanza et al., (1997) and M.A. (2005) have defined service typologies and managed to differentiate services and functions and to aggregate them in wider sub-ecosystems. This groundwork was the runway for a growing body of literature on ecosystem services theories, valuation systems, and application frameworks: economic theory of benefit estimation (English et al., 2009), valuation methods of ecosystem services (King & Mazzota, 2000); dollar-based ecosystem valuation methods (Costanza et al., 1997; de Groot et al., 2012); ecosystem value estimates – benefit-cost (Loon-Steensma &Vellinga, 2013). All agree that difficulties arise when decisions must be made towards protecting and managing natural resources. Trade-offs underlying allocation ecosystem functions after damage or disturbance are closely related to ecosystem recovery options and to current concerns regarding ecosystem response to climate change triggers (Feagin et al., 2010b) (in which saltmarshes, as coastal fringing ecosystems, are included, especially when considering rising sea levels).

Saltmarsh ecosystem services are largely described in the literature: Simas et al. (2001), Möller et al. (2001), Möller (2006), Gedan & Bertness (2010) and Feagin et al.(2010a) analyze the capactity of saltmarshes to mitigate effects resulting from climate change. The capability of saltmarshes to cope with sea level rise and their relation with vegetation allow for different sucessional and topographic states and therefore absorb various flood levels (Morris et al., 2002; Möller & Spencer, 2002; Feagin, 2008; Watson & Byrne, 2009; Reeve & Karunarathna, 2009; Kim et al., 2011); their natural capacity of regulating climate and plague control (barrier effect) (Gedan et al., 2009; Spencer & Harvey, 2012); nutrient cycling (Deegan et al., 2012); carbon and nitrogen sequestration (Chmura et al., 2003; Caçador et al., 2007b; Reboreda et al., 2008; Mattheus et al., 2010); vegetation attributes for cattle (grazing) (Milotić et al., 2010) and cuisine (*Salicornia* pickles) (Gedan et al., 2009); water storage, refugee habitat for fish, crustacean and birds (English et al., 2009), recreational and cultural services, such as bird watching, cannoning and fishing (M.A., 2005).

Land-use changes, sea level rise and coastal erosion (among other factors) are affecting several wetland ecosystems particularly saltmarshes, and one of the primary arguments for protecting saltmarshes is related to preserving and enhancing the quantity and the quality of the lost saltmarsh ecosystem services (Gedan et al., 2009).

The consistency characteristics of an ecosystem are essential for accurate ecosystem service delivery: features such as elevation, extension and stability in an ecosystem are necessary conditions for the regulating and maintenance of coastal protection services (Liquete et al., 2013), such as wave reduction or flood protection (Loon-Steensma and Vellinga 2013). Saltmarshes have a natural capacity of mitigating sea storms and other extreme climate events, such as tsunamis (Simas et al. 2001; Gedan & Bertness 2010; Kim et al. 2011). Costanza et al. (2008), Barbier et al. (2011), and Roebeling et al. (2013) demonstrate that this attribute results from the extension, exposition and orientation of the coastal system within the coastline. The work of Möller & Spencer (2002) added the level of consolidation of floristic communities (stability and maturity) to the list of coastal wetland attributes, particularly when it comes to saltmarshes, to cope with sea level rise and extreme coastal events. Additionally, a model developed by Brampton (1992) and used by Möller et al. (2001) indicates a reduction by 40% of wave height when crossing an 80 m width whose elevation is below sea level (de-embanked saltmarshes) and patches covered with vegetation. These attributes have contributed to absorb the first impact of wave overtopping and reduce the effect of wave propagation, due to the roughness provided by micro-topographic changes and vegetation cover. The results of this study were applied to evaluate the trade-offs of artificial coastal protection (jetties) in the UK, and supported advances in recreation for coastal defence (Möller, 2006).

When ecosystems are artificially modified to increase a particular service (i.e. coastal defence) ecosystem services are manipulated, but how can these enhancements be evaluated when reference values of those services are lacking? The same occurs when saltmarshes recover without human intervention (abandonment, de-embankment, old reclamations), how can ecosystem services be evaluated? For this it needs to discuss which ecosystem services can be provided by a natural saltmarsh and how these can be evaluated (Costanza et al. 1997; 2014; de Groot et al., 2002, 2010; M.A., 2005).

Moreover, the reclamation process of formerly reclaimed saltmarshes is framed by the work of Bonis et al., (2005), Barkowski et al., (2009), Garbutt & Wolters (2008), Almeida et al., (2014) and linked with recovery theory. The concepts of recovery and restoration used in this paper are from Elliott et al., (2007): for restoration to occur (active process – human intervention) on a degraded ecosystem, it implies discarding mitigation, which will result in an improved habitat or in a new ecosystem, which can then be enhanced in the future; conversely, starting also with a degraded ecosystem, recovery is a passive process, without human intervention, that will result in a new ecosystem. Regarding ecosystem services, Elliott et al., (2007) refer that despite the pre-impact state being frequently unknown, functions, structure, goods, and carrying capacity of the ecosystems may not return to their original state.

Syrbe & Walz (2012) brought important insights regarding spatial indicators for the assessment of ecosystem services, however we believe that some innovative methodological approaches are needed, combining the spatial definition of ecosystem services using landscape metrics in order to inform restoration planning. In this context, reclaimed and naturally recovered saltmarshes (Almeida et al., 2014) were the starting point for this exercise that aims to apply landscape metrics to ecosystem service assessments. In this context, metrics with a coastal protection bias were selected to develop a coastal defence index, based on saltmarsh attributes. The aim is to understand if landscape metrics could be a tool applied to saltmarsh passive recovery in order to assess ecosystem service delivery, specifically the coastal protection attribute of saltmarshes. The index developed to measure the coastal protection offered by a specific saltmarsh is of broad application – it can be replicable to other saltmarshes, regardless of their biogeographic region. The proposed set of data, materials and methods (explained in the third section) are of international application, provided geographic information on the targeted saltmarshes is available. If this condition is verified, it will be possible to combine this tool with ecological restoration strategies, thus reinforcing saltmarsh ecosystem services. It would also provide saltmarshes with the capacity of positively responding to future anthropic disturbances or to cope with rising sea levels.

4.3 Landscape metrics and ecosystem services

Müller & Burhard (2012) discuss the informative power of ecosystem services, mentioning that they have the potential to assess the state of ecosystems, acting as indicators. In this context, landscape metrics will be used to establish quantitative boundaries to inform ecosystem services, with the aim of moving from an economic perspective to effective policies of ecosystem diagnosis, to help decision making and to support recovery and protection measures (Müller & Burhard, 2012). Landscape metrics measure and describe spatial structures such as individual patches, patch types or land mosaics, providing information on landscape composition and configuration (Leitão et al., 2006). Composition and configuration are the two main domains in landscape metrics: while the former (composition) refers to the diversity and abundance of patch types, missing spatial character or arrangement; the latter (configuration) has a strong spatial basis, informing also about position and orientation. These two properties are interactive and complementary and not mutually exclusive (Leitão et al., 2006). The same authors demonstrate the application of landscape metrics to five different phases of the planning process: (1.Focus, 2.Analysis) to further inform the planning system (3.Diagnosis, 4.Prognosis) on ecosystem recovery processes (5.Sinteresis). This justifies the interaction of landscape metrics applied to a formerly reclaimed saltmarsh.

4.3.1 Complexity

Leitão et al., (2006) state that size and shape metrics can be used as proxy information about patch ecological and functional features. These two metrics are also informative on the amount of boundaries shared with adjacent landscape, so they were used to understand complexity. Complex patch shapes have a larger number of edges (boundaries) than simple patch shapes, and in ecological terms, edges induce noteworthy effects on the ecosystem functioning. In this context, MSI (mean shape index), TE (total edge) and ED (edge density) are the main metrics used to assess complexity. Also, fractal dimension is an essential metric to understand complexity, namely through the use of MPFD (mean patch fractal dimension) and AWMPFD (area weighted mean patch fractal dimension): normally, a simpler shape has a lower MPFD value. Since fractal dimension measures the reason between area and perimeter: this is lower for simple shapes such as circumferences, and higher for shapes with many edges.

4.3.2. Connectivity and Fragmentation

Connectivity demonstrates the relation between structure and function, and measures the degree to which the landscape facilitates or prevents the exchange of energy flows, materials, nutrients, species, and individuals among habitat patches (Leitão et al., 2006). Therefore it is crucial to establish the relation between landscape metrics and ecosystem services: a higher connectivity increases the probability of delivering services as it maintains biological diversity.

Doody (2008) discusses important insights into the stability and fragmentation of saltmarshes, based on a balanced concession of accretion and erosion states that occur simultaneously. An abrupt change in the connectivity of these patches may interfere with the existing equilibrium of gains and losses of the saltmarsh platform, probably becoming fragmented into smaller and more isolated patches (Leitão et al., 2006; Doody, 2008). In saltmarsh ecosystems, one of the first signs of fragmentation is the appearance of tidal creeks, which facilitates the development of small pools without vegetation, though the loss of habitat produces a fragmented landscape in which the habitat is divided into multiple small, isolated patches: fragmentation is the result of a degradation or interruption of connectivity. Metrics used to calculate connectivity and fragmentation are the Mean Proximity Index (MPI), which measures the degree of isolation (according to McGarigal & Marks, 1995, the absolute value of MPI has little interpretation, so percentages were calculated for a comparative index); and the Mean Nearest Neighbour (MNN) which indicates the average distance of an individual patch in the shortest distance to a similar patch (edge to edge).

4.3.3 Diversity and Richness

Diversity is an important aspect of an ecosystem balance; lower land cover types (lower diversity) facilitate the propagation of disturbances across the landscape (Leitão et al., 2006). Richness (composition) and evenness (structure) are components of landscape diversity, referring to patch types and distribution area, respectively. Cover type can be a measure for diversity; therefore, Leitão et al. (2006) use CAP (class area proportion) as the main metric to understand landscape composition. Despite having the capacity to evaluate patch type abundance, it does not consider position and distribution along the landscape mosaic. In order to complement CAP, patch richness is used to assess the number of patch types present in each landscape: the link between patch richness and ecosystem services is established by diversity metrics, such as PScoV (Patch Size Coefficient of Variation) and SDI (Shannon´s Diversity Index). These metrics evaluate the relative diversity and the level of proportional distribution of patch types across the landscape.

4.4 Materials and Methods

Here is presented a combined theoretical and empirical study of landscape metrics applied to reclaimed saltmarshes in order to understand if they can be a tool to evaluate ecosystem services within a recovery policy framework. To accomplish these objectives, it has been built a database of metrics on case example saltmarshes and applied a new approach to the coastal defence index formulation, based on landscape metrics performance on a PCA analysis.

4.4.1 Area of application

Three saltmarsh areas located in the Portimão municipality (Algarve region, Portugal) were selected. This selection was based on previous works of Almeida et al. (2014), and on extended fieldwork and the co-existence of saltmarsh typologies (Figure 11). Laguna de Alvor (A), which is classified as a Site of Community Importance by the Natura 2000 network and is an estuarine coastal lagoon system protected from the ocean by two sand spits, to which fresh water input and output is constant. Saltmarshes are of estuarine fluvial and fringing classification according to Allen, 2000; and comprises natural, tidally restored and enclosed mix marshes. Boina creek (B) is the west side tributary of Arade´s basin, with 85 Km2 and 23 Km long. Tested area characterized by estuarine transitional saltmarshes, and all the three typologies of saltmarshes were tested. The Arade estuary is classified as a Wetland of International Importance by the Ramsar Convention and has an area of 966 km2. Estuarine fringing marshes (C) were tested including the natural and tidally restored saltmarshes typologies.

Figure 11. Geographical inbound of tested areas and validation area

4.4.2 Data and methodology used

Spatial analysis has considered parameters presented in Figure 12 and Tables 12, using ArcGIS™10 Software and the reference system used was ETRS89 – PT-TM06. Saltmarsh patch typologies were manually classified, using aerial photo (1972) and ortophotomap (2010). This classification individualized and underlined the contribution of the shape, complexity, and connectivity of each saltmarsh patch to the overall landscape (each saltmarsh ecosystem). The need for almost 40 years of difference between analysis dates is justifiable by the self-organization theory of saltmarshes referenced in the work of van de Koppel et al., (2005). This time-scale can provide evolutionary insights into saltmarsh ecosystem self-organization dynamics – preferably, the time scale should be from 40 to 50 years, but the absence of quality and available data from prior to 1972 inhibits a larger time-scale analysis. Descriptive statistics was calculated to landscape metrics

Table 12. Classification of saltmarsh typologies, patch categories and morphology

Typology	**Marsh morphology**	**Patch categories**	
		Natural patches	***Artificial patches***
Primarily (s)	Low	Saltmarsh	Wall
Tidally restored (t)	Low – medium	Unvegetated	Water mill
Enclosed mix marsh (e)	Medium – low	Water entrance	Channel
	Medium		
	Medium – high		
	High		

Landscape metrics were selected according to shape parameters that could indicate variations in shape, such as the perimeter/area ratio and the mean shape index (van Loon-Steensma et al., 2013). Additionally, metrics on shape complexity, which measures the fractal dimension is critical to provide information about roundness of patches: values of MPFD are lower in circular patches and higher in elongated and straight shapes (Leitão et al. 2006). Therefore, the more complex the shapes become, the more coastal protection they might offer due to the edgy and roughness (Möller, 2006). Connectivity parameters (Mean Nearest Neighbour) that measure the degree of isolation between patches (Leitão et al., 2006) were selected to ascertain the degree of patch isolation: the more isolated saltmarsh patches become, the less coastal protection they may be able to offer, due to fragmentation (Doody, 2008).

Geographical parameters, such as average elevation (Möller et al., 2001) and the distance to the coast (Feagin et al., 2010b), provide insights on local bio-physical specificities. These six landscape metrics (perimeter/area ratio; mean shape index; fractal dimension; mean nearest neighbour; distance to coast; average elevation) share the capacity of being used as proxy information on coastal protection.

Metrics were calculated using Patch Analyst 5, extension for contextualization landscape metrics of connectivity, complexity and diversity/richness of the class and landscape scale. V-Late 2.0 beta extension was used for patch scale metrics, particularly mean proximity index (MPI) and nearest neighbour metrics (MNN). The distance to the coast (D_coast) was calculated using the distance from the patch centroid and the coastline. Only vegetated patches were eligible for coastal defence purposes (Möller & Spencer 2002; Möller, 2006), therefore intertidal areas of mudflat were ineligible for patch analysis (category unvegetated, see table 2).

Aiming to increase the knowledge on the ecosystem services provided by saltmarshes, average elevation (AEv) was added to automatically calculate metrics, reporting to context metrics. Average elevation was calculated manually using differential GPS in a WGS 84 coordinated system, resulting from fieldwork on saltmarsh vegetation, which covered samples for each saltmarsh typology (natural, tidally restored, and enclosed mix marsh) and class area (low, low-medium, medium, medium-high, and high saltmarsh). GPS data are reported to fieldwork carried out during the year 2012, which was used to accommodate the images from 2010 (the nearest year available). To face the lack of data available for 1972, the same elevation was used for both years of analysis. In order to address this limitation, it was decided to assign the value of average elevation for both years, bearing in mind that elevation is a key property for saltmarsh development and dynamics, influencing the dispersion of species and ecosystem morphology.

A methodological validation was tested in an artificially restored saltmarsh (2.1 ha) in order to assess the viability of the coastal defence index. This is the only example of a projected recovery within a tidal habitat in the study area (Figure 11). The project is described as an environmental recovery, framed by the Alvor´s waterfront intervention during the years 1999 and 2000 (Figure 12). Dredging works at the estuary channel and the construction of a recreational marina resulted in the destruction of the medium/high saltmarsh with an average elevation of 3-3.5 meters (Rolo, 2007). The recovery project encompassed topographic correction, halophytes replantation, and hydrodynamic replacing (Rolo, 2007). The same methodological approach was applied to the validation area.

Figure 12. Sequence of images of Alvor´s west saltmarsh area, after deposition of shallow tide sediments, excavation and vegetation destruction (Rolo, 2007).

4.4.3 Statistics

Two approaches were developed in the landscape metrics analysis: descriptive statistics were performed using *IBM SPSS Statistics* software. Deltas, standard deviation and average values are presented in the first section of the results. Fuzzy analysis was performed to normalize the landscape metrics between 0 and 1, corresponding to a variation of continuous data. This was obtained by adjusting each metric to the line, giving the equation of the line. Zero was assumed to indicate the absence of coastal defence, while one was assumed to represent the total coastal defence. Linear fuzzy analysis delivered values approximate to one (in crescent) in the following cases: perimeter/area ratio (PAR), mean shape index (MSI), fractal dimension (MPFD), mean nearest neighbour (MNN) and average elevation (A_elev). Regarding the distance to coast (D_coast) metric, higher values represent a greater distance to the coast, so they were considered approximate to zero.

Afterwards, a principal component analysis (PCA) was calculated using *IBM SPSS Statistics* software. The rotation method selected was the *Varimax* with Kaiser Normalization (rotation converted in five interactions). Since this is a method of orthogonal rotation, it allows maximising the variation between the weights of each principal component. This procedure was repeated for each case example area for the two dates (1972 and 2010). The application of PCA found relevance on the work by Cushman et al. (2008), and allowed to extract the main components that contribute to the construction of the index. Each component assembles a group of landscape metrics and corresponds to a specific explanatory capacity (percentage), which not only provides insights into the most important metrics, but also gives the background calculations for the development of the coastal defence index (equation).

In order to spatialize the coastal defence index and considering the global variability of the data, it was used the quantile method of classification.

For the equation measuring coastal protection (Equation ES_CoastDef) it was taken into account the results of principal components analysis (the value of each landscape metric within the component) and the percentage of variation explained for each component, as follows:

$$ES\_CoastDef = \sum_{i=1}^{n} \left(\sum_{j=1}^{n} V_j W_j \right) W_i$$

Where *ES_CoastDef* is the ecosystem service (*ES*) coastal defence (*CoastDef*), V_j the value of metric *j* (with *j*= 1, …, *n*), W_j the explanation percentage of metric *j*, W_i the percentage of variance of component *i* (with *i*= 1, …, *n*) and, finally, *n* stands for the number of metrics and therefore to the maximum of PCA components.

4.5 Results

4.5.1 Landscape metrics

Composition and diversity metrics

Analysing the three studied cases (Arade, Boina, and Alvor) in the two dates (1972 and 2010) there is a predominance of the number of patches in the low saltmarsh category in the natural and tidally restored typologies. Considering changes from 1972 to 2010, the saltmarsh categories that gained more patches were the tidally restored ones: in Arade t_low gained 31 patches in 2010 and t_high_med gained 7 patches; in Boina, t_low gained 10 patches and t_med_low gained also 11 patches in 2010 (see Appendix C). In Alvor, there was an inverse trend because tidally restored categories, which had been the most dominant in 1972, tended to disappear in 2010. The majority of changes between 1972 and 2010 occurred in Alvor with the emergence of new natural saltmarshes and also the enclosed mix marsh categories.

Table 13. Average landscape alteration (Δ) between 1972 and 2010

1972-2010 (Δ)	NumP	ED	Divers	AWMSI	AWMPFD	MPI	MNN
Arade	4.18	56.65	0.48	0.44	0.17	149.85	19.45
Boina	2.56	46.31	3.40	-0.24	-0.01	95.94	-20.19
Alvor	2.69	29.49	79.59	0.93	0.73	190.77	11.23

There was a gain in diversity in the Alvor case example (see Table 13) but the differences within the classes are higher than in the other landscapes (Standard deviation of 121.93, see Table 14). Arade´s landscape gained a remarkably low value of diversity (0.48), which can be related with the few changes in the landscape that have occurred over the years. The categories found in 1972 are very similar to the ones in 2010, although standard deviation shows a considerable heterogeneity among classes (see Table 14). On the contrary, Boina presents more homogeneity in terms of class diversity (7.55 of standard deviation).

Shape and configuration metrics

Shape and configuration metrics, which are given by complexity, are presented as a combined analysis of mean shape index and mean patch fractal dimension (AWMPFD). Both Arade and Alvor have an average positive value, meaning that these landscape patches have become simultaneously less rounded (AWMSI) and more complex (AWMPFD). In 1972, the most circular shape was given by natural low saltmarsh in Arade and Alvor, but in 2010 the most roundness class was t_low in Arade, and e_low in Alvor (See Appendix C). Conversely, Boina's landscape turned slightly more circular and simpler. The t_med_high was the class most responsible for the roundness, but t_low_med became less complex.

Analysing the standard deviation, the greater homogeneity towards roundness is found in Alvor and the higher heterogeneity appears in Arade. Simpler shapes are found in Boina, whereas in Arade there is a greater difference in terms of class shapes. This is supported by a remarkable gain of edge density (see Table 13) in Arade (56.65) but the standard deviation is the highest among the other landscapes (135.74).

Structure and composition metrics

Connectivity metrics measure the degree of fragmentation and isolation of landscapes. MPI results have revealed that Alvor is the most fragmented landscape, which increased from 1972 to 2010, however, on average it has the lowest standard deviation of mean nearest neighbour, meaning that there has been some compaction process in Alvor´s landscape. On the other hand, Boina is the less fragmented landscape, but with significant changes in MNN from 1972 to 2010. Patches became closer, originating a more compact landscape: tidally restored medium-high saltmarsh contributed significantly to a more compact landscape in Boina (see Appendix C), and natural low saltmarshes became further apart.

Table 14. Standard Deviation of landscape alteration between 1972 and 2010

1972-2010	NumP	ED	Divers	AWMSI	AWMPFD	MPI	MNN	N=1972/2010
Arade	9.20	135.74	18.26	1.07	0.49	429.82	89.33	69/104
Boina	4.57	74.72	7.55	0.93	0.09	209.75	45.39	68/78
Alvor	6.84	64.35	121.93	0.90	0.12	559.45	22.05	67/70

Figure 13 represents the 3D spatial distribution of principal component analysis (PCA) for the years 1972 and 2010. Metrics show great diversity and variability among case studies and years of analysis. Homogeneity is better perceived within the Arade case example, than over the years: the most evident pattern is of the metric mean nearest neighbour, which decreased in terms of importance from1972 to 2010, except in Arade.

Since factors derive from the same set of metrics, this could result in a pair grouping of metrics representative of each group. However, we can see that this pattern is only followed by the metrics of the complexity assemblage: MPFD and PAR are presented within very proximate values in every case study in both years studied. Only in Alvor 2010 these same metrics are positioned with positive values. A key aspect of this analysis is that perimeter area ratio (PAR) is below mean patch fractal dimension (MPFD) in all observations, except in Arade 2010.

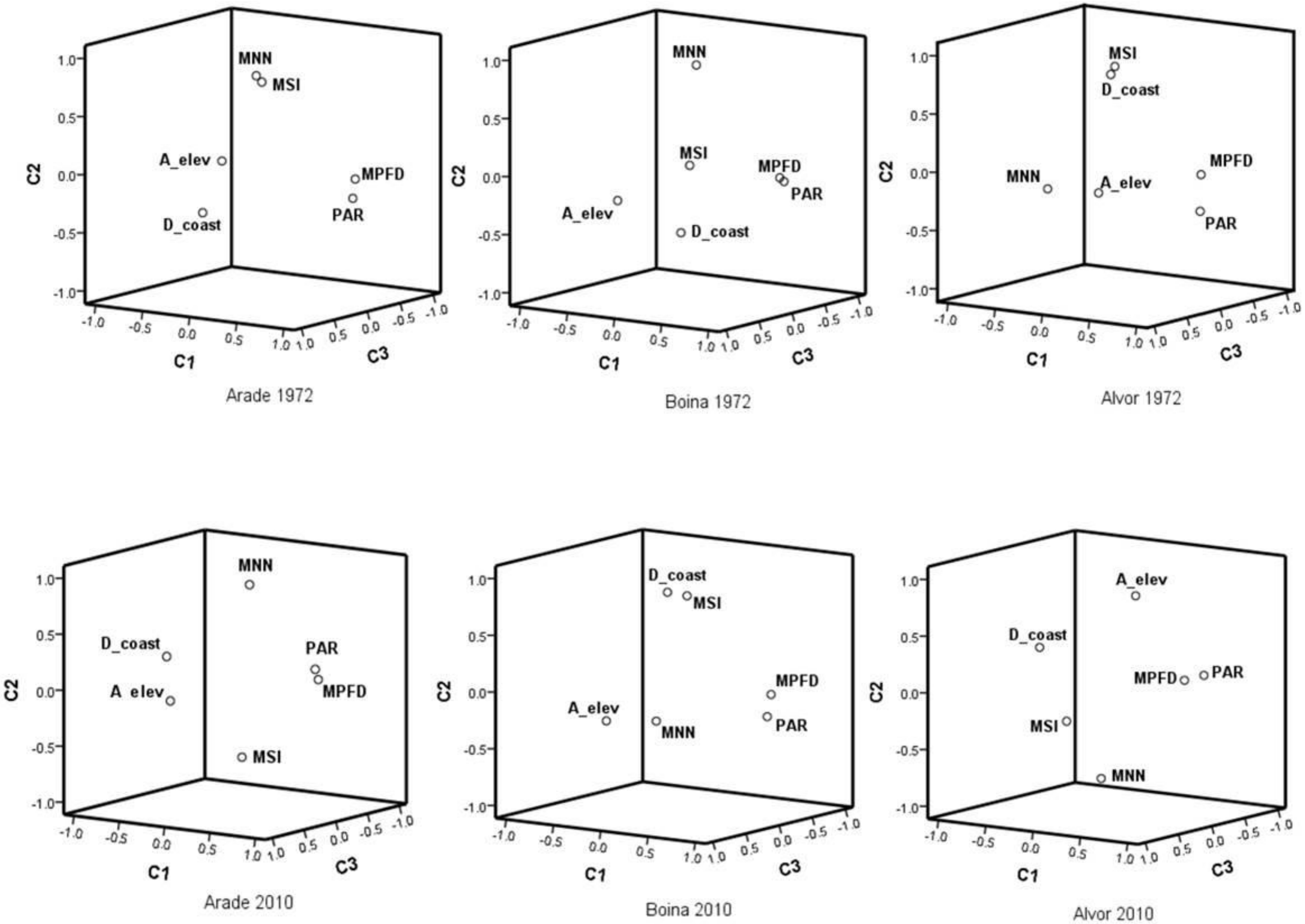


Figure 13. PCA of landscape metrics selected for coastal protection in 1972 (a) and 2010 (b).

Mean shape index has a random behaviour concerning its position towards the other complexity metrics. Great variability in MSI position may suggest its critical contribution for coastal defence, since it acts like a measure of patch and landscape complexity. Both in Alvor 1972 and in Boina 2010, mean shape index appears related with distance to coast (D_coast), but in Arade 1972, there is a strong proximity of MSI and MNN (mean nearest neighbour), while the other cases do not seem to be related to any other metric. Average elevation (A_elev) has a greater explanation capacity in Alvor 2010, than in the other case studies. The metric distance to coast (D_coast) follows that same trend but is more relevant for Boina 2010, where it reaches its maximum capacity of explanation.

In the case of the artificially restored saltmarsh, landscape metrics contributing to explain component 1 are PAR and MPFD (44%) and to explain component 2 are MNN and distance to coast (21%) in 1972. Reporting to 2010, the same metrics were grouped at component 1, but in this case to highlight the negative position of the MSI (shape metrics) and average elevation.

4.5.2 Coastal Defence index

As pointed out in the methods section, the equation used to calculate the coastal defence index functioned as a guide to evaluate disturbances in the provision of ecosystem services, considering each PCA value of each landscape metric within the component and the percentage of variation explained for each component (see Equation).

This was carried out for the three case studies performed at patch level. To increase information on the coastal defence service variability, calculations were made for 1972 and 2010. Therefore, the following results used the standard deviation values to provide a better understanding and comparison, scaled equally in five classes from very low (< -1.5) to very high (1.5-1.7) coastal defence. Data are analysed at patch, class, and landscape levels.

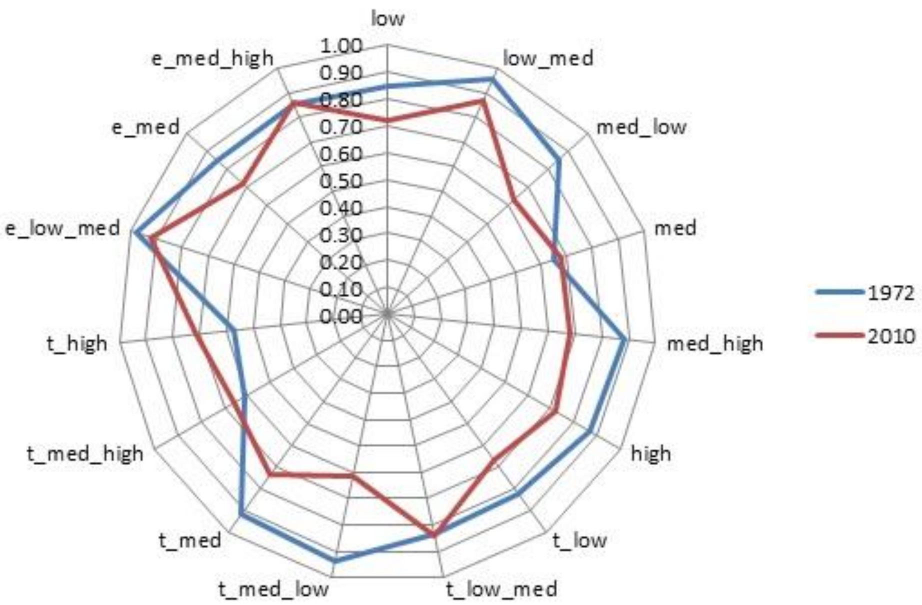


Figure 14. Average Ecosystem Service of Coastal Defence per class (saltmarsh morph-typology) and year.

Legend: (e) enclosed mix marshes; (t) tidally restored saltmarshes; for morphology of saltmarsh patches see table 2; Replicates used in 1972 N=204, and in 2010 N=252.

A comparison of the results reveals an overall decrease of the coastal defence service from 1972 to 2010 in the majority of classes. Figure 14 shows some changes in the classes that tended to offer greater defence – in 2010 higher values for coastal defence are provided by tidally restored and enclosed mix marshes, mainly from medium and high morphologies. Conversely, natural saltmarshes offered significantly higher coastal defence in 1972.

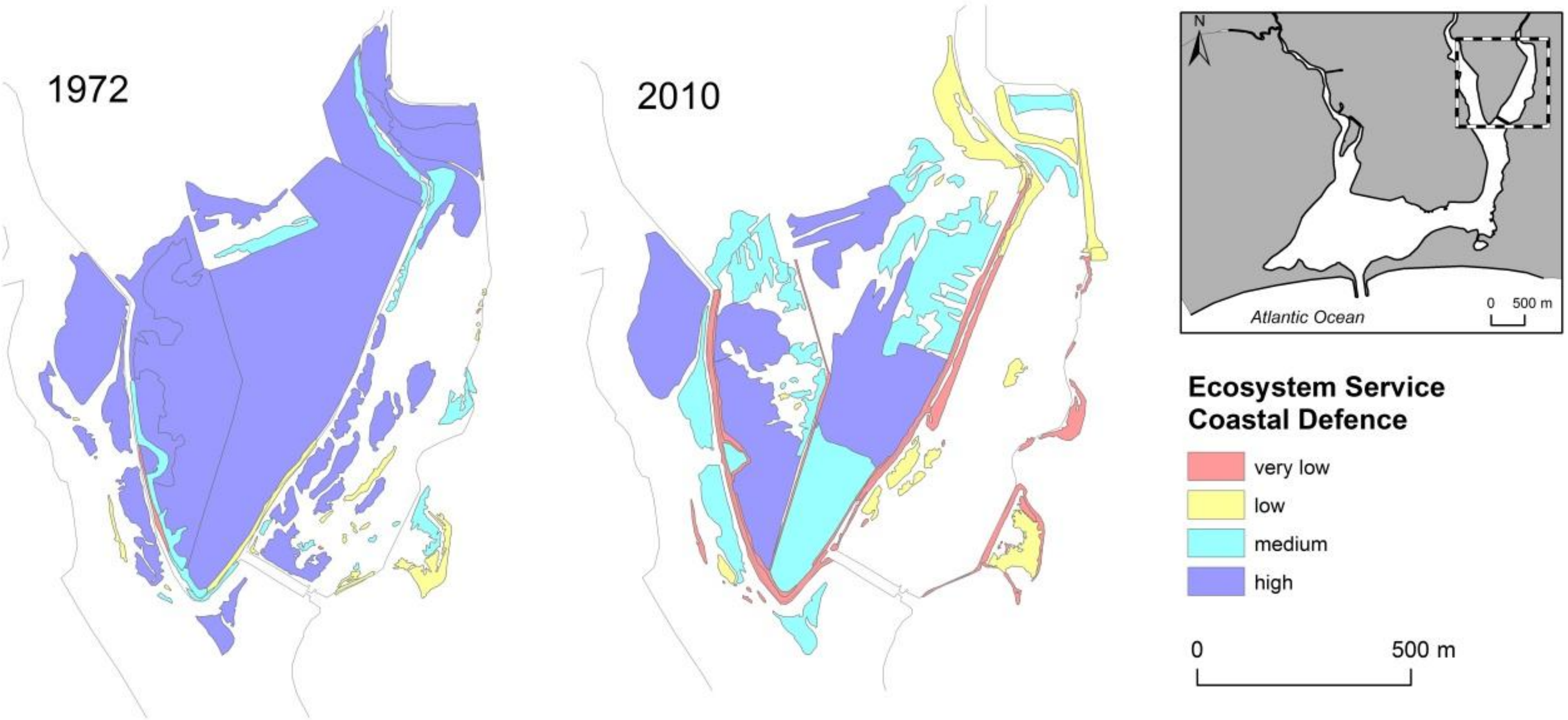
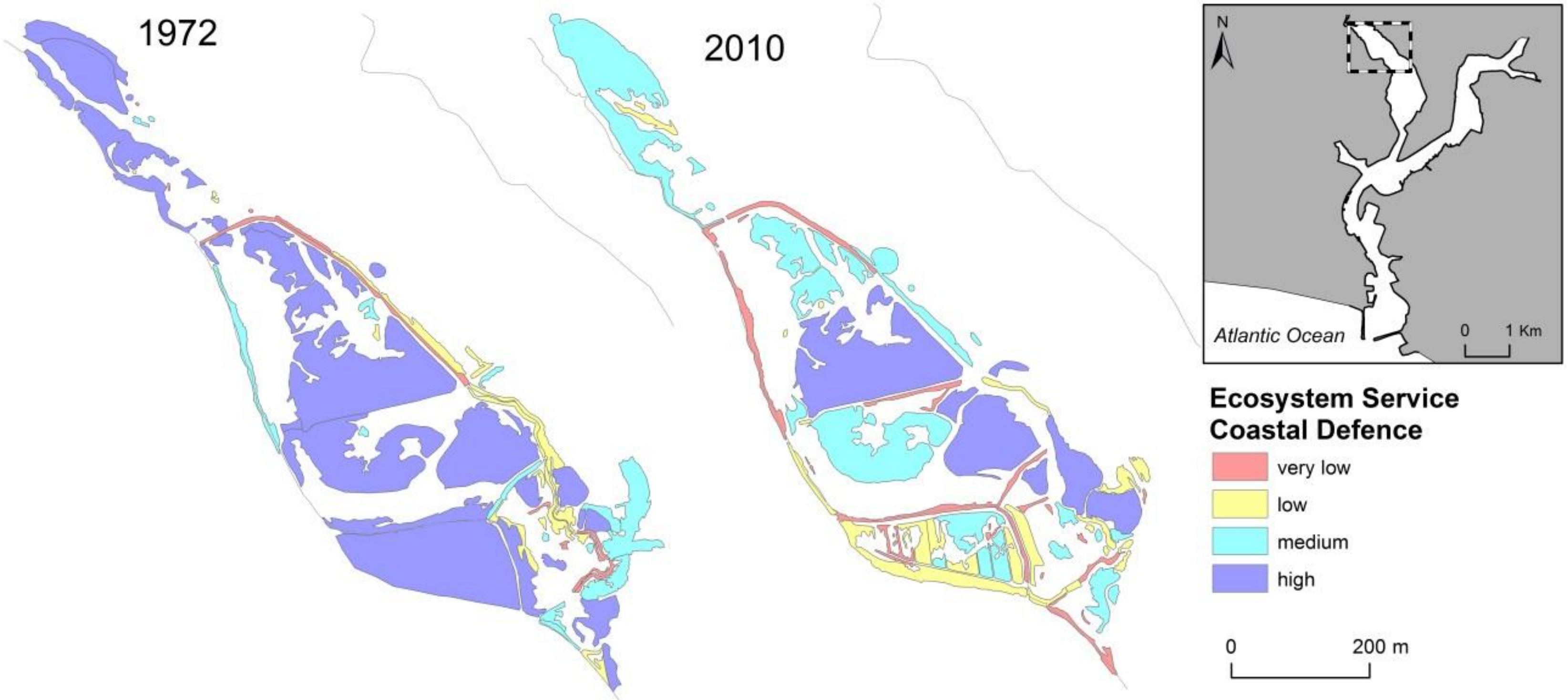

1972
2010
N
Atlantic Ocean
0 500 m
Ecosystem Service
Coastal Defence
very low
low
medium
high
0
500 m

Figure 15.A



1972
2010
N
Atlantic Ocean
0
1 Km
Ecosystem Service
Coastal Defence
very low
low
medium
high
0
200 m

Figure 15.B

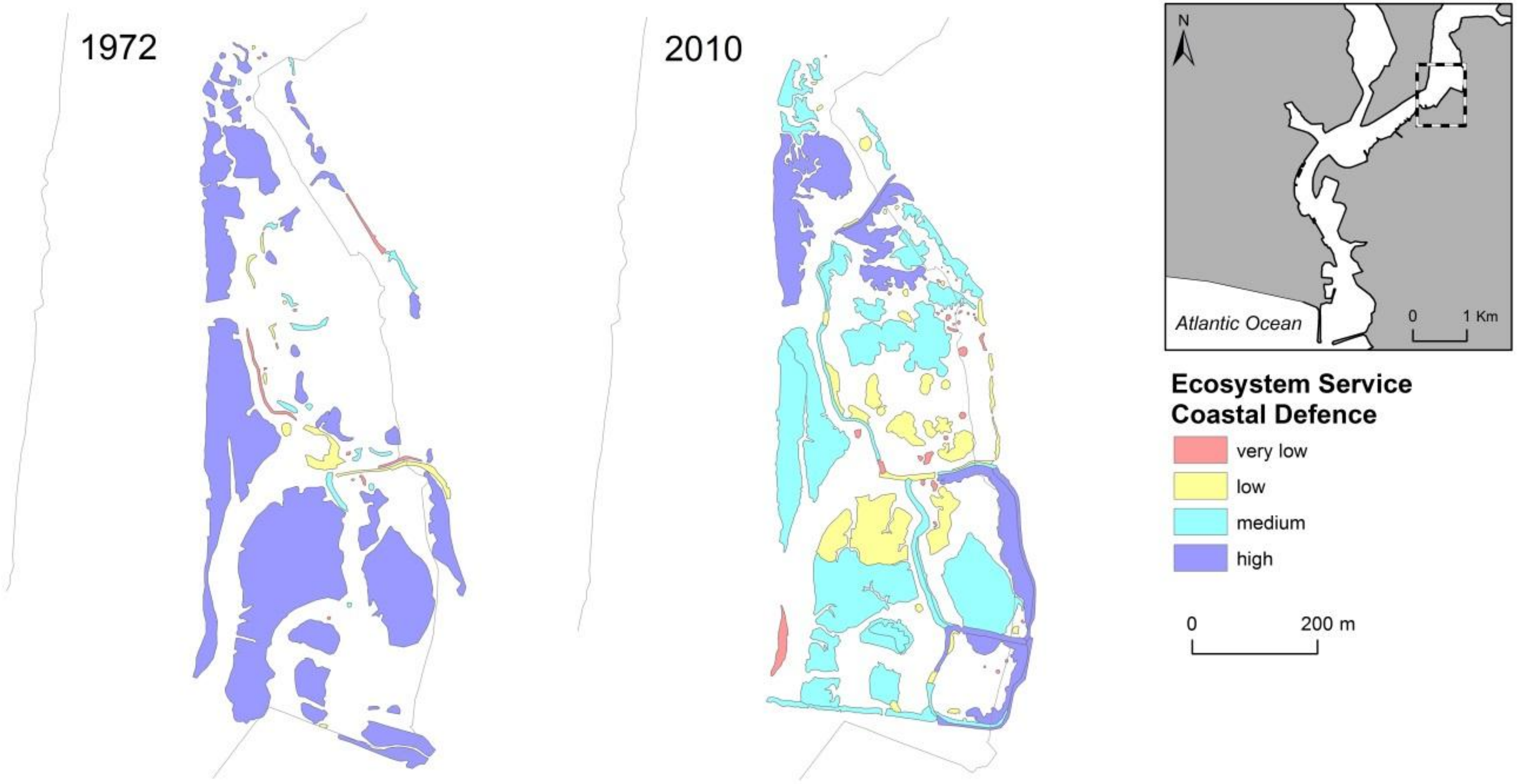


Figure 15.C

Figure 15. Coastal Defence ecosystem service of Alvor (A) Boina (B) and Arade (C), landscapes in 1972 and 2010

This relates to the performance of the coastal defence index – Figure 15 (A to C) shows a comparison of the results for 1972 and 2010. Considering the three saltmarsh ecosystems, there was a significant decrease in the coastal defence capacity from 1972 to 2010. Despite a general increase in the metric perimeter/area ratio, there was a reduction in the number and area of natural saltmarsh patches. Fragmentation might be the cause for decreasing coastal defence provided by natural saltmarshes in 2010, when compared with the same patches in 1972. Additionally, low connectivity and a larger distance between neighbour patches in 2010 led to a fragmented landscape in 2010: smaller, irregular, and less complex shapes have contributed to a decrease of the coastal defence capacity of all these saltmarsh ecosystems.

Considering the artificially restored saltmarsh, there was an increase of coastal defence from 1972 to 2010. Changes in patch number and shape metrics (MSI and MPFD) might have contributed to this pattern change (Figure 16 and Appendix C)

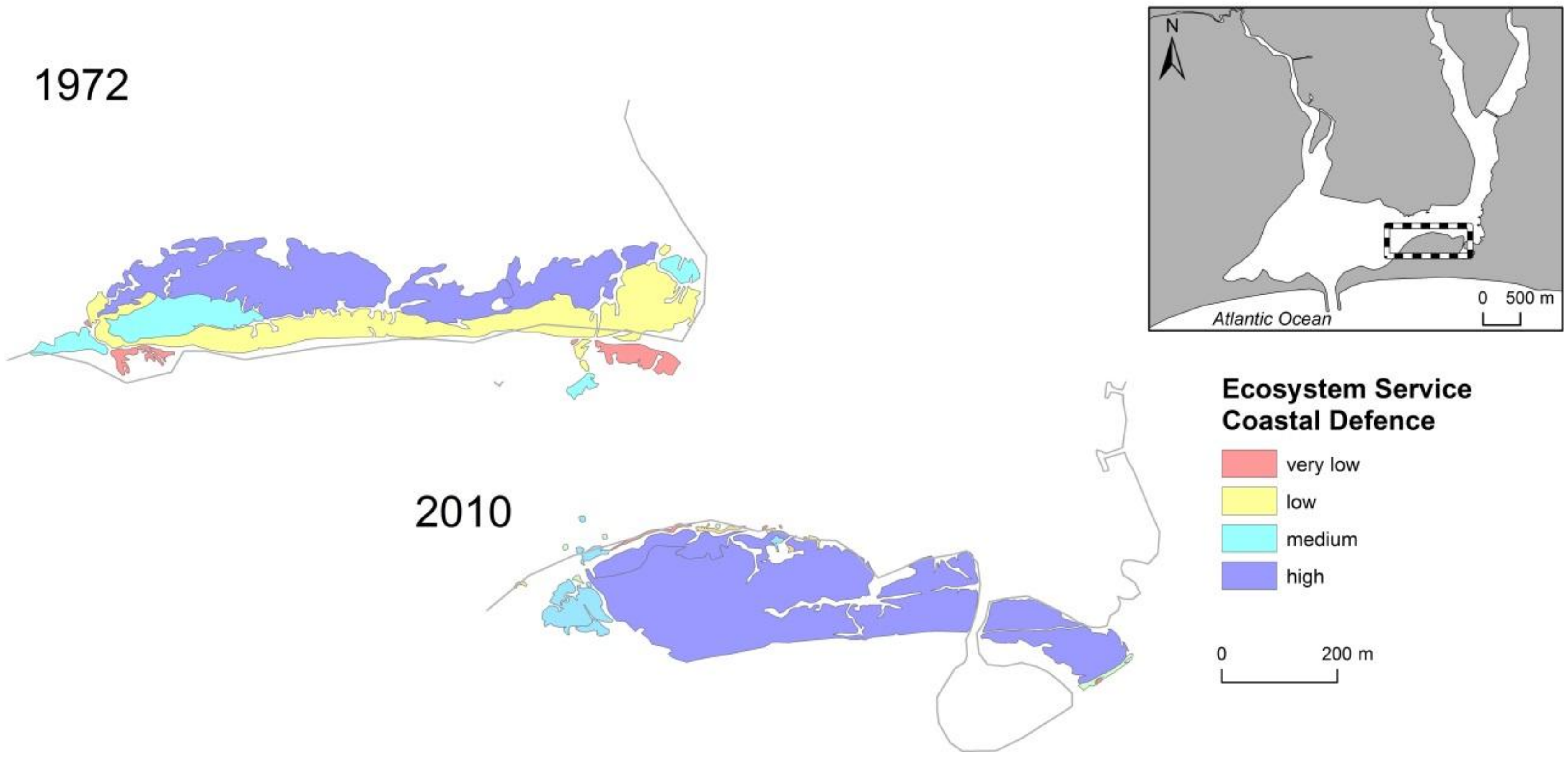


Figure 16. Validation saltmarsh and the coastal defence ecosystem service in 1972 and 2010

4.6 Discussion

Coastal protection is a critical issue on the European environmental agenda (EASAC, 2009), and also a priority investment in many countries - Jones et al., 2011 estimate that in the UK, approximately 3.5% of the national income comes from the economic value of ES. According to IPCC (2014) last findings on sea level rise (0.24 m of mean sea level rise until mid-21st century) and forecasts of ecosystems affected and human lives endangered (Esteves 2014). Decision makers and government agencies must be aware of the capacity of the ecosystems to deliver their services properly. Therefore, public investment is allocated to protect and restore ecosystems, namely estuarine saltmarshes that face serious threats to their environmental integrity (Barbier et al. 2011). In order to address this need and since it is unfeasible to attribute an economic valuation to the ecosystem service (in these case examples), the coastal defence index (ES_CoastDef) was calculated using landscape metrics, which can be acquired worldwide and applied to other ecosystems. This is the major advantage of this index, although it competes with other robust models that combine analysis on ecosystem services assessment as well (i.e. InVEST). Coastal defence/protection models developed with the InVEST interface are very data demanding, since they are based on predictions for sea level rise effects of ecosystems, namely wetlands, saltmarshes and their consequences for human populations and impacts on economic activities. Calculations were tested using InVEST, but the unavailability of suitable data made this impossible to accomplish; the scarcity of data and the unevenness of the scale of analysis have affected the results.

Table 15 sums up the data required for the coastal protection model used by InVEST and the crossing aspects of the ES_CoastDef model presented in this paper.

Table 15. Crossing aspects of data used by InVEST and ES_CoastDef index

	InVEST	**ES_CoastDef**	**ES_CoastDef** *advantages*	**ES_CoastDef** *disadvantages*
Hydrodynamic model	✓			no data on sea level, hydrodynamics
Land use / cover	✓	✓	High detail (patch level)	
Elevation and soil characteristics	✓	elevation		soil characteristics give absorption capacity
Saltmarsh shape	✓	✓	AWMFD; NumbP; Paratio; diversity metrics	
Distance to coast	✓	✓	MNN; MPI	
Plant density	✓	vegetated saltmarsh	marsh morphology: low, medium, high	
Transects along the coast	✓		saltmarsh typologies	
Blue carbon	✓			no data on carbon sequestration or burial

Source: based on Guannel at al., 2014; Natural Capital Project, 2015.

By adding some detail at patch level, the ES_CoastDef index tries to supress some disadvantages when compared with the InVEST model, whose data are more precise at the landscape level and accurate regarding the data from the sea (i.e. hydrodynamic model, storm water, sea level rise, water level). These data can be useful to provide information on the damages to human communities and to manage the reduction of the impact on a 'what if scenario'. Nevertheless, ES_CoastDef deals with more local information, obtained mainly from landscape metrics, which informs the planning system on saltmarsh morphology evolution and response capacity regarding land use/cover changes.

Considering now the experimental evidence, the number of patches increased considerably in Arade, due to the growing tidally restored saltmarshes (low and med_low), which improved the defence capacity within this typology (see Figures 2 and 3). Nevertheless, natural saltmarshes were more fragmented and isolated in 2010 than in 1972 because of the construction of dykes in the southern part of the Arade landscape. Fragmentation occurred due to a greater relevance of distance to coast in 2010, multiplying the number of patches and consequently the number of centroids that resulted in a shortening of the distance to coast metric. Moreover, in the northern part of Arade, natural saltmarshes increased complexity (highly convoluted edges) and connectivity, therefore starting to represent a very high coastal defence in 2010. In the case of Boina, patches were more compact and simpler in shape in 1972 than in 2010. Tidally restored saltmarshes located in southwest Boina are an example of coastal defence reduction associated with an increase in fragmentation. On the other hand, natural saltmarshes in the east improved their ecosystem services considerably, going from low/medium to mostly high and very high coastal defence indexes. The central patches of tidally restored saltmarshes registered few changes from 1972 to 2010, as opposed to the northern natural saltmarshes, where it is possible to find a reduction of area occupied by those natural saltmarshes, leading to a significant decrease of the coastal defence capacity. In the case of Alvor, enclosed mix marshes recorded the highest values of defence. Compactness and connectivity in western natural saltmarshes offered in 1972 a greater coastal defence. On the contrary, eastern natural saltmarshes were classified as tidally restored in 2010 (due to the construction of a dyke). Patches have become smaller and less in number, offering a low and very low coastal defence.

Changes in landscape metrics performance from 1972 to 2010 are of two causes: (i) a decrease in the coastal defence service from 1972 to 2010 and (ii) some changes in the classes which tended to offer greater defence. The decrease in the coastal defence service was due to a greater fragmentation of the landscape, an increment of the overall number of patches, a higher mean proximity index, and additionally a general decrease of mean patch size in all the example cases. These are representative of a more isolated and fragmented landscape, resulting in a decay of the coastal defence capacity due to loss of connectivity. As explained by Leitão et al., (2006) and Doody (2008), the interruption of connectivity may lead to a lower probability of delivering ecosystem services. Through a considerable increase in the mean patch fractal dimension metric from 1972 to 2010, the analysed landscapes became more complex and with a larger amount of edges (total edges also increased significantly). More complex and convoluted landscapes are usually guided by changes in mean shape indexes, which had also occurred from

1972 to 2010 (McGarigal & Marks 1995). Changes in the typology and morphology of the saltmarshes that provided worse coastal defences in 2010 are related to an increase of fragmentation among natural saltmarshes: despite their increase in number, patches of natural saltmarsh (mainly of low morphology) became smaller (reduction of mean patch size) and edgy, contributing decisively to a higher isolation and therefore a connectivity weakening. Facing lower coastal defence in seafront tidally restored and enclosed saltmarshes gained emphasis in 2010, despite the overall decrease of coastal defence. Standard deviation values show an increase in landscape consistency in 2010, indicating homogeneity in providing coastal defence but not a better performance.

Following PCA results, components contributing decisively to the construction of the coastal defence index are divided into three major groups (see Figure 2): the first two are related to landscape metrics and the third is associated with local specificities.

a) Complexity (component 1): mean patch fractal dimension gathers from 45 to 49% of the explanation capacity. This metric is prevalent to determine the coastal defence index because it informs about shape and its complexity, providing insights into the edginess and roughness of a landscape. More complex patches provide greater coastal protection because elongated shapes are more capable of offering protection from coastal flooding (Möller 2006; Van Loon-Steensma&Vellinga, 2013).

b) Connectivity (component 2): mean nearest neighbour and distance to coast explains circa 30% of the ES_CoastalDef index. Therefore connectivity metrics play a crucial role on building the coastal defence index because they inform simultaneously about structure and function (Leitão et al. 2006). Fragmentation affects the exchanges of flows and the effective landscape connectedness, compromising the capacity of coastal defence and even the long-term survival of saltmarshes. Isolated patches do not offer the barrier effect expected of healthy and connected saltmarshes (Möller et al. 2001; Costanza et al. 2008; Feagin et al. 2009; Teixeira et al. 2014), consequently, better performance of connectivity metrics indicate higher delivery of coastal defence services.

c) Regional diversity (component 3): M.A. (2005) and de Groot et al. (2010) point localized factors as drivers for short-term change in ecosystem services provisioning. These can be demographic shifting, legal and institutional frameworks, local policies, economic or cultural losses. In the case example presented, component 3 is dominated by the average elevation metric (explanation 22% to 26%). This might be related to the use of the same average elevation for both years of analysis; nevertheless, shape and connectivity are represented as the third component in the case of Alvor. The emergence of enclosed mix marshes in Alvor 2010 resulting from the abandonment of formerly reclaimed agricultural uses indicates that land use changes can be considered localized factors, stressing the construction of the coastal defence index.

Regarding the tests on coastal defence developed for the artificially restored saltmarsh, the same metrics within the components (1 and 2) provide the context for the construction of the index, although local factors seem to be a lateral aspect in this particular case, probably due to unnatural factors. Tendency of protection changed in the artificially restored saltmarsh, since it appears to offer higher defence than the others in 2010, but also faced a drastic reduction in terms of area and number of patches, which consequently affected other landscape parameters. The compactness and roundness of 2010 patches improve connectivity (Leitão et al., 2006) and contrast with elongated fringing shapes of 1972´s saltmarshes. Additionally, elevation in pre-intervention was significantly lower than the records from 2010 (average elevation 5 m), especially for low and medium marsh terraces, ranging from 7.16-10.75 m. This might support the evidence of different positions of A_elev in the PCA analysis for 1972 and 2010: conditions of higher elevation in the saltmarsh surface enable coastal protection (Loon-Steensma & Vellinga, 2013). Nevertheless, this higher coastal protection was manipulated by cutting off 2.1 ha of saltmarsh for touristic development and artificial sediment filling, in which the restored saltmarsh surface was elevated. Formerly reclaimed land, usually with agricultural uses is a target for managed realignment processes in the UK, accomplishing good results in coastal protection due to higher togography (Luisetti et al., 2011; Esteves, 2013).

This study opposes naturally recovering marshes and an example of artificially restored, demonstrating that fragmentation is threatening coastal defence. The development of saltmarsh patches within formerly reclaimed areas can provide an increase of coastal defence in situations of reducing and/or isolating natural saltmarsh areas. Luisetti et al. (2011), Loon-Steensma and Vellinga (2013) and Esteves (2014) indicate that it is possible to reduce the costs of construction and maintenance of artificial embankments when there is a stable and natural foreshore (saltmarshes).

Discussions around methodologies to evaluate or measure ecosystem services chiefly deal with the complexity underlying the composition of an index, capable of assembling the vastness of other small indicators that compose an ES. The effectiveness of the classification system proposed by M.A. (2005) was challenged by Wallace (2007), who provided a distinction between processes and services in his classification framework, aiming to achieve a balance between human values, services and assets. The findings of Van der Biest et al. (2015) point to an impoverishment of results when we simplify the evaluation of ecosystem services based on land use indicators. The authors argue that a refined land use classification grid brings few improvements to ES evaluation. Spencer & Harvey (2012) uphold holistic approaches when evaluating ES, especially after disturbance or managed realignment strategies. This study advances in delivering an alternative tool to assess coastal defence service, provided by saltmarshes, restored whether with or without artificial intervention.

4.7 Conclusion

In this context, the ES_CoastDef index allowed to obtain precise data on saltmarsh patches (shape, fragmentation, isolation, neighbourhood, perimeter and area, and elevation), which were unknown until this moment but still critical to the planning process. Knowing the state of the saltmarsh ecosystems (diagnosis-based knowledge), it is possible to evaluate conducts, design strategies for conservation and consider ecological restoration. Mainly in situations where protecting structures (although not for this purpose) are going through a degradation process and there are signs of different ways of ecosystem rebooting, counting on metrics to diagnose and calculate ecosystem services – such as the coastal defence index – is fundamental to advance in the managing field. Although the number of patches has grown in some cases, location within the landscape, complexity and shape gives them a lower ES_CoastalDef index in 2010. Higher values for coastal defence are provided by tidally restored and enclosed mix marshes, mainly from medium and high morphologies in 2010, as opposed to natural saltmarshes that offered significantly higher coastal defence in 1972. The analysis of an artificially restored marsh provided good results in testing the coastal defence index, suggesting similar landscape metrics to build the index. Additionally, results demonstrated that higher elevation conquered artificially played a fundamental role in increasing the coastal defence service. Thus, with this index it is possible to outline target coastal defence parameters that the recovered saltmarsh should achieve in order to maintain or enhance the index performance.

Therefore, landscape ecology metrics to study formerly reclaimed saltmarshes as a way to measure and evaluate ecosystem services proved to be valuable and of broad application. The ES_CoastalDef index can be replicable to other saltmarshes, regardless of the biogeographic region, because data needed and methods applied to achieve the coastal defence index are of international availability, provided geographic information on saltmarshes and landscape metrics are available.

Chapter V – Vegetation diversity and trajectories in realigned and naturally recovering saltmarshes

Submitted paper: Almeida, D; Neto, C; Costa, JC 2016. Vegetation diversity and trajectories in realigned and naturally recovered saltmarshes, to *Journal of Nature Conservation,* April 2016.

5.1 Abstract

Managed realignment has been guiding the last decades of active recovery in saltmarsh ecosystems, although the development and composition of vegetation has been questioned. Saltmarsh vegetation that derives from passive recovery has higher similarities in floristic composition than that derived from active recovery. We analyze and compare the results of vegetation recovery and the flora development fostered by active interventions and passive recovery cases. Methods include a reference dataset of vegetation developed in realigned Atlantic saltmarshes and floristic surveys conducted on Mediterranean saltmarshes. We have considered natural, tidally restored, and enclosed mix marshes, which were statistically analyzed by applying multivariate analysis. An unweighted similarity index was performed to compare floristic diversity. Vegetation present in test saltmarshes is dominated by exclusive species; ruderals represent a third of the total species composition. Higher floristic diversity is found within enclosed mix marshes, while natural saltmarsh characteristic species tend to be of broader onset. Floristic composition of passive recovery saltmarshes is within a local species pool, supported by a higher similarity index found in Mediterranean saltmarshes. Findings support passive recovery options, reducing major changes in the local species pool and assuring floristic similarities with natural saltmarshes.

Key-words: active recovery, passive recovery, floristic composition, vegetation similarity, species pool, managed realignment

5.2 Introduction

Considering the theories on ecosystem behavior, equilibrium, chaos, and resilience, van de Koppel et al. (2005) studied emergence as a key property of complex systems, demonstrating that self-organization improves the functioning of saltmarsh ecosystems on short time scales (30 years), rather than on larger time scales (60 years or more); the disturbance dynamics starts to dominate the system, therefore self-organization can lead to an avalanche-like dynamics. This may imply that the future saltmarsh plant succession is compromised due to the inability of saltmarsh spatial organization (van de Koppel et al., 2005). Also, Doody (2008) emphasizes that the equilibrium in saltmarsh ecosystems results in a combined state of erosion and accretion, which are not mutually exclusive. Van der Wal et al. (2008) demonstrate the fundamental role played by local feedback mechanisms, which is more important than the whole saltmarsh cycle or spatial pattern. Self-organization regulates saltmarsh propagation and erosion, being responsible for in-system dynamics and changes (Duarte et al., 2011). In saltmarsh cycles, which can take from years to decades, ecosystem services' gains and losses are a part of the ecosystem's balance (Cortina et al., 2006).

Habitat restoration and nature conservation are within the European policy (EU, 2004; Liquete et al., 2013), and accommodate incentives for wetland creation, restoration and management (Crépin, 2005; Kotowski et al., 2013; Hedberg et al., 2014). The interest on saltmarsh recovery and habitat restoration started to be associated with rapid changes in erosion and accretion rates endangering the human communities that live in the borderline saltmarsh (Boorman et al., 2002; Doody, 2013). Low-lying areas left by successive embankments and land reclamation over centuries fail in the absorption of wave energy and in the capacity to accommodate flooding. These aspects started to be valued in a context of climate change, particularly rising sea levels, and these concerns, in the first phase, have led to the construction of breakwater structures (Elliot et al., 2007; Doody, 2013). Nevertheless, these hard measures started to degrade and sustainability issues arose whenever breakwaters and groins needed repairing or reconstruction (Esteves, 2014).

We aim to explore the vegetation composition of stabilized saltmarshes (henceforth 'reference saltmarshes') and compare them with the vegetation development of nearby restored areas in international examples of managed realignment (hereinafter referred to as MR) sites (active recovery). These will provide the reference to assess the progress of vegetation development from the passive recovery observed in Western Algarve saltmarshes (Portugal) when compared with natural saltmarshes. This comparative approach can guide future interventions in saltmarshes elsewhere because it establishes recovery trajectories based on the similarity of vegetation cover and degree of presence (species-by-species). The novelty of this work comes from the fact that Portuguese saltmarshes are in the Mediterranean biogeographic region but facing the Atlantic Ocean, which makes them very different as far as floristic composition is concerned when compared with Atlantic saltmarshes. Therefore, this research is able to provide valuable comparison of vegetation succession both in Atlantic and Mediterranean (Costa et al., 2009a) saltmarshes in different contexts of recovery (active or passive), trying to provide further insights into unmanaged saltmarsh development.

5.3 Methodology

The methodology used in this paper relies on the literature review on coastal ecosystem recovery, bringing to light the scattered concepts on intervention options. We analyze and compare the results of vegetation recovery and the path of flora developments (frequency, cover) derived from active interventions (international examples of MR) and passive recovery cases (Western Algarve´s saltmarshes).

Vegetation data have two origins: for international examples on saltmarsh active recovery, we used floristic surveys found on the literature, which studied, analyzed and compared saltmarsh vegetation in a pre-intervention state, and accompanied the vegetation evolution in managed realignment or restoration sites (Table 15).

Test saltmarsh data are derived from the work of Almeida et al. (2014), who identified two saltmarsh typologies resulting from: a) the end of saltmarsh reclamation for agriculture; b) dike breaching; c) the abandonment of other protecting structures and salt pans. Saltmarsh typologies emerging from these processes are tidally restored saltmarshes (hereinafter referred to as TRS) (Fig. 17), which correspond to the former diked land, where the rupture of protecting dikes allowed tidal reactivation; and enclosed mix marsh (hereinafter referred to as EMM) (Fig. 18) occurring inlands, in which agriculture has been truly practiced and therefore there were remnants of saltmarsh plant communities. The geographical location of these saltmarshes is Portugal – the Algarve region (Mediterranean biogeographic region); N 37º07.097´ W008º30.189´ (Arade estuary) and N 37º07.991´ W008º37.440´ (Alvor estuary).

Figure 17. Vegetation structure of a tidally restored saltmarsh in the Arade estuary.

Figure 18. Mediterranean salt steppes of *Limonietallia*, habitat reference 1510 (enclosed mix marshes in the Alvor estuary).

5.3.1 Reference data set

Vegetation of managed realignment and restoration projects was compared with reference marshes (Atlantic saltmarshes) found in the literature, and synthetized in Table 16. Floristic surveys applied the Braun-Blanquet (1979) scores method.

Table 16. Correspondence between reference and test saltmarshes, goal, methods, and data involved.

Goals/methods	*Data*	*Correspondence with test saltmarshes*	*Author / site*
Aims: species pool, site suitability, and regional/local species availability **Methods**: species monitoring; surface elevation; saturation index; stepwise discriminant analysis	70 sites	Tidally restored saltmarsh (TRS) (83 *relevés)*	Wolters et al. 2005; 2008 Western Europe MR saltmarshes
Aims: investigating the composition of communities and their change over time on the MR site; **Methods**: Bray–Curtis similarity index; K-means cluster analysis	145 *relevés*	Enclosed mix marshes (EMM) (49 *relevés)*	Mossman et al. 2012b Brancaster, North Norfolk – U.K.
Aims: vegetation trajectories of barrier island restored saltmarshes (<u>behind low dams</u>) **Methods:** Species-by-species analysis; ordination by using a correspondence analyses (CA)	170 *relevés*	Atlantic vs. Mediterranean saltmarshes (368 *relevés*)	Van Loon-Steensman et al. 2015 DutchWadden Sea – The Netherlands

5.3.2 Vegetation surveys

Floristic surveys were conducted in Southwest Algarve (Fig. 5, Chapter I) during 2 entire years (2012 to 2014) in different saltmarsh environments and periods to include floristic seasonality. Preparation for the floristic surveys was carried out according to the Zurich-Montpelier methodology in order to address floristic differences between marshes. The floristic surveys were conducted following the abundance/dominance scores method (Braun-Blanquet, 1979) by using sampling quadrats of 2 m^2 along selected transects. The floristic surveys were conducted in randomly selected saltmarsh areas: 167 in Alvor saltmarshes, and 201 in the Arade River, totaling 368 surveys. We found 60 different *taxa* (Costa et al., 2012).

The degree of presence (Braun-Blanquet, 1979) was calculated to evaluate the differences in species richness between saltmarsh areas. The presence is estimated in percentages of a species and classified according to a chosen scale into a set of 'classes of presence'. The classes were defined according to Costa et al., 2009: r (< 6%); + (6-10%); I (11-20%); II (21-40%); III (41-60%); IV (61-80%); V (> 81%). Braun-Blanquet´s 'cover-abundance' scale was used in the floristic surveys, but for the statistical analysis it was transformed using the average coverage of Leps and Smilauer (2003) and Portela-Pereira (2013): (5) 87.5%; (4) 62.5%; (3) 37.5%; (2) 15%; (1) 3%; (+; r) 0.5%. Nomenclature and ecological classification (definitions of exclusive and preferential species) follows the main flora works by Castroviejo (1986-2012), Franco (1971, 1984), Franco & Rocha Afonso (1994, 1998, 2003), and Rivas-Martínez (2005).

5.3.3 Data analysis

A hierarchical classification, description, and floristic interpretation of test saltmarshes was made using SYN-TAX 2000 (Podani, 2001). The vegetation data were hierarchically classified (UPGMA) using the Bray-Curtis coefficient, which results in a dissimilarity clustering of vegetation. This was performed separately for each geographical area (Alvor and Arade) and saltmarsh typology (natural, tidally restored saltmarshes, and enclosed mix marshes) of the test inventories. Additionally, frequency of occurrence, average coverage, and species relevance were calculated. Moreover, a principal component analysis (PCA) was performed on test saltmarshes, in which ordination was defined by the species cover-abundances. Here we used CANOCO software and selected the inter-species correlation option. Further on, we applied a multivariate analysis to arrange floristic surveys along the axes, based on their floristic composition (Capelo, 2003), this allowed us to test the importance of the local species pool.

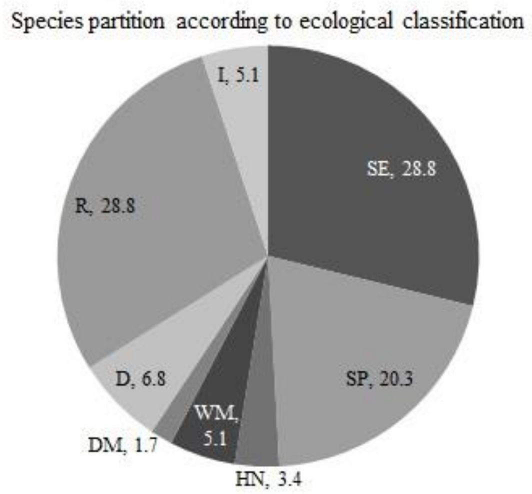
We employed a species-by-species analysis (van Loon- Steensma et al., 2015) based on frequencies in both the test and the reference data sets. This method provided detailed insights into the characteristic species and those transitioning from pre-state to managed realignment saltmarshes. The unavailability of reference floristic surveys made it impossible to develop a more robust statistical analysis.

For a comparison between and within groups (see Table 17), we applied the unweighted similarity index developed by Byers & Chmura (2007): Si=(2A/[2A+B+C]) * 100%, where A equals the number of species the two saltmarshes have in common; B stands for the species which are exclusive to saltmarsh 1; and C represents the species exclusive to saltmarsh 2 - the closer to 100%, the highest the index of similarity.

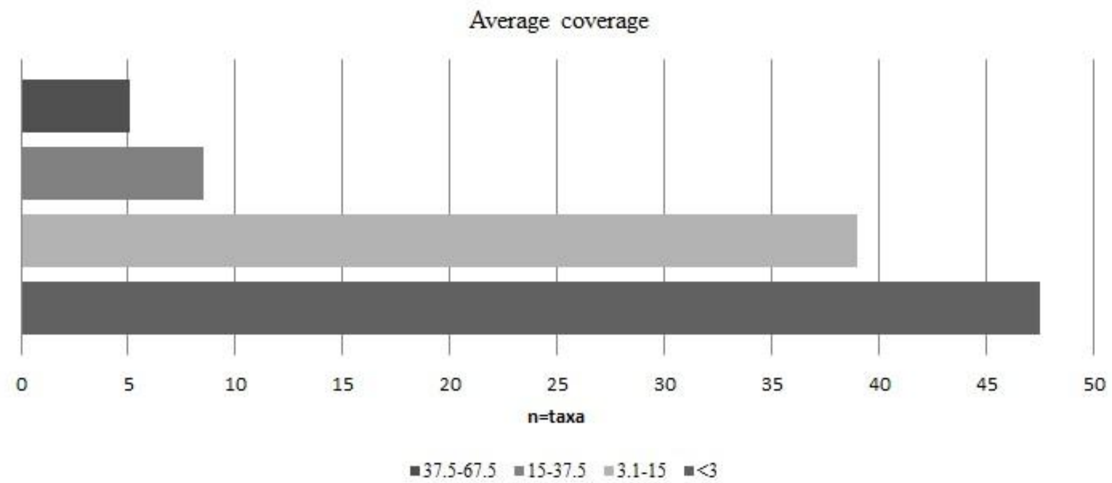
5.4 Results

5.4.1 Vegetation composition of Mediterranean saltmarshes

The vegetation present in test saltmarshes is dominated by exclusive (29%) and preferential (20%) species (Fig. 19a and 19b). Ruderal species represent almost a third of the total species composition (29%) and species preferring dune habitats represent 7%. Invasive species represent 5% of the total *taxa* and the remaining 10% is distributed among fresh water species and halo-nitrophilous.



a)



b)

Figure 19. (a) Species partition according to ecological classification [**SE** saltmarsh exclusive; **SP** saltmarsh preferential; **R** ruderal; **D** dunes; **HN** halo-nitrophilous; **I** invasive; **P** parasite; **WM** wet meadows; **DM** dry meadows] and (b) Average coverage according to scales (%)

Average cover (Leps & Smilauer 2003) according to scale grouping revealed the majority of species (47.5%) cover on average less than 3% of the surveyed area. There is no evidence of average coverage equal or higher than 87.5%. To better understand the distribution of the characteristic species, Figure 20 represents the species coverage of the 15 to 67.5% scales. *Halimione portulacoides* (=*Atriplex portulacoides*) is among the most common species, the average coverage is 62%, followed by the *Sarcornornia* species, which together form the most regular associations in lower/medium saltmarshes in Portugal (Costa et al. 2012).

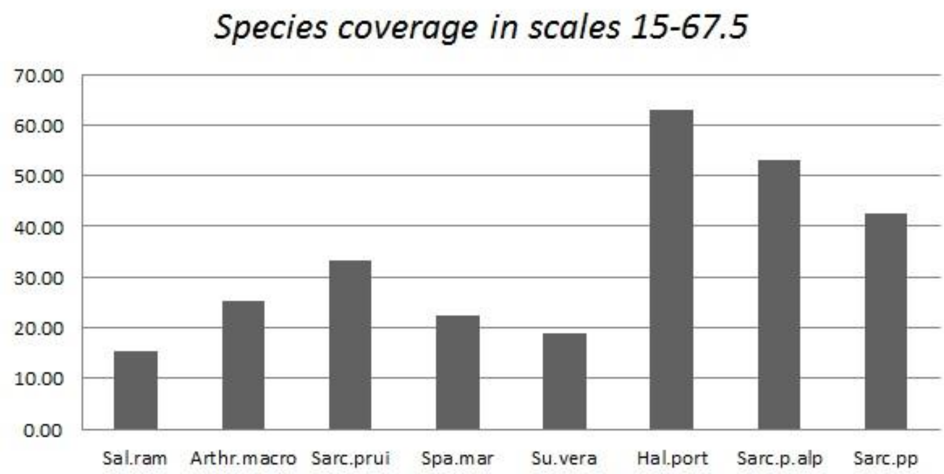


Figure 20. Species covering scales 15% to 67.5%: *Salicornia ramosissima* (Sal.ram); *Arthrocnemum macrostachyum* (Arthr.macro); *Sarcocornia pruinosa* (Sarc.prui); *Suaeda vera* (Su.vera); *Halimione portulacoides* (Hal.port); *Sarcocornia perennis* ssp.*alpini* (Sarc.p.alp); *Sarcocornia perennis* ssp. *perennis* (Sarc.pp)

Natural saltmarshes: in both Alvor and Arade, characteristic species of saltmarshes appear in well-defined groups, representing a clear hierarchical partition in ecological terms. In the Alvor saltmarsh, exclusive and preferential species are grouped separately but sequentially; followed by the invasive *Oxalis pes-carpae* (linked with ruderal *Calendula avensis*) and *Carpobrotus edulis*; *Cotula coronopifolia* linked with *Hypochaeris radicata* (wet meadow) and *Polypogon maritimus* (saltmarsh preferential). Most of the ruderal species are within the same large group (i.e. *Mellilotus segetalis*, *Scorpius vermiculata*, *Emex spinosa* or *Medicago polymorpha*). The last group is composed of a miscellaneous of dune species: halo nitrophilous (*Frankenia laevis*), dry meadows (*Asparagus albus*), and fresh water (*Bolboschoenus maritimus* var. *maritimus*).

In Arade, two groups stand out from the UPGMA analysis: i) saltmarsh exclusive and preferential species; ii) composed of ruderal, invasive, and wet meadows species. Within the first group, *Limonium* and *Artemisia* genus come together and are followed by saltmarsh exclusive species. Characteristic species of high marsh stand out from the other exclusive or preferential species. Low marsh *Spartina maritima* and *Salicornia ramosissima* mark the close-up of the first group, which includes the invasive species *C. edulis*. The second group encloses a set of saltmarsh preferential species, mostly ruderal and other invasive species.

Tidally restored saltmarshes**:** in contrast with natural saltmarshes, there is a less evidence of species partition and organization. In the case of Alvor, ruderal species (i.e.

Anagallis arvensis, M. polymorpha, Spergularia bocconei) appear associated sequentially with high marsh and fresh water species. At the tail of the group, we find the exclusive species. Arade´s UPGMA draws similarities with natural saltmarshes, as characteristic species of high, medium and low marshes are grouped sequentially; followed by a large grouping of ruderal and saltmarsh preferential species. The localisms *Limonium algarvense* and *Limonium lanceolatum* are grouped together with the invasive *O. pes-carpae*, *S. maritima*, *Juncus acutus*, and *Puccinellia maritima* are linked directly at the top.

Enclosed mix marshes: in Alvor's enclosed mix marshes, there are three large groups: the first group mixes the main low and medium saltmarsh characteristic species (i.e. *Arthrocnemum macrostachyum*, *H. portulacoides*, *S. ramosissima*, *Sarcocornia. perennis* spp *alpini* and *Sarcocornia pruinosa*); the second group is composed of fresh water and ruderal species, as well as the invasive *C. coronopifolia*; the last group aggregates few ruderal and the saltmarsh preferential (i.e. *Salsola vermiculata, Suaeda albescens* or *Sonchus maritimus*). Considering Arade´s EMM, the sampled area reveals a similar composition to natural saltmarshes. High and low saltmarsh characteristic species are very well defined and grouped separately, corresponding to the first and third groups, respectively. The middle group is composed of a mix of preferential species (*Elytrigia juncea*, *Elytrigia elongata*), with ruderal species (we would like to highlight the absence of invasive species).

The multivariate analysis allowed us to understand the species distribution according to ecological classification and saltmarsh typology. Axes vary between -1 and 1. Figure 21 shows different typologies share exclusive and preferential species. Nevertheless, saltmarsh exclusive species (SE) are represented in the outer quadrants, revealing a narrow distribution, particularly targeting natural and tidally restored saltmarshes.

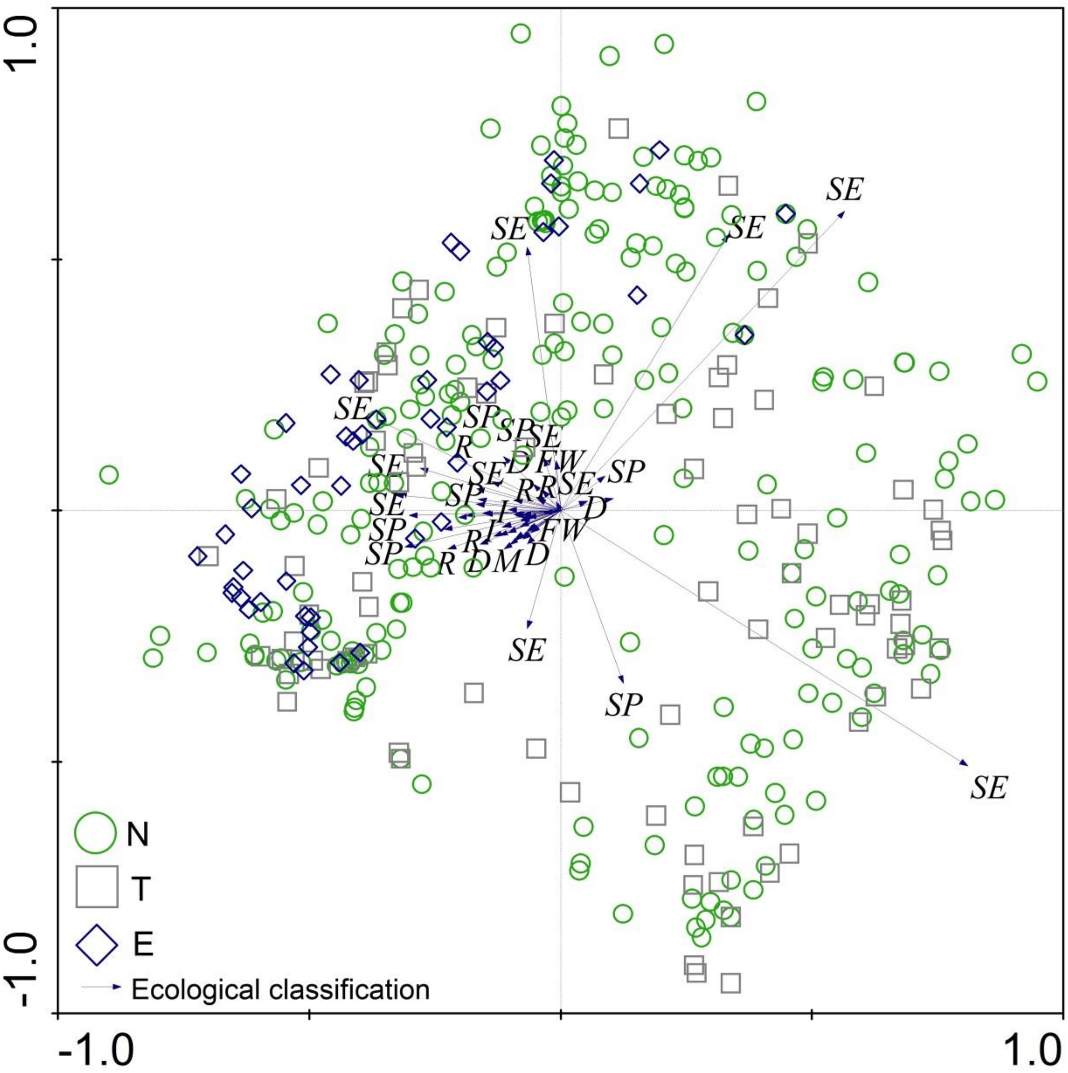


Figure 21. Multivariate analysis based on PCA for natural (N), tidally restored saltmarshes (T) and enclosed mix marshes (E), indicating main groups of saltmarsh communities according to ecological classification (SE-saltmarsh exclusive; SP-saltmarsh preferential; R-ruderal; I-invasive; FW-fresh water; DM-dry meadows; D-dunes)

Enclosed mix marshes seem to gather higher floristic diversity, as ruderal, invasive species and species that are typical of other habitats fall within this typology. Ubiquitous species (ruderal, fresh-water, dry meadows) appear to be almost transversal to the three typologies, but with low penetration in natural saltmarshes. The chart shows ubiquitous species in the central axes, corresponding to their wider distribution and preference for habitats with signs of disturbance (enclosed mix marshes and some tidally restored saltmarshes); allowing the entrance of some invasive species and adjacent habitat species (i.e. dunes). Saltmarsh preferential species (mostly within tidally restored saltmarshes) tend to be of broader onset, showing a distribution that is closer to natural saltmarshes.

5.4.2 Species colonization and similarity indexes

The similarity index (Byers & Chmura, 2007) was calculated to assess the relation of similarity between saltmarsh species in a natural habitat and the species found after the active restoration. This study used the floristic surveys available in the literature containing information on the species that colonize natural saltmarshes and also included post-intervention assessment on floristic composition. To simplify calculations, we noted the number of repeating species that appeared simultaneously in both inventories, reference natural saltmarshes and managed realignment (A); and the number of times that a certain species was exclusive to the pre-intervention saltmarsh (B) and the same for MR sites (C) (Table 17).

Table 17. Similarity index results and areas of application

Author and case	**Application**	**Si**
Byers & Chmura 2007 *Dike breaching*, North Atlantic (U.S.A.)	Natural saltmarshes towards diked marshes: two examples of each (DH/SR; WP/JL)	DH/SR=69.7 WP/JL=42.1
Loon-Steensma et al. 2015 *Managed realignment*, North Atlantic Europe	Dutch Wadden Sea reference marshes towards Grïe MR	57.3
Davy et al. 2011 *Tidal reactivation*, North Atlantic Europe	Brancaster MR *vs.* Enclosed mix marshes	20.7
Wolters et al. 2005 *Managed realignment*, North Atlantic Europe	Blackwater MR *vs.* Tidally restored saltmarshes	24.7
Test saltmarshes of Alvor and Arade	Natural saltmarshes *versus* Tidally restored saltmarshes	88.7
Passive recovery, Mediterranean saltmarshes	Natural saltmarshes *versus* Enclosed mix marshes	68.2

Here we compared pre-meditated MR options of actively recovering damaged saltmarshes with the absence of intervention and abandonment (passive recovery). Reference and managed realignment sites show values of similarity over 50%, Table 17 shows cases and application areas by author and the results for similarity calculations.

With the aim of studying recovery hypothesis of naturally breaching marshes in Fundy Bay (U.S.A.), Byers & Chmura (2007) combined vegetation analysis at two recovering and two reference marshes to assess the progress of recovery. Dipper Harbour-DH (natural) and Saint´s Rest-SR (diked) have a higher similarity (69.7). *Spartina alterniflora* and *Spartina patens* are the only common species in these four saltmarshes.

For the record, we excluded from the original surveys of van Loon-Steensma et al. (2015) frequencies of algae, fungi, bryophytes (absent from the reference saltmarsh but highly increasing in the MR survey) and arboreal species. A stone barrier was built around the marshes to prevent erosion, but simultaneously restricting the tidal flow. Species-by-species analysis demonstrated that almost 90% of the recorded *taxa* are found in the reference site, but only 49% colonized the MR marsh. Brackish communities tend to maintain but suffered substantial reductions. The similarity index calculated to compare the reference marsh with the outcome Grïe MR was of 57.3%.

Wolters et al., (2005, 2008) reported a de-embankment in a 21ha saltmarsh in the Blackwater estuary, South-East England, U.K.; pre-state was an embankment in the late 18th century and transformed into agricultural land. The species pool, site suitability, and regional/local species availability were brought together in a comprehensive study on target species colonization. The most relevant species are *Salicornia* ssp., *S. maritima, Aster tripolium* ssp. *pannonicus,* and *P. maritima*, which overlap with Wolters et al. (2008) data on target species arrival order: the first two species arrive early, and the second two species arrive a short while later. The site suitability approach taken by Wolters et al. (2005) made this case comparable with tidally restored saltmarshes, whose similarity index is 24.7%.

In 2002 tidal reactivation in a 7.5ha saltmarsh in Brancaster, North Norfolk, U.K. (Davy et al. 2011) brought a new life to the saltmarsh reclaimed during the18th century to provide a fresh water grazing area. The managed realignment marsh resulted in a large-scale environmental manipulation, where elevation and redox potential projected influences on species distribution – species relevance order: *Salicornia europaea*, *S.*, *Elytrigia atherica*, *H. portulacoides*. This is more similar to enclosed mix marshes (EMM) because of agriculture reclamation and habitat manipulation for fresh water species, therefore marsh communities disappeared during this process. The similarity index between Brancaster MR and EMM is 20.7%.

Regarding the test saltmarshes, comparisons between natural and tidally restored saltmarshes (TRS) delivered 88.7% of similarity. Among the 43 common species, they share the saltmarsh characteristic species and also most of the ruderal species. Only three are exclusive to the tidally restored habitat: the ruderal *A. arvensis* and *Sedum sediforme,* and the dune species *Mesembryanthemum nodiflorum*. The *Limonium* genus is mostly exclusive to natural saltmarshes. Considering the other typology, EMM share 30 species with natural saltmarshes. This includes saltmarsh exclusive and preferential species, with the exception of *S. maritima, Myriolimion diffusum, L. lanceolatum,* and *Limonium narbonense*. The six species that appear only in the EMM surveys are predominantly ruderal (i.e. *Cynara cardunculus, Melica minuta*) – here the similarity reaches 68%.

5.5 Discussion

Concepts of restoration differ according to authors, countries, and ecosystems, though major active recovery types share the same objectives: tidal reactivation, sediment supply, biotic production, regaining ecosystem services of degraded sub-habitat types and many more. Recovery is frequently used to describe all forms of improving ecosystem services, functions and goods, embracing secondary concepts, such as restoration, re-creation, enhancement, adaptation (Elliot et al., 2007). The core of the concept of recovery is given by the action: passive or active recovery. Formally, ecosystems that are in need of any intervention are triggered by natural or anthropic change. Moreover, anthropic involvement is required to react towards stressors, or natural processes within the ecosystem functions contribute to healing through passive recovery. The discussion bellow is guided by the hypothesis of higher floristic similarity among passive recovery saltmarshes than in active recovery works (i.e. managed realignment, restoration).

Conducting an active recovery project guides the path of the intervention, but also has an influence over its results, whether in terms of the species pool or the possibility of colonization. Species colonization patterns are less similar in MR sites (Dutch Wadden Sea SI=57.3) than those found in naturally recovering marshes (one case of Fundy Bay=69). In the case of van Loon-Steensma et al. (2015), erosion prevention and tidal flow limitation were the guidelines for Grïe MR, therefore there was a reduction of halophytes diversity. The species pool has a great penetration of freshwater communities into the MR site, with particular focus on the reduction of ruderal species. The similarities among reference and managed realignment were satisfactory, since sixteen species were exclusive to Grïe MR i.e. *Elytrigia* x *obtusiuscula*, *Bromus hordaceus, Solanum nigrum*. On the other hand, the fact that Fundy Bay dikes breached without human action (sea storms or coastal flooding) makes this particular example an interesting case to support the understanding of natural recovering marshes by providing evidence of proxy similarity indexes. The higher similarity between the first pair of compared saltmarshes (Byers & Chmura, 2007) is greatly induced by a higher variety of species distribution in the natural marsh, rather than a predominance of exclusive species of the diked marsh.

Besides being slightly different from other situations, Mossman et al., (2012b) case study shows the degree of success undertaken by a MR project on a year basis, providing good insights into vegetation recovery trajectories. Assessing a MR on a year basis contributes to monitor species colonization and arrival order, and improves knowledge on active recovery future strategies. Ruderal species (i.e. *Atriplex prostrata*, *Festuca rubra*) were absent in the first year of Brancaster MR (Mossman et al. 2012b) but tended to colonize significantly in the second and third years – ruderal i.e. *Cochlaria anglica* appears only in the last year of monitoring. Saltmarsh preferential (i.e. *Spergularia media*, *Juncus maritimus*, *Trignochin maritimam*) colonize from the second year on and become less frequent by the end of the fourth year of MR. Third and fourth years represent the state of maturity for the majority of species (higher frequencies and mean covers registered). A similarity index was applied to this case and it is interesting to notice that the similarity index approximates (66.7) the values of similarities of the passive recovery marshes of Alvor and Arade.

However, the differences increase when compared with MR and test saltmarshes (Alvor and Arade). The floristic distance between Atlantic and Mediterranean saltmarshes is notable due to the great variety of the species pool and habitat types. Surveys of the North Atlantic Europe were compared with all of the 368 surveys carried for Alvor and Arade (Mediterranean context); as a result, the similarity index is the lowest given by calculations (SI=18.2%). Low similarity indexes incorporate differences in the species pool, driven by biogeographic region and local dissimilarities in climate, sediment supply, and the regional species pool (Costal et al., 2009; Castillo et al., 2008). Additionally, dissimilarities increased by comparisons between MR projects and specific passive recoveries: tidally restored saltmarshes tend to show higher similarity with the MR data set (24.7) than enclosed mix marshes (20.7). Nevertheless, reference saltmarshes tend to be richer in saltmarsh exclusive and preferential species, while MR are dominated by ruderal and preferential species, and marked by some penetration of other species from adjacent habitats (dune, fresh water meadows). Moreover, fresh water availability increments the appearance of wet meadow species, mostly ruderal, which evidences the existence of disturbance, i.e. land displacement, hydrological correction, elevation changes (Jacobs et al., 2009; Esteves, 2014; Friess et al., 2014). A significant reduction of saltmarsh exclusive species´ frequency and cover suggests failure in the species pool colonization considering the habitat characteristic species. These MR projects might have targeted the species survival, rather than the endurance of saltmarsh characteristic species (Morris, 2013). The documented successes of the hybrid *Spartina anglica* (*Spartina.maritima* x *Spartina.alterniflora*) in the U.K. and in the U.S. saltmarshes is explained by the capacity of colonizing almost any level in the tidal range, occupying a level of limited competition in the low marsh band (Doody, 2008). Additionally, *S. anglica* is used in MR projects due to its characteristics as a sediment stabilizer, succeeding in erosion control projects and winning incentives for wetland creation policies (Doody, 2008).

Tidally restored saltmarshes present a high similarity index with natural saltmarshes, because their vegetation was not ripped out when diking took place, and also because no other ecosystem was truly implemented instead (Almeida et al., 2014). The establishment of target species in TRS was possible without anthropic intervention due to the presence of those species in the community species pool (Wolters et al., 2008), i.e. *S. maritima* tussocks colonizing the low marsh in TRS areas (Doody, 2008). *S. maritima* is prolific from rhizome fragments, showing higher sediment elevation in the center of the patch but also lower density of shoots (Marchant&Goodman, 1969; Sánchez et al., 2001), which makes it a week candidate for replantation projects for active recovery in saltmarshes, contrary to what happens with *S.densiflora* or *S.anglica* which propagules by seeds (Doody, 2008; Mateos-Naranjo et al., 2008)

Enclosed mix marsh lower similarity index derives from successive embankments and agricultural exploitation, since halophytes were replaced by beet or barley cultures (Alvim 1964). However, we can see from the similarities with natural saltmarshes that many species established and may develop successfully. In these cases, there is a more significant assortment of ubiquitous species than the characteristic species registered in the 5-year period studied by Mossman, et al. (2012b). Wolters et al. (2005, 2008) demonstrated an average of 74% of the species present in the local species pool was established in the restoration sites at most 13 years after de-embankment. Although for the regional species pool, only an average of 45% of the species managed to establish for passive recovery success, the great relevance of the availability and dispersal of the target species from the natural saltmarsh to the restored area, in which target species should arrive spontaneously (Wolters et al., 2008), is demonstrated. Without recovery or any other managed realignment objectives guiding the destiny of the naturally restored saltmarshes of Alvor and Arade, the species going from natural saltmarshes to the other topologies are faithful to the natural saltmarsh species pool and structure (*Arthrocnemum macrostachyum*, *Halimione portulacoides*, *Sarcocornia perennis* ssp. *perennis*, *Sarcocornia perennis* ssp. *alpini* and *Sarcocornia pruinosa*). Gaps are found in low marsh pioneer *S. maritima* or *Bolboschoenus maritimus* var. *maritimus*, related with elevation and incoming tide succession. Despite this fact, biodiversity is an unnecessary gain of this particular ecological evolution, which in active recovery projects can be easily controlled, as well as invasive species penetration. The international examples of MR analyzed demonstrate a loss in some saltmarsh characteristic species (loss of exclusive species), rather than an increase in diversity (Loon-Steensma et al., 2015).

Species recovery capability is of various origins and within this investigation we are circumscribing it to the frequency of occurrence after anthropic interventions. Additionally, plant communities on restored saltmarshes tend to show a slower development when compared with natural sites (Mossman et al., 2012b). We would like to stress that species emergence can be strongly related with the objectives set for the active recovery – excavating and planting new sites in an effort to restore the Tijuana estuary (U.S.A.) is an example of short-lived species being outcompeted by perennials in the absence of topographic heterogeneity (Zedler & West, 2008). Species introduction would favor some species over others (Strange et al., 2002) but if the target is hydrological restoration (Davy et al., 2011), floristic diversity would not be on the top priorities. For Mediterranean marshes, Castillo et al., (2008) demonstrated the competitive potential of invasive *Spartina. densiflora* in colonizing the center of circular *S. maritima* native species, by altering the vegetational zonation. Fourthly, as Esteves (2014) demonstrated, the decay of accidentally breached dikes or structures may produce similar results to those of managed realignment. Floristic dissimilarities are found in restoration sites even after long time periods, raising doubts as to the sustainability of these projects. A destructed natural habitat can be replaced with an artificial one (Briggs et al., 2009), comprising biodiversity changes and other environmental impacts, regardless of the ecological outcome (Stagg & Mendelssohn, 2010; Chapman & Underwood, 2011). This work can contribute to further investigation on the causes of colonizing species derived from passive recovery and provide a framework for guided interventions to enhance saltmarsh passive recovery.

5.6 Conclusions

Despite the causes of the differences in the colonization patterns of restored saltmarshes, and regarding this research, species appear to be within a local and regional species pool. Although saltmarsh characteristic species decreased, the multivariate analysis showed that characteristic species of tidally restored and enclosed mix marshes match those of natural saltmarshes, providing a solid basis to support passive recovery within Mediterranean saltmarshes.

The absence of reference MR projects in Portuguese saltmarshes that had been monitored over the years and provided a time-series of relevant data for comparison is a limitation of our study. Our findings are consistence with our hypothesis: a higher similarity of the flora of test saltmarshes both when compared with reference saltmarshes and managed realignment. Additionally, it supports the distance of floristic composition among Atlantic and Mediterranean saltmarshes and confirms lower similarities between natural and managed realignment sites. As for the hypothesis proposed in this work, results attest that passive recovery delivers a higher similarity index supported by the regional and local species pool. Nevertheless, active recovery can play a fundamental role when environmental catastrophes take place or characteristic species are unable to germinate or root.

Chapter VI – Floristic Catalogue

Nomenclature follows the main flora works by Castroviejo (1986-2012), Franco (1971, 1984), Franco & Rocha Afonso (1994, 1998, and 2003) and Costa et al. (2012).

It follows the floristic catalogue of the saltmarsh and transitional marsh communities found within the study area, organized by family, genus, species, subspecies and variety; also indicating their ecological classification. Within the study area, there were found 64 *taxa*, 46 genus and 18 families.

AIZOACEAE

Carpobrotus edulis (L.) N.E.Br.

Camephyte

Invasive

Mesembryanthemum nodiflorum, L.

Terophyte

APIACEAE

Ferula tingitana*,* L.

Hemicryptophyte

ASPARGACEAE

Asparagus albus, L.

Fanerophyte

ASTERACEAE

Artemisia crithmifolia L.
Camephyte ligneous
European endemism

Artemisia gallica Willd. subsp. ***gallica*** = *Artemisia caerulescens* L. subsp. *caerulecens*

Camephyte ligneous

Aster tripolium L. subsp**.** ***pannonnicus*** (Jacq.) Soó
Hemicryptophyte

Calendula arvense L.
Terophyte

Cotula coronopifolia L
Terophyte
Invasive

Cynara cardunculus L
Hemicryptophyte subarrosetado

Hypochaeris radicata L. subsp. ***radicata***
Hemicryptophyte subarrosetado

Inula critmoides L.
Camephyte ligneous

Sonchus maritimus, L.

Geophyte

Taraxacum officinale, (L.) Weber ex F.H. Wigg

Terophyte

CARYOPHYLLACEAE

Spergularia bocconei, (Scheele) Graebn. in Asch. & Graebn.

Terophyte

Spergularia media (L.) C. Presl.

Camephyte , Hemicryptophyte

CHENOPODEACEA

Arthrocnemum macrostachyum, (Moric.) Morris

Camephyte ligneous

Atriplex halimus, L.

Fanerophyte, Camephyte

Atriplex patula, L.

Terophyte

Atriplex prostrata, Boucher ex DC. in Lam. & DC.
Terophyte

Halimione **portulacoides**, (L.) Aellen
Camephyte ligneous

Polycnemum arvence, L.
Terophyte

Salicornia fragilis P.W. Ball & Tutin
Terophyte

Salicornia patula Duval-Juv
Terophyte

Salicornia ramosissima J. Woods
Terophyte
Salsola soda, L.
Terophyte

Salsola vermiculata, L.
Camephyte ligneous

Sarcocornia pruinosa Fuente, Rufo & Sánchez-Mata (***Sarcocornia pruinosa***, (L.) A.J. Scott Sensu aa. lus.)
Camephyte ligneous

Sarcocornia perennis, (Mill.) A.J. Scott subsp**. *perennis***

Camephyte

Sarcocornia perennis subsp**. *alpini*** (Lag.) Castrov.

Suaeda albescens Lázaro Ibiza

Terophyte

Suaeda vera Forssk. ex J.F. Gmel.

Camephyte, Nanofanerophyte

CRASSULACEAE

Sedum sediforme, (Jacq.) Pau

Camephyte

CYPERACEAE

Bolboschoenus glaucus (Lam.) S.G. Sm.

Geophyte

Bolboschoenus maritimus (L.) Palla in W.D.J. Koch var. maritimus (= Scirpus maritimus L. var. maritimus)

Geophyte rhizomatous

Bolboschoenus maritimus (L.) Palla in W.D.J. Koch var. compactus (Hofmm.) Lej. (= Scirpus maritimus L. var. . compactus (Hofmm.) Lej.)

Geophyte

FABACEAE

Lotus creticus, L.

Proto-hemicryptophyte

Medicargo polymorpha, L.

Terophyte

Scorpiurus vermiculata, L.

Terophyte

FRANKENIACEA

Frankenia laevis, L.

Camephyte

JUNCACEAE

Juncus acutus, L.

Hemicryptophyte, Helophyte

Juncus maritimus, Lam.

Geophyte rhizomatous

LEGUMINOSAE

Mellilotus segetalis, (Brot.) Ser. in DC

Terophyte

OROBANCHACEAE

Cistanche phelypaea (L.) Cout.

Geophyte

OXALIDACEAE

Oxalis pes-caprae, L.

Geophyte

Invasive

PLUMBAGINACEAE

Limonium algarvense, Erben

Hemicryptophyte

Limoniastrum monopetalum, (L.) Boiss. in DC.

Fanerophyte

Limonium lanceolatum, (Hoffmanns. & Link) Franco

Camephyte

Portuguese endemism

Limonium vulgare, Mill.

Camephyte

Myriolimion diffusum (Pourr.) Lledó, Erben & Crespo (= *Limonium diffusum* Pourr.)

Camephyte

Myriolimon ferulaceum (L.) Lledó, Erben & Crespo (= *Limonium ferulacem* L.)

Camephyte

POACEA

Brachypodium phoenicoides, (L.) Roem. & Schult.

Hemicryptophyte

Bromus lanceolatus, Roth

Terophyte

Digitaria sanguinalis, (L.) Scop.

Terophyte

Invasive

Elytrigia elongata (Host) Nevski (= *Elymus elongatus*, (Host) Runemark)

Hemicryptophyte

Elytrigia juncea (L.) Nevski subsp. ***boreoatlantica*** (Simonet & Guin.) Hyl. (=*Elymus fartus* (Viv.) Runemark ex Melderis subsp. borealiatlantica (Simonet & Guin.) Hyl.)

Hemicryptophyte

Melica minuta, L.

Proto- Hemicryptophyte

Polypogon maritimus, Willd.
Terophyte

Puccinellia iberica, (Wolley-Dod) Tzvelev
Hemicryptophyte

Spartina maritima, (Courtis) Fernald
Proto-hemicryptophyte

Sporobolus pungens, (Schreb.) Kunth
Proto-hemicryptophyte

POLYGONACEAE

Emex spinosa, (L.) Campd.
Terophyte

PRIMULACEAE

Anagallis arvenses, L.
Terophyte

Chapter VII – Discussion and Conclusion

7.1 Introduction

As demonstrated in the previous chapters, Alvor and Arade saltmarshes' resilience and conflict response capacity gave them the ability to adapt and evolve into differentiated typologies. These typologies present distinct markers of land use and land cover, and different vegetation patterns. Adittionally, they developed differently in terms of sedimentation and grain size. The deeper understanding of the ecology and dynamics of these Mediterranean saltmarshes has highlighted habitat management priorities that the restoration policy should adress. This section discusses the results accomplished in the previous chapters and contributes to develop a framework to guide restoration options in terms of habitat management. The hypotheses raised in the first chapter will be discussed in the following sections and finally summarized in a table that compiles the responses to each hypothesis and the corresponding chapter where it was dealt with.

7.2 Pressures

7.2.1 Pressures from land-use changes

In the Algarve (Alvor and Arade), results have shown significan changes in land use during two time periods, which can be explained by major social transformations in society that have affectedeconomic and social options, as well as political decisions: a massive investment in agriculture from 1953 to 1973 across the country (Planos de Fomento), which despite the profund assessment of the available resources that was previously made did not result in the desirable effects everywhere in the country, particularly in Alvor and Arade. Between a political shift in 1975 and the integration of Portugal into the EEC in 1986, the old guidelines towards saltmarsh reclamation for agriculture stopped dramatically, leaving several hectares of saltmarsh abandoned. As a consequence of this abandonment, saltmarsh ecosystems found the place to redistribute flows and species across the landscape, according to the regional species pool. Other forms of vegetation patterns, grain size distribution and ecosystem functions and services arose, modifying the characteristics of the habitats.

Together with these transformations, some pressures derived from urban and touristic projects were also identified. The attractiveness of the coastaline has increased the demand for infrastructures to improve the mass delivery of residential and seasonal spring-summer touristic services (PENT, 2012), which are a cause for concern due to urban development and land exploitation. The Algarve region has suffered from decades of land-use and land-cover changes, resulting in the reclamation of natural areas to address increasing construction, population, and tourism needs (Almeida, 2012). Not surprisingly, the most affected areas of land reclamation are within the coastaline, endangering the most valuable natural habitats in terms of protection and conservation in Portugal, the groundwater table, and shore stability. Both municipalities in which Alvor (Lagos-Portimão) and Arade (Portimão-Silves) are located share the urban areas that have expanded the most in the last decades, and are also those with the longest touristic stays: hotels, golf camps, private beach resorts, a racing circuit, an aerodrome, apartment complexes, exibition pavilions, and camping sites (Petrov et al., 2009) are some examples of the vast and varied touristic offer with known impacts in the watersheds of Alvor and Arde, mainly nutrient enrichment via groundwater or surface runoff (PROT-Algarve, 2004).

7.2.2 Pressures from short-term accretion and erosion dynamics

Different rates of short-term accretion were registered in the saltmarsh typologies. The estuarine fringing marsh, as the natural saltmarshes of Arade (sample site 2, see Introduction, Figure 4), registered a considerably high erosion rate, although it is the only saltmarsh in a declining situation. On the other hand, more sheltered areas, such as tidally restored saltmarshes, tend to have very low accretion rates, associated with the rhythmic entrance of the tide, as well as the lower elevation where they develop the most — these differences in elevation between natural and TRS are related with the abandonment process which different saltmarshes went through. A vast majority of Arade TRS were excavated and drained for agricultural purposes, which had a significant impact on lower elevations, even when compared to natural saltmarshes. The third level of accretion (higher rates) is seen in the typology of enclosed mix marshes, framed by a former agricultural usage, and a significant presence of coarser sediments (higher percentages of sand). Grain size analysis confirms these findings, as natural saltmarshes were expected to be richer in silt and clay; however, in Laguna de Alvor, a mature saltmarsh community is colonizing sand banks. These sand bank communities are contextualized by the geomorphology of the Laguna, classified as a coastal lagoon greatly influenced by marine sediments. In the Arade river, the riverine background provided a greater abundance of silty and clayey saltmarshes, as well as a higher concentration of organic material.

These findings play a crucial role on vegetation distribution, and may influence the landscape metrics (patch size, edge density, shape, and connectivity at patch and class level).

7.2.3 Pressures from connectivity and complexity

Landscape metrics allowed a comprehensive knowledge of saltmarsh patches, their position within the typology, and the identification of different relations of connectivity and complexity among and within different patches. Comparisons made between 1972 and 2010 brought to evidence changes in the landscape types with respect to complexity, which have major implications on connectivity issues. Nevertheless, natural saltmarshes tended to be made of elongated and edgy patches, within a short distance from neighbouring patches, providing sustained complexity of the landscape and also good connectivity among patches (in 1972). The other typologies presented a more fragmented landscape. After forty years (in 2010) of land-use and land-cover transformations, secondary saltmarshes consolidated as independent areas, since they grew in patch number and size, and delivered a good performance in terms of shape, fractal dimension, and connectivity metrics. Regarding enclosed mix marshes, in which tidal influence plays a part due to the combined factors of salt ascension and freshwater accumulation, transformations occurred towards patch size reduction, simpler shapes, and poor connectivity. Patches with higher values belong to low and medium saltmarshes from the tidally restored typologies, supporting the emergence of these secondary marshes as a recovery of natural saltmarshes under environmental pressure. Considering these pressures, it must be said that fragmentation is a threat mainly to natural saltmarshes and EMM.

Patch connectivity and complexity influence the pressures on plant communities, and ultimately play an important role on the recovery options or restoration projects to be undertaken in these areas.

7.2.4 Pressures from environmental change

Climate and environmental changes pose an increasing stress on the primary production of the ecosystems and severely affect biodiversity in terms of genetics, species, and at the biome level (Rinawati et al., 2013). Predictions for the Mediterranean Region point to an increase in repeated days with high temperatures and a reduction in humidity, as well as changes in the hydrologic regime, with long dry periods in summer and torrential rain peaks in winter (IPCC 2014). Environmental stress caused by drier and warmer periods may lead to changes in floristic composition, mortality rates, and the colonization by herbaceous invasive species. Additionally, predictions indicate an acceleration in rising sea levels, in which coastal habitats face major challenges concerning sea storms and coastal flooding (LPN 2014, IPCC 2014).

The impact of sea level rise on coastal ecosystems has been widely studied (Moreira, 1992; Eliot et al., 1999; Miller et al., 2001; van Wijnen & Bakker, 2001; Wu et al., 2002; Cahoon et al., 2006; Ferreira et al., 2008; Marfar & King, 2008; Vermeer & Rahmstorf, 2009), strengthening the link between environmental change and rising sea levels — IPCC (2007) refers that an increase by 3ºC on average temperatures may consequently contribute to a loss of 30% of coastal wetlands. According to Bindoff et al. (2007), during the 20th century, there was an increase in the global mean sea level by 1.7 ± 0.5 mm/year, while ocean temperatures have increased by 0.6ºC since 1950. These values are associated with a warming effect in the coastal atmosphere, which has been suffering from decades of anthropic pressure.

Saltmarshes are no exception to these scenarios of environmental change, since they represent a fringing habitat, and are therefore directly vulnerable to anthropic activities and sea level rise. The erosion of mudflat platforms and the retreat of saltmarshes have been gradually occurring, especially regarding the compactness and the shrinkage of those saltmarshes facing artificial barriers that are preventing them from migrating inlands — in the Arade case study, registers of erosion in the natural marsh slope may indicate a loss of saltmarsh area in that segment, probably due to enhanced hydrodynamics in the presence of engineering structures (e.g. around bridge pilings or in front of dykes). Modelling results (Teles et al., 2013) show that during ebb tide in the Arade, there is an intensification of tidal flow speed (0.25 to 0.50 m/s) close to the sampling area 2 (see Figure 3) forced by the presence of the bridge (for further information on this topic, see Appendix B). The intensification of the tidal flow leads to the erosion of the marsh scarp (loss of 22.55 mm/year) and the resulting loss in vegetation cover, as reported elsewhere (Moreira, 1992; Allen, 2000; Marani et al., 2011; Francalanci et al., 2013).

7.2.5 Pressures on plant communities

Plant communities compete for survival along transects of tolerance and adaptation. Comparisons between vegetation surveys of natural and secondary saltmarshes (Chapter 4) showed that naturally recovering saltmarshes have greater similarities in terms of vegetation composition than managed realignment datasets. Saltmarsh plant communities of the secondary marshes face pressures derived from (i) land-use changes, (ii) abandonment processes, (iii) dissimilarities in sedimentation, and (iv) pressures resulting from patch size and fragmentation. Saltmarsh-characteristic species are of a narrow ecological amplitude, while preferential species tend to appear associated with the parameters of environmental variables (grain size, organic material concentration, elevation, salinity, inundation, among others). Low marsh colonizers present dispersion variations according to the typology of the saltmarsh, and are found particularly in natural saltmarshes and with less abundance in TRS. The floristic pattern verified in the saltmarshes restored by tide is identical to saltmarshes in their natural state, which is a reason for concern regarding ecosystem evolution, resilience, and adaptation.

Diversity is found within enclosed mix marshes, sharing ruderal and some invasive species with tidally restored saltmarshes. In the Alvor case, pioneer saltmarsh species share the habitat with dune communities, particularly in the high saltmarsh and transitional areas, such as enclosed mix marshes. Thus, similarities between natural and tidally restored salt marshes are within the local species pool, while in the case of enclosed mix marshes, pressures from the adjacent habitats, such as dune and/or fresh water meadows, as well as ruderal and invasive species are eminent. The differences found between the study areas are related to local specificities of grain size and both complexity and connectivity characteristics, but also with local patterns of land-use changes: while in Arade, agriculture was poorly developed and soon abandoned allowing a secondary marsh to emerge with a stronger relation with tide and the local species pool (provided by more abundant and remaining natural marshes), in Alvor, enclosed mix marshes are more frequent due to the reuse of former crops for salt exploitation and for aquaculture afterwards, delaying any chance of a secondary marsh to develop, eliminating floristic similarities within the local species pool.

7.3 Challenges

Alvor and Arade saltmarshes are dealing with major challenges to address the complex mesh of anthropic and physical pressures they have been put through. These challenges are not necessarily a target response to pressures, but instead a guide for policy making — this section lists the top challenges the case study saltmarshes are facing due to lack of habitat management options. The first challenge is the ecosystem services provided or delivered by these saltmarshes. Chapter IV provided the identification of ecosystem services through the use of landscape metrics as a tool to achieve specific services or functions. It allowed to differentiate levels of coastal defence by saltmarsh typology and to evaluate how these typologies would effectively imply a differentiation on ecosystem service delivery. The — service of coastal protection was selected, and subsequently an index of defence was calculated for each saltmarsh typology, later linked to studies on vegetation recovery, based on the hypothesis of passive recovery of the plant communities (Chapter V). For this reason, other options of restoration versus recovery in the scope of the challenges faced by saltmarshes shall be addressed with the purpose of informing the future management of the studied saltmarshes.

7.3.1. Ecosystem services

Ecosystem services and functioning are frequently confused (Wallace, 2007): services may represent an economic value to society (directly or indirectly) and functions possess an intrinsic value (see Figure 22). King & Lester (1995) and Costanza et al. (2014) mentioned three factors that influence the value attribution regarding ecosystem services: 1) the majority of ecosystem services are public, which includes a sustainable use by the present and future population; 2) many of those services are affectd by external factors, i.e. a polluted estuary would affect the quality of the fish, with consequences to bird nesting; 3) ownership issues may interfere with the ecosystem service delivery, e.g. a private property or unknow ownership may disturb the legal interventions to manage slatmarsh areas (for further discussion on this topic, see section 7.3.3).

Considering planning purposes, knowing the exact value of the ecosystem services may provide a suitable background for intervention, but the calculations using dollar-based metrics are not consensual among authors (de Groot et al., 2002; Costanza, 2008; Fisher & Turner, 2008; Fisher et al., 2009). There is a debate on the complexity underlying the factors that sustain the attribution of an economic value and highlighting the existence of benefits (i.e. esthetical, cultural, recreation values and the right of existence *per se*) that cannot be quantified in a monetary sense.

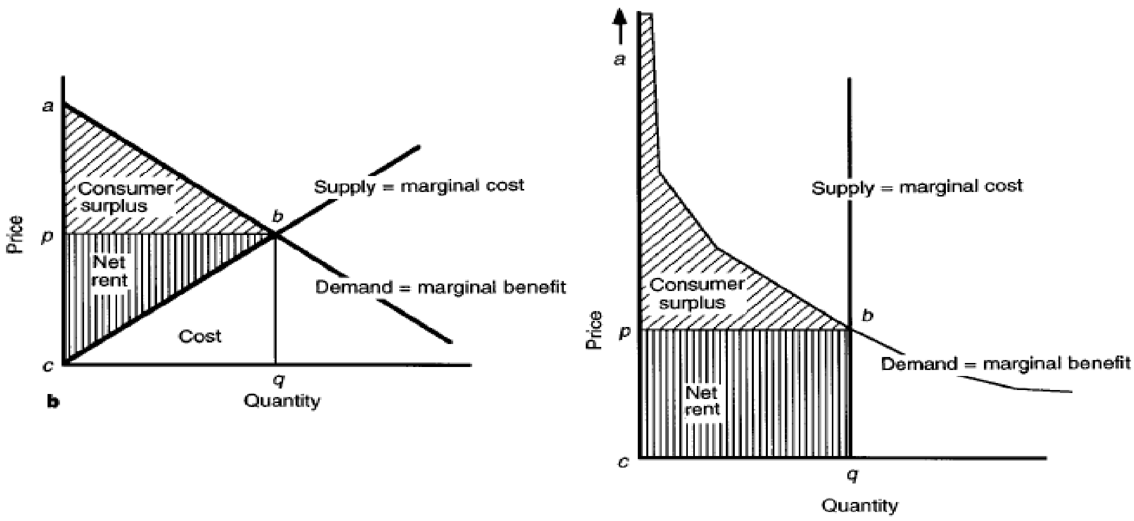


Figure 22. Supply and demand curves, showing costs, profitability, and consumption (Costanza et al., 1997).

The final and the intermediate products (from ecosystems) are linked by biodiversity and services overlapped by functions, which ultimately raises concerns about the economic and social values to address costs associated with maintenance and restoration (Markandaya et al., 2008). Biological diversity is a key aspect in the delivery of ecosystem services; usually, higher biodiversity is related with improved quality and quantity of the ecosystem services, where resilience is an essential condition. Resilience depends on many factors, i.e. higher inter-specific and gene variety in the same structure (M.A., 2005), but according to Gedan et al. (2009), resilience in the face of anthropic and natural impacts is the key aspect to gather interest on the identification and assessment of ecosystem services, i.e. this is 'ecosystem-based management' (Gedan et al., 2009). In fact, the notion and measurement of ecosystem services have been integrated into European and US policy; however, the initial focus was on environmental compensation (Möller et al., 2001; Feagin et al., 2010). Saltmarshes appear on the top studied ecosystems due to their diversified regulating and support services and their resilience and capacity of plant colonization, arguments that have been used to support restoration projects.

Devising saltmarsh recovery as a form of promoting ecosystem services finds resonance in the work developed by Möller et al. (2001), who studied the possibility of a 80 m health and mature saltmarsh to reduce 40% of the wave energy. The lower elevation offered by coastal or estuarine saltmarshes plus their plant cover are the main ingredients to absorb the overtopping wave, reducing its intensity as it reaches inlands. Additionally, Feagin et al. (2009) showed that saltmarsh vegetation plays a significant role in modifying sediment dynamics, avoiding lateral erosion that results from sea level rise. The effectiveness of saltmarshes in responding to sea level rise is tied with the fact that vegetation is able to develop at different elevations and different sucessional states, and is therefore able to provide a great variability in terms of abortion and reduction of flooding velocity (Morris et al., 2002; Feagin, 2008; Watson & Byrne, 2009; Reeve & Karunarathna, 2009; Feagin et al., 2010; Kim et al., 2011b).

Arguments for policy shifting regarding saltmarsh habitat restoration are sustained by the work of Möller et al. (2001) and Möller (2006), whose results were used to assess maintenance costs of breakwaters in the UK. Saltmarsh ecosystems are currently at the edge of foreign policy — UK (*Making Space for Water*), Germany (*Water Protection Policy*) or Netherlands (*Water Governance in Netherlands*) — integrating ecological restoration into local planning (Esteves, 2013) as a substitute for the previous 'hold-the-line' measures. Ledoux et al. (2005) refer that the former coastal protection measures based on the use of concrete are no longer viable in economic terms, they are not in line with social awareness, besides representing a barrier to ecological structures.

Contrasting with the great coastal defence engineering works that have taken place in Europe during last centuries, Alvor and Arade seem to demonstrate a slow recovery of their saltmarsh ecosystems, mainly due to land-use changes and resulting from long-term passive de-embankment. Allen (2000), Doody (2008) and Esteves (2013) refer significant changes in the UK coastline, where the first engineer works (hard-measures) tried to mitigate major sea storm damages to coast and people. After dredging, embankment works, and saltmarsh reclamation for agriculture (Robinson et al., 2004; Garbutt & Wolters, 2008; Barkowski et al., 2009), the coastal line continued to change originating a massive coastal erosion and wetlands habitat shrinkage, altering the political concerns for coastal management (Markandya et al., 2008). Sea level rise and environmental change, which gave place to significant sea storms, coastal flooding and the destruction of concrete breakwaters, opened up space for a rationale based on soft-measures. Within soft-measures, there is saltmarsh recreation and other wetland recovery using environmentally friendly options, relying on an ecosystem service boost (Spencer & Harvey, 2012a). Crépin (2005) and Wolters et al. (2008) describe soft measures as returning the ecosystem its original functions, or generating natural conditions for those habitats that are similar to their state before any disturbances. Alvor and Arade saltmarshes responded naturarlly to damage, embankement, desiccation, and other disturbances previous to abandonment, where resilence and the local species pool in the nearby secondary marshes allowed the development of new saltmarsh typologies adapted to mix grain size and tidal influence. Despite adversity, the coastal defence index is higher in tidally restored and enclosed mix marshes than in natural saltmarshes, proving their value as passively recovered and consolidated marshes.

Here lies the ultimate challenge: Do we want to create new saltmarshes to achieve more or improved ecosystem services? Or should we enhance marsh charateristics through recovery to accomplish environmental metrics and account for coastal defence?

7.3.2 Recovery or restoration?

The evidences that stem from this work place reclaimed Alvor and Arade saltmarshes in a context of passive recovery, assuming that the abandonment of dyke structures, saltpans, and agricultural usage implied an opportunity for saltmarshes to develop — new land uses and different saltmarsh typologies gained distinctiveness in the Alvor and Arade landscapes.

The triggering factors

The interest on saltmarsh recovery and habitat restoration started to be associated with rapid changes in erosion and accretion rates that endangered the human communities living in the borderline saltmarsh (Boorman et al., 2002; Doddy, 2013). Low-lying areas left by successive embankments and land reclamation over centuries fail in the absorption of wave energy and in the capacity to accommodate flooding. These aspects started to be valued in a context of climate change, particularly rising sea levels. These concerns first led to the construction of breakwater structures (Elliot et al., 2007; Doody, 2013). Nevertheless, these hard measures started to degrade and sustainability issues arose whenever breakwaters and groynes needed repairing or reconstruction (Esteves, 2014).

Despite the consequences that hard measures represent for the overall ecosystem functioning (nearshore erosion, changes in sediment supply, among others), ecosystem services did not represent a major concern during the late 1980s and early 1990s, regarding the resilience of coastal ecosystems. Aiming to reverse some of the abovementioned problems and driven by sustainability awareness, some surgical measures (soft measures) were taken to contain the path of erosion and biodiversity loss experienced by coastal ecosystems, mainly saltmarshes.

Among these soft measures, some definitions are frequently confused (i.e. restoration, recovery, re-creation, managed realignment), but all of them represent different approaches to habitat or ecosystem intervention towards improving state-based conditions, regardless of the final objective or the disturbance typology. Costanza et al. (1997), Strange et al. (2002), and

English et al. (2009) argue that compensatory measures should not be guided only by ecological parameters, but should also be regulated by economic models to assure quantification.

<u>Time-line intervention: passive or active recovery?</u>

The triggering factors raise questions about the timeline between damage and intervention, and also about recovery paths. As Figure 23 exemplifies, recovery can find two paths — active or passive — which most often depend on the typology of the stressor or damage and are also closely related to the time between events. Most active recoveries are intended to fight stressors, re-establish lost habitats, or act as compensatory measures due to a specific damage; ecosystem services can be included in those active recovery projects or not, but enhancing previous recovery works tends to be more associated with the option of improving or giving back ecosystem services. On the other hand, passive intervention on ecosystems can be taken wilfully using soft measures, or it can occur naturally as a response of the ecosystems to a specific disturbance. Concepts underlying passive recovery are connected with the internal aspects of the ecosystem, such as its resilience and caring capacity, plus the external factors that create the opportunity, such as abandonment or causality — the ultimate objective is adaptation, instead of compensation (see Figure 23).

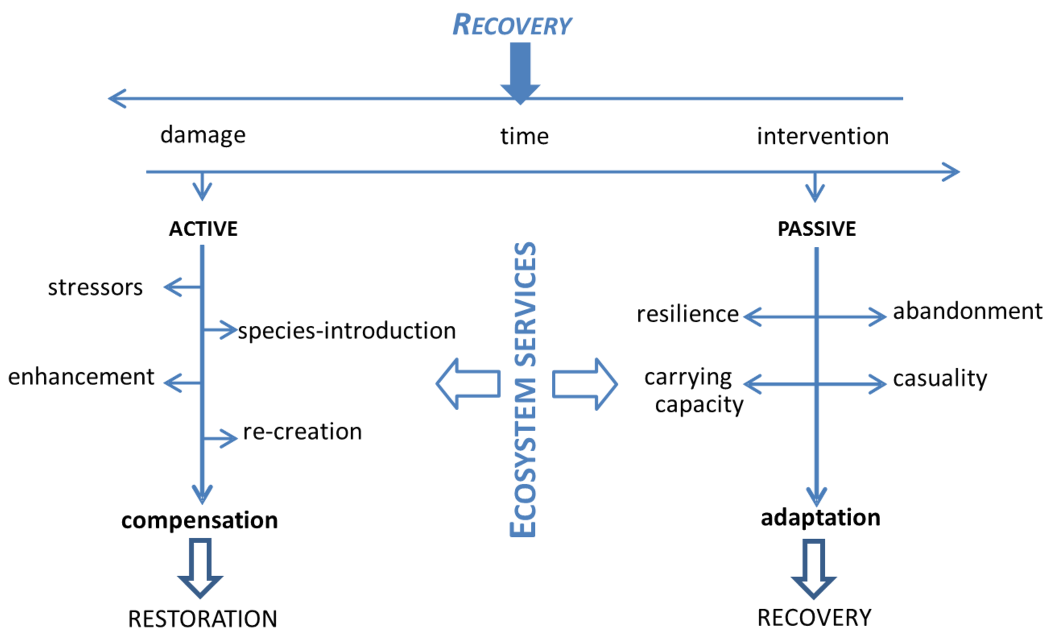


Figure 23. Deconstructing recovery processes (adapted from Elliot et al., 2007; Duarte et al., 2013).

Recovery is frequently used to describe all forms of improving ecosystem services, functions, and goods, embracing other additional concepts, such as restoration, re-creation, enhancement, and adaptation (Elliot et al., 2007). Central to the entire concept of recovery is the action: passive or active (see Figure 23). Elliot et al. (2007) present a definition of recoverability (quoting MarLIN Glossary, 2005), which is the concept to be used in this thesis: recoverability is ´the ability of a habitat, community or individual (or individual colony) of species to redress damage sustained as a result of an external factor´. This definition is closely related to two other concepts inherent to natural ecosystems: 1) 'carrying capacity' indicates the maximum limit of an ecosystem to support successive interferences in its functioning; it implies a balanced relation between disturbance and adaptation, meaning that the ecosystem has the capacity to endure some disturbance, altering interior processes in order to respond to stressors; adaptation is a strategy that results from the carrying capacity, which can be translated into resilience; 2) 'resilience' is noted as the capacity of an ecosystem to return to its original functioning after a damage or disturbance — this implies a relation between time and balance. Formally, ecosystems that need any intervention are triggered by natural or anthropic change; furthermore, anthropic involvement is required to react to stressors, or natural processes within ecosystem functions that contribute to healing through passive recovery.

7.3.3 Saltmarsh ownership, property and public domain

Human occupation within the Alvor and Arade watersheds dates back to the Neolithic and Chalcolithic periods (2000-1600 BC). Original designations (Roman and Arabic) of the urban areas of Portimão (*Portus Hannibalis*) and 'Villa de Alvor' (see Figure 1, Chapter I) referred to the amenity of sea waters and land fertility (Loureiro, 1904). The most appealing features of these areas are related with agriculture, fishing and fish curing using salt, and saltpans exploitation, which were supported by sea (Alvor and Arade) and river ports (in the case of Arade), generating a significant long-lasting economic dynamism (Loureiro, 1904). Urban and economic development anchored its foundation in the strong relation with saltmarshes and other wetlands, and also in the activities derived from the primary sector, maintaining a great dependence on the river and the ocean. These physical conditions and economic vitality attracted an elite of merchants, to whom later on property and exploitation rights were conceded, namely over the saltmarsh areas of Boina (Arade´s tributary) and Morgado de Arge (inland Boina creek) (Vidigal, 1993; Sampaio, 2009) (see Appendix A). Changes in saltmarsh location and consequent ownership occurred as a result of the tsunami caused by an earthquake in 1755. Alvor´s coastal morphology was profoundly transformed, as well as its river basins, in which the sea entered 667 m inlands; these factors also contributed to transformations in crop production and salt exploitation (Loureiro, 1090). As a consequence of these changes, saltmarshes assumed a new position in the Laguna de Alvor, causing the need for a dyke aiming at farm irrigation in Mexelhoeira Grande and Arão creeks — this dyking process resulted in the siltation of some parts of the Laguna, and determined the end of the sea port of Alvor (Mariano, 2010).

Later, in 1818, successive embankments (as a consequence of the tsunami effects) contributed to several saltmarsh reclamations (Vieira, 1911). Salt wetlands of Boina and Arge (Arade) were dyked aiming to develop agriculture (rye, oats, and lupins), transforming saltmarshes into private property (Vieira, 1911) — salt wetlands and marsh areas represent 1/3 of the municipality of Portimão. Saltmarsh reclamation and embankments lasted until the first half of the 20th century, associated not only with agriculture or industrial uses, but also with urban expansion, and, afterwards, touristic demand — private property and real estate speculation.

In the UK, policy undertaken to manage saltmarsh areas has implemented a 'landowner compensation' agreement, in which the retreat area is being purchased from landowners (DEFRA, 2008) — the Programme implemented by the Environment Agency Flood and Coastal Erosion Risk Management R&D (UK) has the capacity to finance the creation of 'new' saltmarshes as an integral part of the flood defence areas (Saltmarsh management manual, 2007).

Legal restrictions: Public Hydric Domain (Law No.468, 5th November 1971) and Law of Water (Law No.58, 29th December 2005)

Framed by the European Directive of Water (2000/60/CE), the Law of Water (No. 58/2005) aims to protect, preserve and assure the sustainability of hydric resources. The Hydric Public Domain (HPD) is one of the oldest laws (regal law from 1864) and constitutes the base for coastal policy (civil code from 1867). It defines a 50-meter territorial portion as State property of public utility; it is inalienable, unpledged and imprescriptible, integrating all types of public waters, both natural or artificial, comprising maritime, fluvial and lacustrine domains, as well as water channels, waterways, open ditches and respective margin beds, natural swamps, pluvial waters, below ground waters, built reservoirs, and public fountains. The assets within the HPD may be of private use by Government authorization and concession licence. Several amendments to the law have been made over the years, but Law No. 78/2013 introduced a timeline for private ownership to be recognized. Pursuant to this law, private holders had to request the public recognition of their properties and were obliged to make requalification works in the real estate and clearly define the limits of their property, i.e. in the case of upstream

Arade saltmarshes (Figure 24), a former abandoned watermill was intervened and the surrounding marshes were bordered.

Figure 24. Saltmarsh with a fence, upstream Arade river (by the author, December 2013).

<u>Real estate speculation vs. environmental compensation: saltmarshes to justice, the first case in Portugal</u>

Quinta da Rocha is an emblematic area of the central peninsula of Laguna de Alvor dominated by important natural values and protected by national and international environmental laws (Natura 2000 Network), i.e. *Thymus camphoratus* (Law 316/89; Habitat Directive 92/43/CEE, Law 140/99); *Linaria algarviana* (endemic specie). In 2006, a private company that owns Quinta da Rocha initiated the first works of saltmarsh destruction, leading to several accusations and administrative offences regarding the attempts to protect species and habitats. Several environmental ONGs gathered in defence of theses natural values, and successfully moved a court order to force the owner to fully replace the habitats lost, as a consequence of fields ploughing and vegetation uproot. These profound changes in Quinta da Rocha in terms of landscape and habitat destruction provided the basis for a restoration plan, which the owner is forced to submit to the National Conservation Institute (ICNF), besides paying a fine for environmental damages. Despite this kind of sentences being framed by the national land planning tools, rarely is environmental law really enforced. The group of ONGs that accompanied this case (Grupo de Acompanhamento da Laguna de Alvor) highlighted recovery measures focusing on the ecosystem´s resilience and recovery capacity, using as few (physical and economic) resources as possible. This marked a new beginning for those reclaimed saltmarshes towards recovery, whose assessment is part of the further work to be developed.

7.4 Managing saltmarshes: policy orientation and DPSIR

A careful analysis of the Portuguese tools that act in the coastal zone (POOC – Planos de Ordenamento da Orla Costeira) identifies a disregard for inland waters and inter-tidal ecosystems, such as saltmarshes — the protection and assessment of the POOCs are mostly aware of beaches, management issues focusing 500 meters of potential uses (natural, social, cultural, economic), and coordination and harmonization in the scope of other land management and water planning tools (APA, 2016).

Therefore, this section describes the application of a tool to assist in the identification of the pressures and the way they influence their state, as well as contribute to determine the habitat value and build targeted responses. The DPSIR methodology has the capacity to assist management, particularly habitat restoration, re-creation or creation (Doody, 2008). The focus of the DPSIR framework (Drives, Pressures, States, Impacts, and Responses) is based on the interconnection between natural systems, designed systems, and social systems (Atkin et al., 2011). This framework has been widely used for the purposes of assessing marine and coastal ecosystems (Rekacewicz, 2005; Curtin & Prellezo, 2010; Cooper, 2013; Kelble et al., 2013).

The application of the DPSIR framework is frequently used to assist decision-making in many steps of the decision process; it takes the essential features of a system, describing its composition, complexity and variety of internal and external relations. This constitutes the Response part of the DPSIR framework in which Drivers (human demands on the systems) and Pressures (the precise activities leading to change) result in State Changes (in the natural features) and Impacts on the socio-economic uses of the systems (McLusky & Elliott, 2004). The DPSIR framework requires a response in order to reduce, mitigate and/or compensate adverse effects; each driver and corresponding indicators suggest management options (EEA, 1999; Duarte et al., 2006).

Table 18 shows the Drives, Pressures, States, Impacts and Responses (DPSIR) for the saltmarshes of the study area based on the work developed in the previous chapters.

Table 18. DPSIR for Arade and Alvor saltmarshes

Driver	Pressures	State	Impact	Responses
Saltmarsh reclamation	Erosion and saltmarsh retreat Agricultural dykedland	Disappearance of great saltmarsh patches Changes in saltmarsh area	Reduction of natural saltmarsh area Vegetation change and biodiversity loss Coastal erosion	Implementing a policy of saltmarsh valuing and conservation – EU environmental policy through Natura Networks 2000
Saltmarsh abandonment	Saltmarsh recovery and emerging new typologies	TRS and EMM with vegetation and sediment differences	Changes in ecosystem service delivery Changes in permanent stages of vegetation and morphological succession	Implementing a plan for saltmarsh management Monitoring passive recovery of former reclaimed saltmarshes based on typologies
Land development	Urban and touristic development Recreational boat circulating	Increase of residential tourism Great touristic complex constructed and projected Golf courses	Continuous saltmarsh reclamations Water and land pollution/discharges Eutrophication	Sustainable tourism and navigation Environmental awareness of ecosystem services and conservation

Driver	Pressures	State	Impact	Responses
Changes in the first sector activities	Degradation of saltpan infrastructure Abandonment of agricultural fields in the marsh surroundings	Brackish water accumulation Investment in aquaculture Physical limitations for saltmarsh growth	Degradation of water quality Nutrient enrichment Changes in sediment availability	Implementing managed realignment for tidal re-activation Promoting supervised de-embankment Regulating and limiting aquaculture activities in areas of high natural value
Environmental change	Sea level rise Coastal flooding Marsh migration	Wave erosion Coastal squeeze Changes in species zonation	Decrease in coastal defence capacity of saltmarshes Erosion of the marsh scarp	Managed realignment to allow landward saltmarsh migration Restoring the coastal line and promoting passive recovery Creating elevation changes for pioneer vegetation to colonize

Combined pressures and drivers have culminated in a specific state and associated impacts, which together point out responses to management. Saltmarshes are worth managing regardless of the potential doubts about their ownership or primary natural state. Many services are provided by saltmarsh ecosystems, i.e. climate regulation, filter pollutants, barrier to the spread of pests (Spencer & Harvey, 2012a), sinks for flood waters, biodiversity preservation, carbon sequestration and burial (Caçador et al., 2007; Reboreda et al., 2008; Mattheus et al., 2010), biogeochemical cycle (English et al., 2009), bird watching, and water sports play an important role (Currin et al., 2008); vegetal components are used by cosmetics and pharmacy (Gedan et al., 2009); the pioneer species *Spartina maritima* is an excellent source of food for cattle (Loureiro, 1904; Vieira, 1911) and provides an extensive grazing area (Milotić et al., 2010); species of the family of Salicornia nourish the pickle industry and are incorporated into the daily cuisine (Gedan et al., 2009).

Nevertheless, most works point out the capacity of saltmarshes to cope with rising sea levels (Simas et al., 2001; Gedan & Bertness, 2010; Kim & Bartholdy 2011a), and concerning the response capacity in the cases of coastal flooding or tsunamis (Möller et al., 2001). The unique characteristics of saltmarshes, such as their extension and geographical position on the coastline, combined with vegetated tussocks with a certain level of maturity, allow the reduction of the wave energy projected through the saltmarsh, diminishing the intensity of possible impacts for human communities (Möller & Spencer, 2002; Möller, 2006). Among these many potential ecosystem services, coastal protection in a context of environmental change is a specific service on which management should focus targeted measures in order to restore and return endangered and lost saltmarshes.

7.5 Recommendations for Alvor and Arade saltmarhes

The policy guidelines focusing on estuarine and lagoon ecosystem restoration and watershed management arise from the European policy in the scope of the Water Framework Directive (Directive 2000/60/EC), which is, according to Plater & Kirby (2006), one of the most ambitious parts of environmental legislation at the European level. This piece of legislation addresses saltmarshes as part of an eco-hydrological tool, and applies mainly to river and estuarine management. The EU habitat directive recommends the creation of a new saltmarsh wherever a natural saltmarsh is lost to coastal development or erosion caused by sea-level rise. A set of biological characteristics of a certain level of equivalence to the natural condition are required. In addition, the creation of perimarine wetlands enhances saltmarsh response capacity to the rising sea level and consequently ecological restoration achieves the objectives defined by the Water Framework Directive. Under the coastal zone EU policy, the ICZM was implemented in 1981 for the purposes of coastal sustainability:

> Integrated coastal management aims for the coordinated application of the different policies affecting the coastal zone and related to activities such as nature protection, aquaculture, fisheries, agriculture, industry, off shore wind energy, shipping, tourism, development of infrastructure and mitigation and adaptation to climate change. It will contribute to sustainable development of coastal zones by the application of an approach that respects the limits of natural resources and ecosystems, the so-called 'ecosystem based approach' (http://ec.europa.eu/environment/iczm/index_en.htm).

The concept of ICZM gathers all the phases of collecting information, planning, decision-making, management, and monitoring, working with the stakeholders as a crucial part of the whole process. By 2013, the Commission adopted a Directive establishing a framework for maritime spatial planning and integrated coastal management, although it focuses mainly on shared seas and maritime activities.

In fact, Portuguese saltmarshes have been given less importance in the national planning context, although efforts have been made towards gathering the basis for an integrated coastal zone management at the national level, mainly in what regards estuaries and lagunal systems (GIZC, 2007). Despite a great part of the Portuguese saltmarshes are simultaneously protected by European environmental policy (Natura 2000 Network) and the national environmental laws, areas like Alvor and Arade estuaries have faced centuries of land transformations, and non-institutional care has been taken to prevent the permanent loss of some saltmarshes. However, few measures have actually been taken to counter the trend of saltmarsh shrinkage and fragmentation, and to understand the causes and consequences of this process. Spatial and temporal dynamics and patterns of land-use and land-cover changes have poorly been studied in these areas. Additionally, the full track of saltmarsh natural recovery is also missing from the national framework. The absence of large time series of high-resolution spatial information that could provide insights into changes in land use and cover, vegetation cover, and the evolution of the coastline, is compromising an effective planning.

The work developed in this thesis tries to provide some insights into the broad application of land-use/land-cover changes to understand the evolution of saltmarsh ecosystems, combined with sedimentology and short-term accretion works, as well as a landscape metrics tool to assess a coastal defence index, one of many ecosystem services provided by saltmarshes; and combines all of these sectorial works in the same geographical area (saltmarshes of Laguna de Alvor and Rio Arade) with a phytosociological assessment. Additionally, it provides guidelines to promote saltmarsh passive recovery, using low financial investment and maintenance.

7.5.1. Restoration (active recovery)

There are various typologies of restoration, which differ according to authors, countries, and ecosystems. The distinction relies on the typology of the ecological restoration practice:

7.5.1.1 Primary and compensatory restoration

Focus on science-based metrics, including the notion of returning the resources to their natural (i.e. pre-disturbance) state as much as possible. Nevertheless, quantitative metrics are added to biological parameters with the objective of increasing their economic value (i.e. ecosystems services) (English et al., 2009).

It is mainly based on economic principles (value-based), which means that an economic value is attributed to each ecosystem component that is being targeted for restoration (example of primary restoration in Figure 25). Compensatory measures can ensure the substitution of some lost ecosystem services by others with the same biological function (Elliot, 2007). Managing the two or multiple cause-effect issues is a challenge that can be overcome by compensatory restoration (English et al., 2009), accomplishing a gain in ecosystem services when compared with the ones lost during the disturbance period.

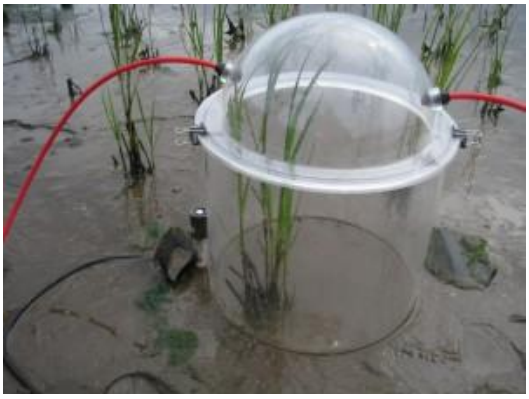


Figure 25. *Bolboschoenus maritimus*var *maritimus* (available at http://synergy.st-andrews.ac.uk/serg/projects/saltmarsh/)

7.5.1.2 Habitat Equivalency Analysis (HEA) vs. Resources Equivalency Analysis (REA)

HEA is a wider application model emerging from the compensatory restoration on saltmarshes — for Strange et al. (2002), this model concentrates efforts on habitat restoration, emphasizing service delivery by the internal functioning of the ecosystem. This relates with the service-to-service model discussed by Elliot (2007). On the other hand, for English et al. (2009) HEA consists in the balanced relationship between the services lost in a damage situation and the economic gains that arise from the restoration (see Figure 26). Resource Equivalency Analysis (REA) is another model based on the equivalence of gains and losses. It is mainly focused on those resources that were damaged or injured during the disturbance period, and on what restoration can do to enhance or regain those resources (English et al., 2009).

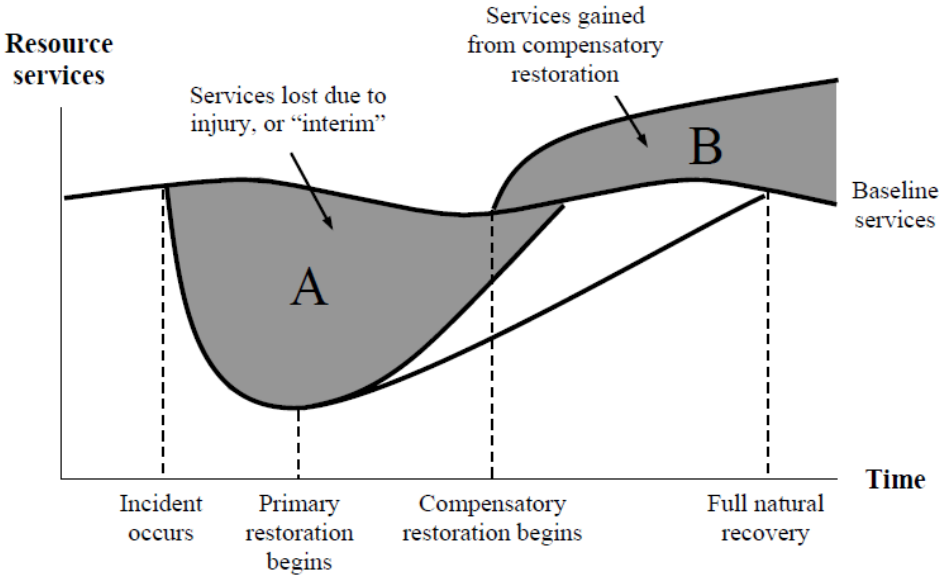


Figure 26. Scheme of the use for compensatory tools (English et al., 2009)

7.5.1.3 Species introduction and/or invasive species removal

One of the most common techniques applied to saltmarshes is using the pioneer vegetation of low marsh communities (*Spartinetum*), which presents a radial growth. Strange et al. (2002) and English et al. (2009) highlight the great efficiency of species introduction because saltmarshes are highly productive (efficiency in ecological terms) and associated costs are considerably low (economic benefits): in a two-year period, planted *Spartinetum* usually registers an increase in height and density by around 95% when compared with natural saltmarshes. In terms of plantation techniques, English et al. (2009) argue *Spartinetum* should be planted in band in order to enhance spontaneous growth, with direct effect on sediment stabilization (preventing initial erosion). Considering subsoil biomass growth and a five-year period of monitoring, there was only 7% of new biomass at 10 to 20 cm depth. Compared with natural saltmarshes, in which bio-chemical processes are accelerated, low biomass in planted low saltmarshes fails short in terms of organic material concentration (English et al., 2009). In this context, the restoration capacity of other saltmarsh species is discussed; mainly those typically from medium and high saltmarshes, particularly in situations of poor maturity and consolidation of low marsh communities (Wolters et al., 2008). Doody (2008) demonstrates the success of managed realignment projects (UK; US) using the hybrid *Spartina anglica* (*Spartina.maritima* x *Spartina.alterniflora*) whose colonization capacity spans all levels of the tidal range, and functions as a sediment stabilizer, therefore making *S. anglica* a good candidate for 'species introduction' projects.

Phragmites australis is commonly used in restoration across the UK due to its rapid growth and productivity (Doody, 2008). The invasive *Spartina versicolor* and *S. densiflora* are also used to achieve rapid growth rates by facilitating the accumulation of sediments and organic material (biomass) and consequently increasing saltmarsh elevation. One disadvantage of rapidly increasing elevation is the narrow range of the tidal flow, undermining the survival of other marsh colonizers (Windham & Lathrop, 1999; Rooth & Stevenson, 2000). Restoration of intertidal wetlands may find long-term success through the eradication of invasive species of the marsh spectrum (i.e. *Spartina densiflora*) that colonize Southern European marshes (Mateos-Naranjo et al., 2008) and other invasive species, typical of disturbed marshes, and occurring in both study areas. Despite being more evident in the natural saltmarshes of Alvor (back barrier marsh, sandy soils, and target of compensatory restoration at the beginning of the year 2000), the most dominant invasive species are *Cotula coronopifolia*, *Carpobrotus edulis*, and *Oxalis pes-caprae*. The Irradiation of these invasive species could be a complement to restoration projects.

Figure 27. *Bolboschoenus maritimus* var. m*aritimus* in Boina creak (by the author).

7.5.1.4 Re-creation

Re-creation is related to saltmarsh enhancement, and has been applied in the US for decades. Incentives to the creation of wetlands are present in the national policy, mainly saltmarsh habitats (Crépin, 2005; Wolters et al., 2008). Figure 28 demonstrates the works undertaken by recreational practices, using sediment recharge offshore to create a significant platform of mudflats to assure the protection of the created saltmarsh. Saltmarsh re-creation refers to the use of normalized scores to evaluate the injury of sub-habitat types of a marsh complex — the main goal is biotic production, regaining ecosystem services through recreation of injured sub-habitat types (English et al., 2009).

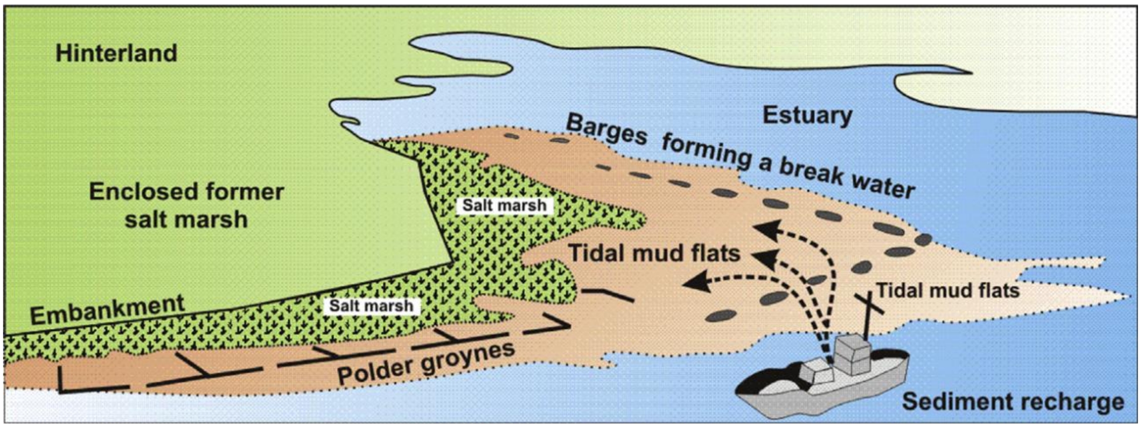


Figure 28. Sediment recharges to tidal habitat re-creation (Doody, 2013). Enables saltmarsh to develop in the border line of the embankment; tidal mud flats would create a natural barrier to break water in case of coastal flooding.

7.5.1.5 Managed realignment (MR)

MR incorporates a set of actions to compensate for the loss of mudflat and saltmarsh habitat associated with multiple land developments (urban construction, tourism growth, port installation, and others) (Chapman & Underwood, 2011; Morris, 2013; Esteves, 2014). It aims to increase flood water storage and serve as wave attenuation (Möller, 2006; Friess et al., 2008); to provide environmental benefits through the re-creation of natural habitats, including landward movement, or to set-back the defence line (Mazik et al.; 2010, Esteves, 2014). Doody (2013) adds that MR interventions allow shorelines to move backwards or forwards, using controlled criteria (see Figure 29). Although the concept of MR and what it represents has evolved with time, it has also assumed regional particularities (Esteves, 2014). The threshold is frequently the loss of intertidal areas and the coastal squeeze, in which saltmarshes play an important role as buffers and impact absorption areas (Esteves, 2013). Managed realignment is one of the most used and active interventions on coastal wetlands across Europe (broader application in the UK). It focuses on baseline re-establishments and is strongly associated with the consequences of rising sea levels for coastal ecosystems. Additionally, MR deals with the effects of climate change, such as coastal flooding, and sea storms, among others.

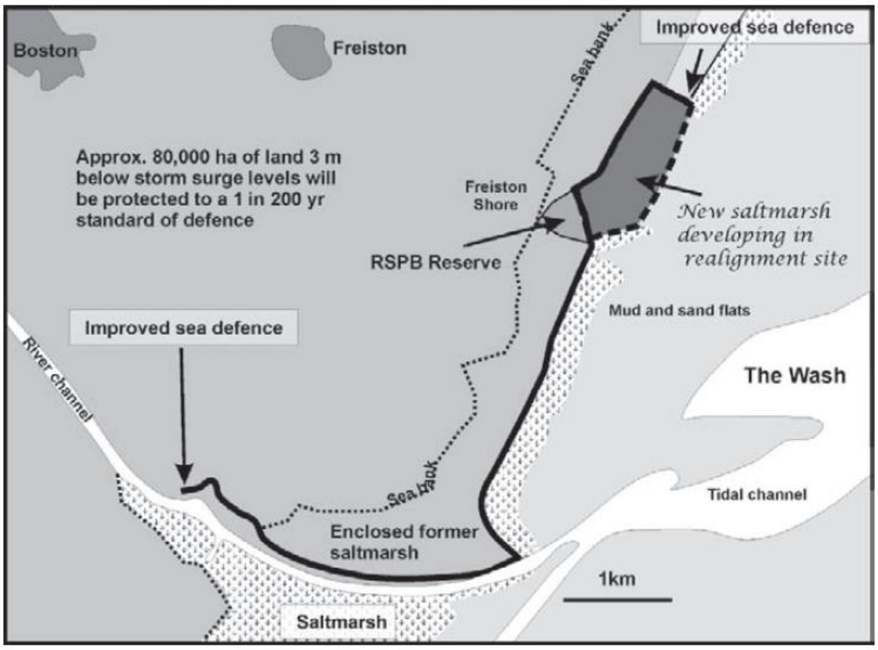


Figure 29. Seawalls realignment with improvements of sea defence through saltmarsh developing. An enclosed marsh is between sea breaks, the new sea defence and the new "natural" marsh development, aligned with existing saltmarsh**.**

7.5.2. Recovery (passive recovery)

The contribution of international action lines is vital for the establishment of a viable land-use planning tool that enables the understanding of the saltmarsh spatial-temporal dynamics in order to diagnose its condition (erosion, accretion, or stability) and intervene occasionally in damaged areas to stimulate self-recovery.

This recovery would probably be a slow transformation from a reclaimed saltmarsh by directly or indirectly installing tidal dynamics, which would boost the hydro-geomorphology system and enhance the installation of pioneer vegetation, thus creating different patterns. On the other hand, international studies point to more targeted measures, such as compensatory ecological restoration, or even actions incorporating managed realignment. Beyond these methodologies to approach saltmarsh ecosystem management, the objectives are to strengthen the coastal defence capabilities of saltmarshes to cope with sea level rise and protect human settlements and activities in cases of environmental change (ocean storms or coastal flooding).

Compensatory measures may ensure the substitution of lost ecosystem services by others with the same biological function, which is why understanding and diagnosing the ecological connectivity and evaluate resilience is a key aspect to make decisions based on the ecosystem. The concept of passive recovery has been understood as the process from which ecosystems — mainly saltmarsh ecosystems — are able to self-improve, overpass, and supplant a set of marker characteristics (vegetation, granulometry, shape, morphology…) through an inner process. It derives from the understanding of recovery as a non-artificial process, privileging little human intervention, using as few resources as possible, accounting for environmental impacts and economic motivations. It is an anthropogenic-free intervention, which means that the ecosystem structure and functioning recovers passively by removing the stressor, whether it is a natural or an anthropic stressor. This includes for example, a self-cleaning process after an oil spill (Duarte et al., 2013), or wave damage in an abandoned dykedland saltmarsh, allowing tidal dynamics to settle and saltmarsh pioneer species to re-install (Almeida et al., 2014).

Figure 30. a) TRS in Boina in former dykedland b) TRS in Alvor near former saltpans

7.5.2.1 Tidally restored saltmarshes

<u>Tidal Reactivation</u>

Tidal reactivation may assume various forms, depending on the marsh morphology (back barrier, estuarine fringing marshes, or fluvial marshes) — it includes poldering and dyke breaching.

Poldering

- Principles of application: frequently used in saltmarshes derived from the historical approaches to marsh reclamation or embankment for use as agricultural land (Doody, 2008). It aims to stimulate the colonization of the mudflat by pioneer vegetation.
- Technique: parallel lines designed to create a scheme of channels that allow the tide to slowly spread and provide inundation to former agricultural land (see Figure 31).

- Advantages: sedimentation is increased, providing both protection from erosion and coastal flooding by creating an extensive flat area where marsh vegetation develops (see Chapter 3 for poldering structures and Chapters V and VI for vegetation details). It also allows the protection of landward mature saltmarshes (Figure 31).
- Disadvantages: it is an ancient and very labour-demanding technique, whose results are not always successful (Doody, 2008). Grazing starts to be a frequent activity in these areas, limiting the full development of saltmarsh vegetation, which tends to be short, and species-poor.
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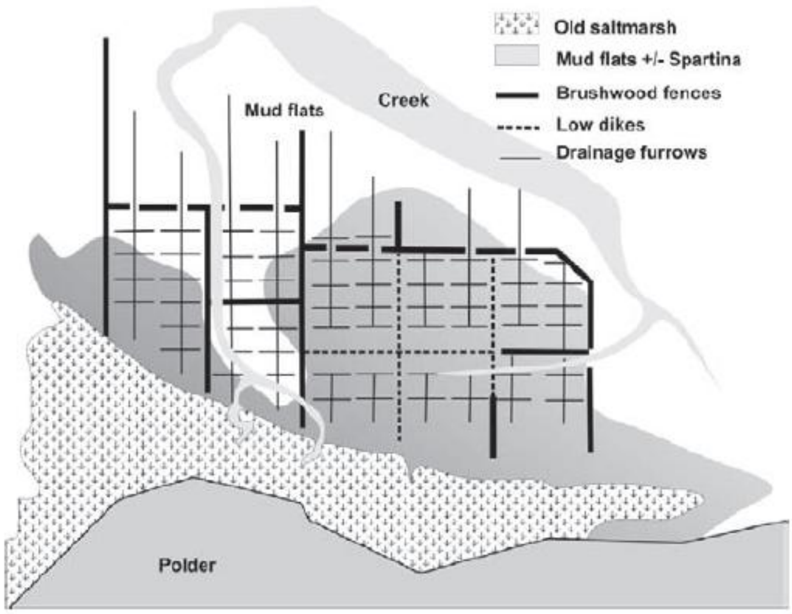


Figure 31. The complex scheme of intermittent dikes that help sediment circulation and enhance saltmarsh development through tidal reactivation (Doody, 2008). It protects the mature saltmarsh and the enclosed polder (Netherlands).

Dyke breaching

- Principles of application: the abandonment of old dyke structures that collapse unintentionally falls in the scope of this technique (TRS of Arade and Alvor); as well as premeditated measures of hydrological restoration. The aim is habitat restoration.
- Technique: part or the whole defence structure is maintained and a circuit of culverts and sluices work together to promote tidal influence to spread along the saltmarsh — Regulated Tidal Exchange (RTE) and Controlled Reduced Tide (CRT) (Esteves, 2014); in the cases where the dykes breached naturally, the tide entrance is made by single or

multiple breaches, in which natural channelling may conduct the tide through the whole former dykedland.

- Advantages: with the shoreline moving landwards, space is created for an intertidal habitat to generate, resorting to controlled tidal exchanges in a semi-sheltered condition; within the tide, salinity levels are restored and sediments are supplied, allowing mud banks to be colonized with saltmarsh pioneer vegetation and therefore 'naturally' restore the habitat (see Chapters III and V).
- Disadvantages: low mudflat colonization may be a problem due to differences in elevation between natural saltmarshes and former dykedland or embanked for cultivation purposes. The lowest accretion rates were measured within TRS (see Chapter III) due to the reduced and controlled tide. In the cases of Boina creek, TRS are above mean sea level and high water (Figure 32, situation (c)).

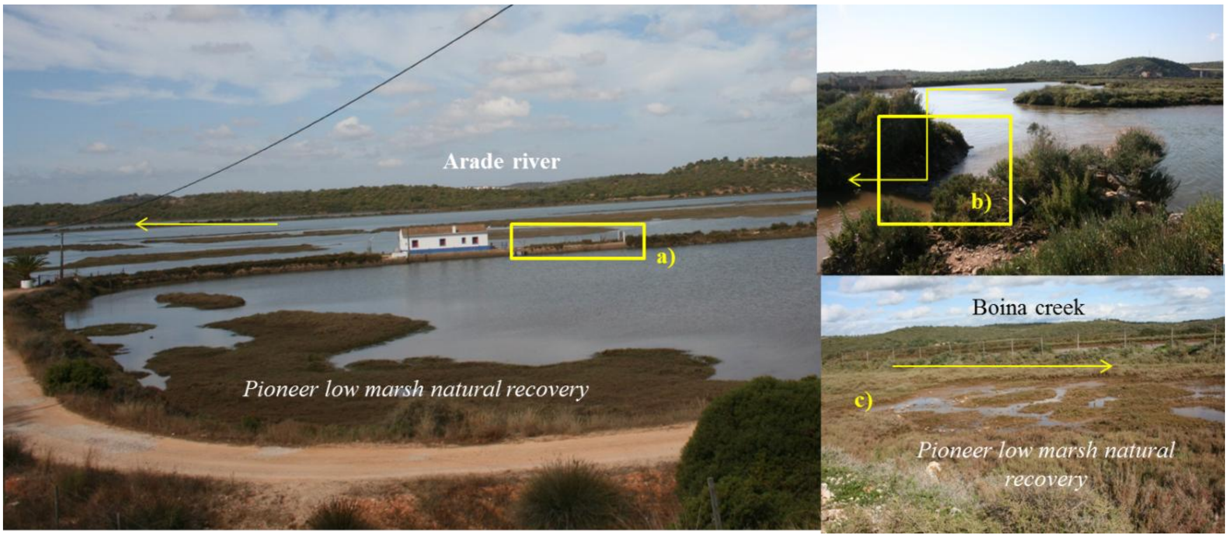


Figure 32 Arade´s and Boina´s examples of Tidally Restored Saltmarshes (TRS); arrows indicate the tidal flow; a) artificial culvert (water mill) enables the tidal influence inland TRS, showing low marsh (*Spartina maritima* and *Sarcocornina perennis* ssp.*perennis*); b) linked with the TRS (a), there is a set of former dykedlands where dykes breached naturally (abandonment, obsolete structures) and where the tide enters in its cycles and goes from one TRS to another; c) TRS in Boina creek are elevated and the tide enters through smaller cracks.

Marsh creation

- Principles of application: it relies on mudflat planting using low marsh pioneer species, such as *Spartina sp*. with the aim of increasing sediment stability due to the binding effect of the roots (see Chapter V).
- Technique: plantation or reseeding with marsh vegetation increases sediment stability due to the binding effects of the roots, increasing shear strength and decreasing erodibility (Esteves, 2014).
- Advantages: this measure reinforces the role of saltmarshes as coastal protectors by absorbing wave energy, proving to be a cost-effective coastal protection policy. Marshes also provide cost-effective protection against flooding by absorbing wave energy. Former saltpans and other obsolete structures could be useful for the creation of new marshes, providing available area for marsh regeneration through tide entrance and seeding. Species in the local species pool would colonize the new marsh with the provided conditions of salinity and hydrodynamics. In a sheltered environment, low pioneer saltmarsh species would have the opportunity to grow, creating a break water barrier (especially in the case of Alvor).
- Disadvantages: marsh plantation may be at risk in severe cases of erosion (see Figure 33) or rising sea levels (UK), due to the insufficient accumulation of fine sediments necessary for marsh creation, and consequently can lead to increased marsh scarp and collapsing (EU, 2004; Doody, 2008; Esteves, 2014).

According to Doody (2008:94), the ‘ideal’ conditions for reseeding or plantation are the following:

- ✓ wind and waves sheltered areas;
- ✓ restricted tidal flow;
- ✓ located between mean sea level and high water;
- ✓ an average slope from 3 to 5%;
- ✓ accretion rates of 3 to 10 mm/y;
- ✓ silty or clayey grain size, preferably firm and oxygenated;
- ✓ lower elevations.

Figure 33. Arade natural saltmarsh scarp (above); TRS erosion process of the marsh baseline (below).

7.5.2.2 Enclosed mix marsh

Alvor is richer in cases of EMM due to a more evident and long-term effective use of agricultural land (see Chapter III). The natural conditions of this coastal lagoon (see Introduction) and the existence of four creeks with alluvial plains that provide favourable land for agriculture, were combined with extensive reclamation of estuarine fluvial saltmarshes (Chapters II and III). Reclaimed marshes were cultivated and abandoned later; hence the vegetation emerging from these areas is a mix pattern of fresh and brackish communities (Chapter VI). Soft de-embankment and realignment are the two most adequate options that combine passive recovery principles with targeted restoration.

Soft de-embankment

- Principles of application: it creates, maintains or enhances fresh water accumulation (rain), and ensures the slow feeding of salt intrusion to the upper lands of the EMM. It aims to maintain the secondary marsh communities that established in the EMM. This could be applied in extremely dry years or in case of excessive runoff.
- Technique: it is based on avoiding excessive sediment accumulation in the ditch and managing natural dykes to prevent landslides and debris runoff.
- Advantages: the ditch plays a crucial role in the whole EMM environment and different species maintenance because they are organized according to salt and inundation tolerance. Soft de-embankment would be applied to EMM with detected strangling of brackish water, providing isolated interventions to re-establish the ideal balance of fresh water and salt intrusion.
- Disadvantages: it requires monitoring and financial investment; in some places, the access to machinery may endanger other transitional habitats; the dominance of coarser sediments (in Alvor, former agricultural land and sand dispersal were found) may facilitate the obstruction of the ditch (see Figure 34); lack of planning for soft de-embankment may lead to a ‘tidal reactivation’ measure; and the disregard for the protection of the existing mix marsh may jeopardize the whole ecological restoration.

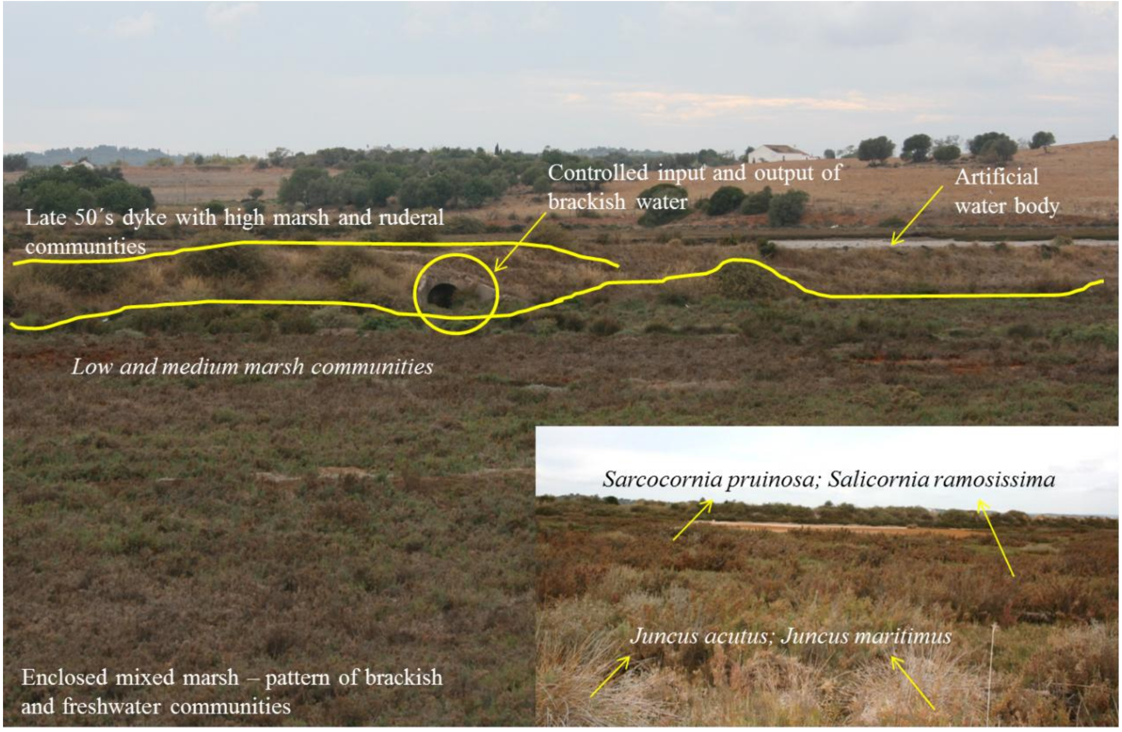


Figure 34. Example of an EMM with a controlled input/output that is obstructed and prevents water circulation; brackish water arises by capillarity and salts accumulate in the roots, allowing low and medium marsh communities to develop in the lowest elevations (accumulation), and at higher elevations, marsh succession presents typical transitional communities, such as *Juncus acutus* and *Juncus maritimus*, representing the presence of fresh water.

<u>Realignment</u>

- Principles of application: realignment *per se* means the deliberate attempt to create a saltmarsh habitat inside an embankment (Doody, 2008). It aims to increase flood water storage and serve as wave attenuation focusing on compensation for loss, habitat creation, and erosion prevention (Figure 35).
- Technique: one of the most widely used interventions on coastal wetlands across Europe associated with multiple land developments (urban construction, tourism growth, port installation, and others): landward movement, or to set-back the defence line.
- Advantages: it is associated with sea level rise consequences to coastal ecosystems and additionally with the effects of climate changes (e.g. coastal flooding, sea storms), and it aims to provide environmental benefits through the re-creation of natural habitats, including ecosystem service enhancement or reboot (see Chapter IV).

- Disadvantages: potential loss of the pre-established plant communities; possible dissimilarities with the local species pool (see Chapter V); works may imply changes in elevation.

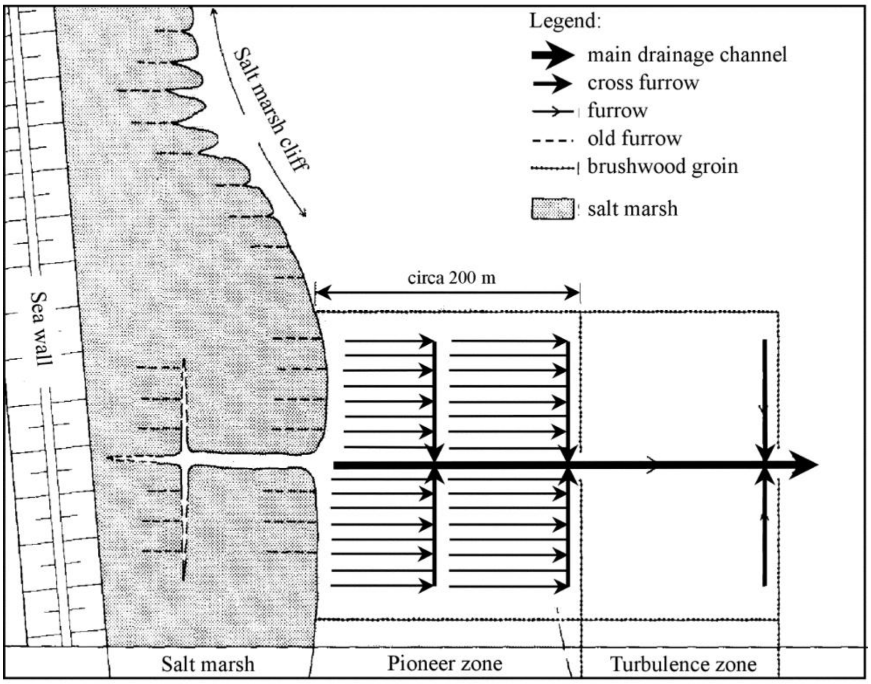
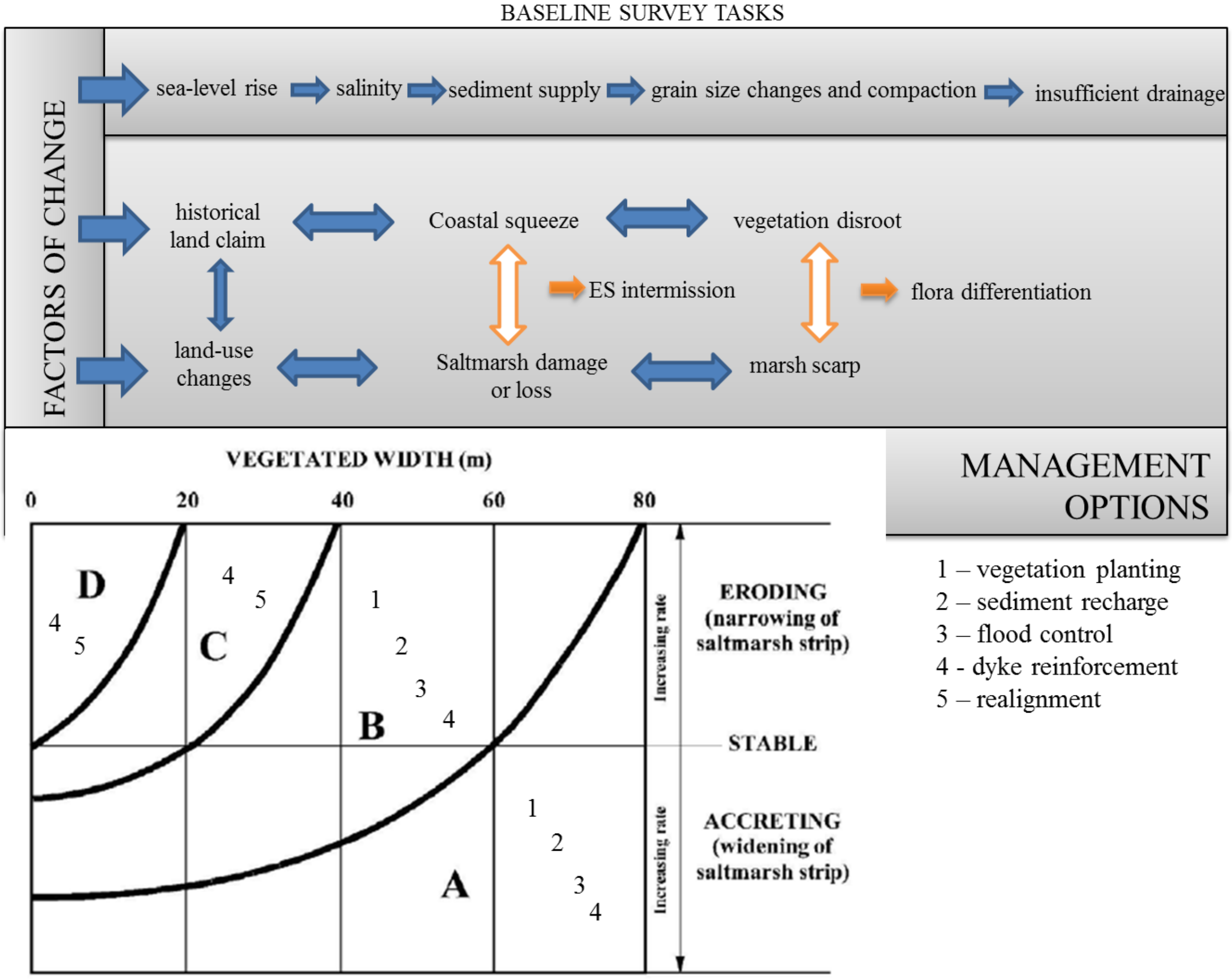


Figure 35. Example of saltmarsh management using the echelon system (Hofstede, 2003).

7.5.3 Recommendations towards habitat management and ecological restoration

The argument for managing coastal saltmarshes through restoration or re-creation lies in the eminent risk of habitat loss at the European scale (Atlantic and Mediterranean saltmarshes). In the cases of Alvor and Arade, the estuaries are small (< 50 km2) and therefore management practices might not always be transferable to them (Callaway et al., 2014). Large estuaries, such as Tagus, Sado, or even *Ria* de Aveiro and *Ria* Formosa (also coastal laggons), have higher socio-economic importance. Additionally, potential higher impacts are expected to these areas, implying wider policies and management options. Due to their size, internal and external processes may have a consequence on the overall estuary environment, rather than injuring a limited area (which happens in larger estuaries). The same applies to disturbances and changes which may affect the morphology and the ecology of the small estuaries as a whole (Callaway et al., 2014)

The extent of saltmarsh accumulated losses over the past decades — and some of them recently enhanced by the various factors of environmental change (i.e. coastal flooding, sea storms, sea level rise, alien species colonization) — implies a status of ‘natural conservation’. In an eroding marsh, natural changes within the ecosystem will arise regardless of the boundaries defined by the protected status. Part of the habitat management problem is that statutorily designated areas (protected areas, nature reserves, among others) have boundaries that neglect their ecological framing, putting into check any efforts to apply a full restoration work towards nature conservation. Therefore, considerations about scale, habitats, and pre-state conditions are mandatory for an accurate management.



A – no immediate risk to sea defences; monitor only;
B – investigate causes of change; consider implementing low cost maintenance and enhancement solutions, increase monitoring efforts;
C – urgent need for study and probably intervening actions;
D – may not be possible to re-establish/maintain saltmarsh without major works or realignment.

Figure 36. Scheme for adopting saltmarsh management for ecological recovery (adapted from (DEFRA, 2008).

Understanding the factors that motivated the changes is essential to decide on the management options to be undertaken. A schematic framework highlights the main driving forces for change identified within Alvor and Arade saltmarsh habitats. Additionally, stressors and the most important consequences are addressed — coastal squeeze (vegetation uproot), saltmarsh damage or loss (marsh scarp), affecting ecosystem service delivery, and creating high floristic dissimilarities.

The natural saltmarshes of Arade are between situations (B) and (C). This difference is supported by vegetation width and stable saltmarsh communities in some places, although in other segments erosion starts to be a threat. In the case of Boina, its sheltered condition and increased sedimentation process place it between situations (A) and (B), mainly due to very well stabilized saltmarsh communities and a minor influence from wave and wind energy. Considering the back barrier and estuarine fringing natural saltmarsh of Alvor, they clearly are positioned in (A) due to the remarkable accretion rates, influenced by sand transportation upward the Alvor´s estuary. Those natural saltmarshes of open embayment in the Alvor were not monitored in terms of accretion, but the studies of land-use change and vegetation surveys revealed that they are positioned between (D) and (C). Management options may include vegetation planting in eroding sections, followed by sediment recharge and flood control.

The tidally restored saltmarshes of the study area are in stabilized and/or expansion situations, falling into (A) and (B): vegetation is widening and occupying the sediment accumulation zones (bordering dykes) and developing *Spartina maritima* banks (which is mostly the case of Arade). Managing these areas requires a compromise between flood control and dyke reinforcement.

With regards to enclosed mix marshes (mostly those in Alvor), accretion rates are the highest of the study area; however, the vegetation pattern developing within these areas is a mix of brackish, freshwater and ruderal species, allowing the entrance of invasive species due to a large anthropic use associated with these enclosed areas. Habitat management depends on the established plant communities, and should consider implementing low cost maintenance and enhancement solutions, as well as assure monitoring.

The concept of passive recovery that resulted from this work is tied with the understanding of recovery as a non-artificial process, independent of anthropic interventions as much as possible. This means that it is a process in which ecosystem structure and functioning have recovered by the removal of the stressor(s), whether these were natural or anthropogenic. This includes ‘self’ ecosystem-based processes, i.e. self-cleaning, pioneer species recovery, adaptation and resilience of saltmarsh communities; but also soft ecological recovery engineering works, such as tidal reactivation, marsh creation, soft de-embankment, or realignment. Ecosystem service delivery and climate change adaptation/mitigation should be in the background of habitat management and ecological recovery.

7.6 Final remarks and further work to be developed

The main objective of this research was to study the processes and dynamics underlying the Alvor and Arade saltmarshes´ evolution, and ultimately to contribute to the knowledge of habitat management and ecological restoration. This main objective was achieved by studying the saltmarsh typologies which emerged from the land use changes in both study areas – the floristic structure, the grain size analysis and the ecosystem services assessment support the idea of tidally restored saltmarshes and enclosed mix marshes be considered cases of involuntary managed realignment, being simultaneously the background for passive recovery "options". Considering the floristic surveys and the similarity indexes developed in the Chapter V, is desirable to implement volunteer measures of MR to other Portuguese saltmarsh that fulfil the same characteristics of Alvor and Arade, as well as adopt the framework (presented in this section) to monitor natural saltmarshes, towards erosion control and implement soft passive recovery. Habitat management must be implemented in the consolidated TRS and EMM to address the principles of an ecological recovery, which should be a milestone for the saltmarsh policy.

This research placed the hypothesis in *the possibility of the abandonment resulting from the evolution of former reclaimed saltmarshes, allowing the emergence of saltmarsh typologies. These typologies are the reflex of different evolution patterns, as well as the way tide influenced a passive recovery with regards of flora and vegetation composition.* The hypothesis was confirmed by the worked developed along the process of investigation:

Chapter II	
There are consequences of past anthropic activities in the saltmarsh evolution and composition.	✓ abandonment and lack of management tools have led to multiple patterns of evolution; ✓ land-use changes posed a typological differentiation between saltmarshes based on flora composition — related to Chapter 4;
The main consequence of saltmarsh reclamation is the major decrease of saltmarsh patches and the reduction of the area occupied by natural saltmarshes.	✓ natural saltmarsh area reduced considerably but secondary marshes developed in specific conditions; ✓ variations among secondary marshes relate to tidal influence or the input of freshwater plus salt intrusion — related to Chapter 2; ✓ connection with Chapter 3 – landscape metrics.
Limitations to the research	• difficulties in access to data, mainly historical mapping

Chapter III	
Reclaimed saltmarsh dynamics and evolution is similar to natural saltmarshes, but there are differences in floristic composition and grain size.	✓ reclaimed saltmarshes registered differences in grain size related to local specificities and topographic variability of anthropic origin — related to Chapter 1; ✓ Alvor — coarser sediments and few O.M.; ✓ Arade — silty and clayey with more O.M.;
There are differences among accretion and erosion rates, and those differences are related with grain size characteristics, floristic structure, and saltmarsh typology.	✓ methods used to measure accretion delivered similar results, and may depend on the previous knowledge of the location to improve assessments; ✓ natural saltmarshes of Alvor are accreting significantly related to sand marine and aeolian transport, but eroding in Arade; ✓ tidally restored saltmarshes registered very low accretion rates — controlled

	tidal influence.
Limitations to the research	• data collection period for shot-term accretion would ideally be of 2 years, but a PhD research finds some to time limitations on environmental measurements.
Chapter IV	
Landscape metrics can provide information on saltmarsh shape, complexity and connectivity and it can be a tool to assess ecosystem services.	✓ shape, complexity and connectivity are key aspects for ecosystem service delivery; ✓ they can assess ES in a geographical and spatial perspective, based on the combination of different metrics; ✓ statistics provide in-depth knowledge and guide metrics selection;
Former reclaimed saltmarshes are able to provide complementary coastal defence in a context of passive recovery.	✓ coastal defence decreased in natural saltmarshes and is enhanced in secondary marsh typologies; ✓ these findings place a great importance on ES protection and support passive recovery — connected to Chapter 4; ✓ apply or develop landscape metrics to assess a coastal defence index that can be used elsewhere and proved to be a good tool in situations when big data are missing.
Limitations to the research	• It would be useful to test ecosystem services in INVest, but the amount of data requirements couldn´t be fulfil, partially due to time issues, and in some cases due to data inexistence.

Abandoned reclamation areas and passive tidal reactivation are the answer to passive saltmarsh recovery.	✓ secondary marshes have proved to be of great importance in the context of saltmarsh habitats — related to Chapters 1 to 3;
The combination of a Mediterranean climate and soft de-embankments support floristic and sediment recovery in former reclaimed saltmarshes that can evolve to become natural saltmarshes.	✓ involuntary tidal reactivation produced TRS typology, which largely assembles with natural saltmarshes; ✓ floristic similarities between natural saltmarshes and TRS are of circa 90%, while with EMM are of circa 70%; ✓ targeted recoveries of Atlantic marshes placed higher differences in species pool than passive recovery;
The saltmarshes of Alvor and Arade combine several characteristics that make them good candidates for ecological recovery.	✓ this study indicates that given the characteristics of TRS and EMM and the advanced state of passive recovery, it can be replicated as a set of 'best-practices' to inform decision-making processes in the scope of habitat management.
Limitations to the research	• the research would benefit from vegetation studies carried out in Portuguese saltmarshes with artificial recovery; and of a deeper study on species recovery in secondary marshes. Nevertheless, the selected study areas have been the object of very few studies on vegetation composition and dynamics.

Further work to be developed would take under consideration the ecological restoration as a compensatory measure, and try to incorporate it as a response to both the cases of saltmarsh loss or damage, but as a tool for conservation and enhancement of ecosystem services. We want to develop a group of metrics that enables the categorical classification and quantification of ecosystem services gained through managed realignment or other cases of passive recovery. We want to build a reference framework to evaluate the saltmarsh area pattern and the geographical scale at which restoration meets habitat management goals in order to assist environmental planning. Additionally, we would like to carry out a study on some saltmarsh areas that will be, in the near future, object of targeted intervention to compensate for habitat loss.

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APPENDIX A – **Land-use changes, water basins and coastal transformations**

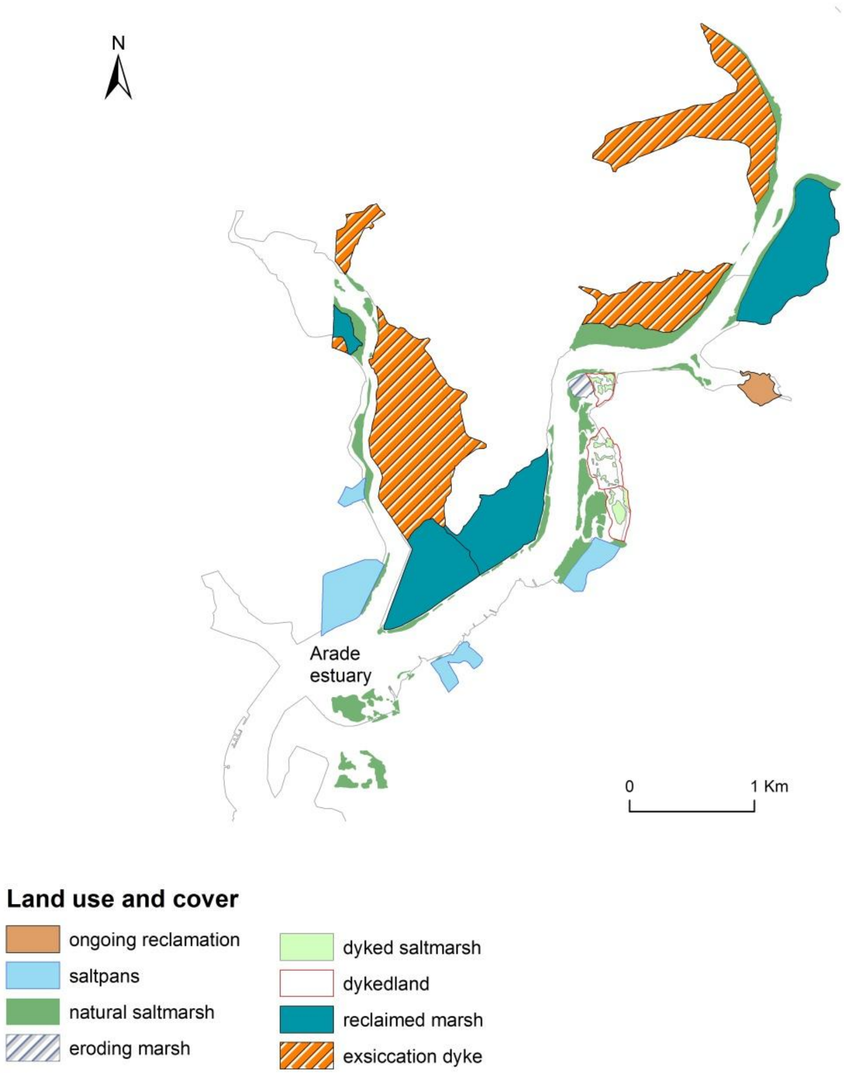


Fig. A1 Land use and cover in Arade and Boina, 1958

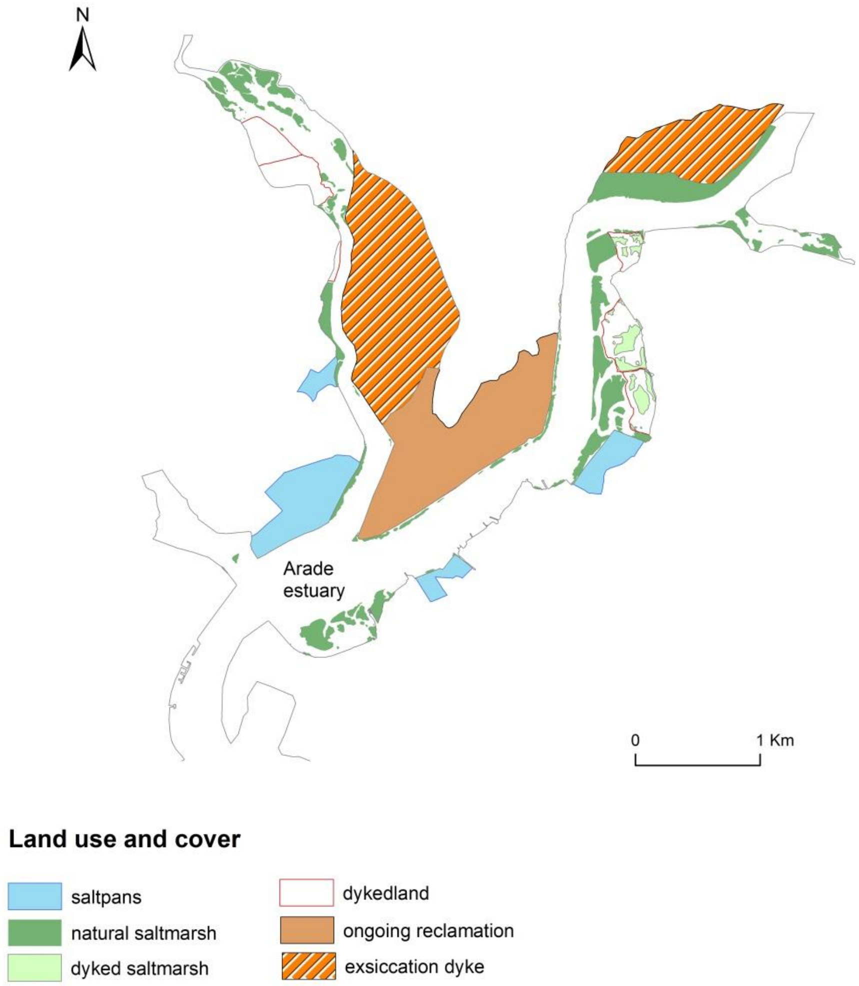


Fig. A2 Land use and cover in Arade and Boina, 1972

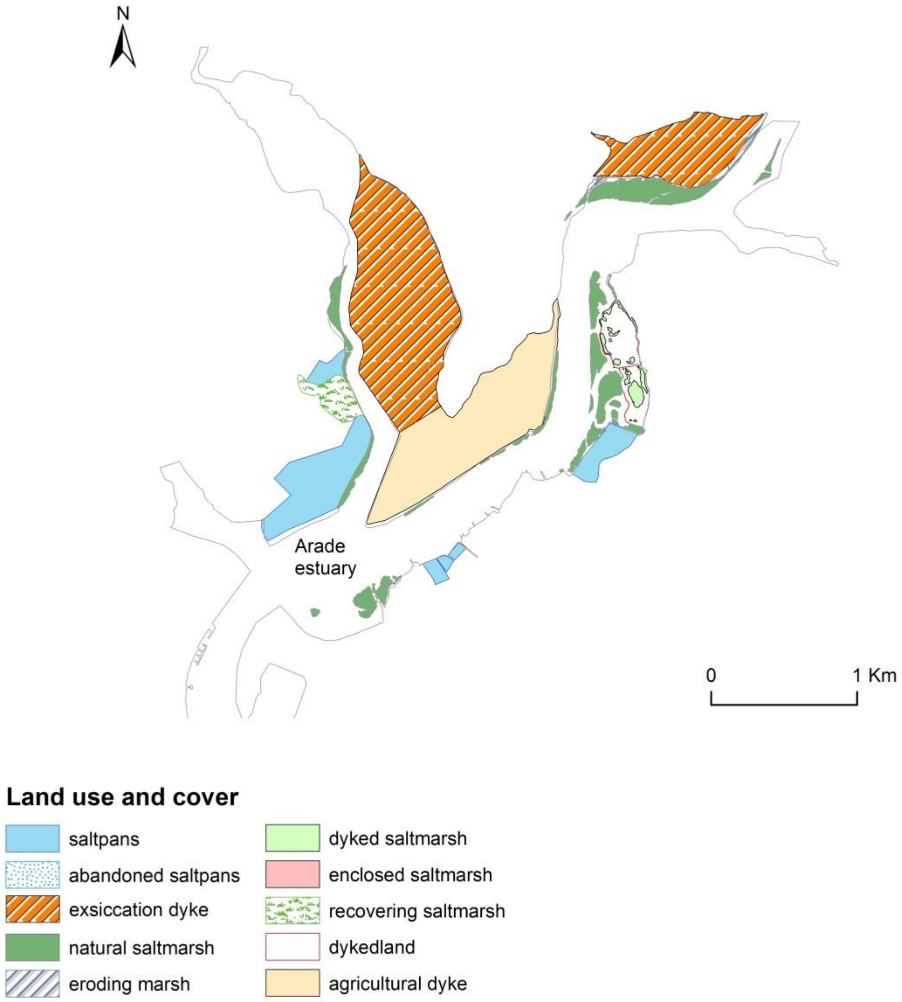


Fig. A3 Land use and cover in Arade and Boina, 1987

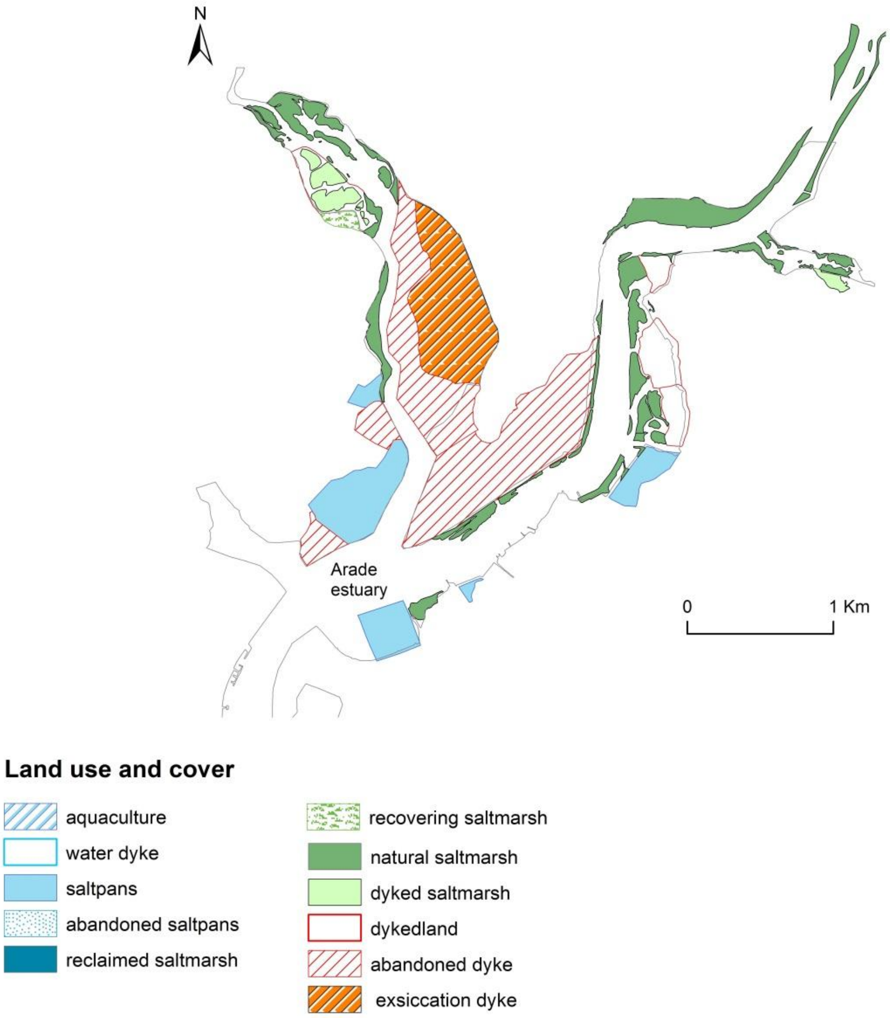


Fig. A4 Land use and cover in Arade and Boina, 1995

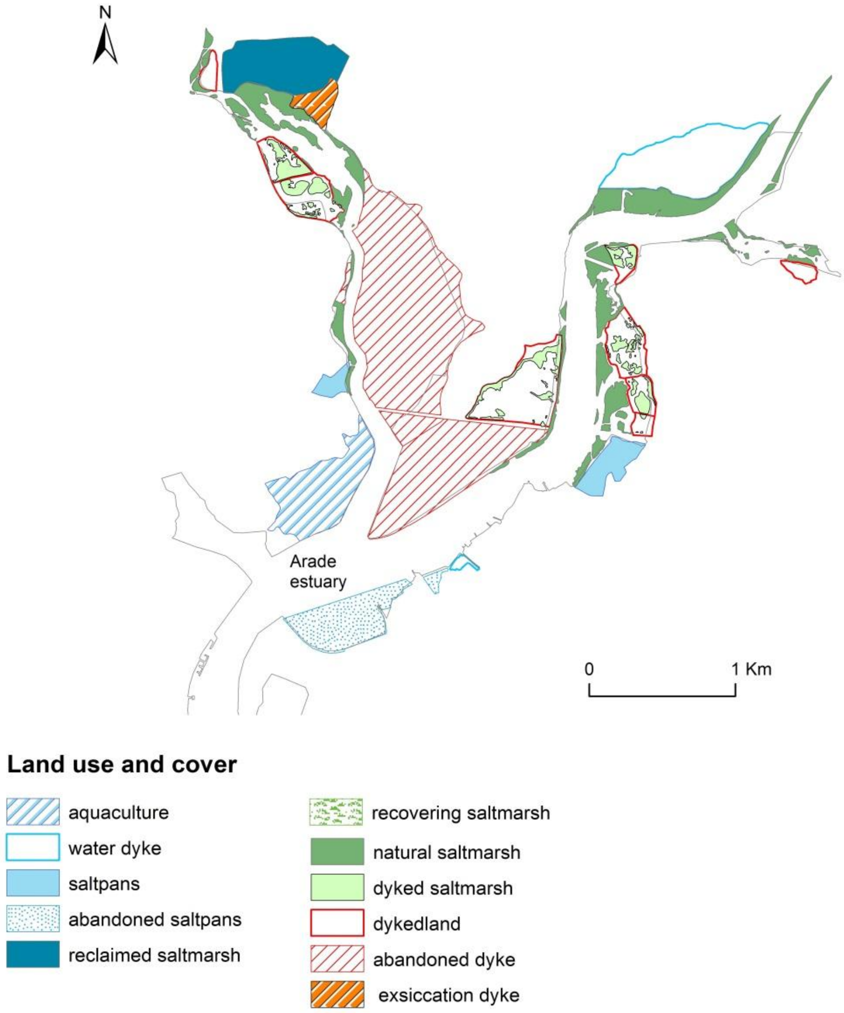


Fig. A5 Land use and cover in Arade and Boina, 2005

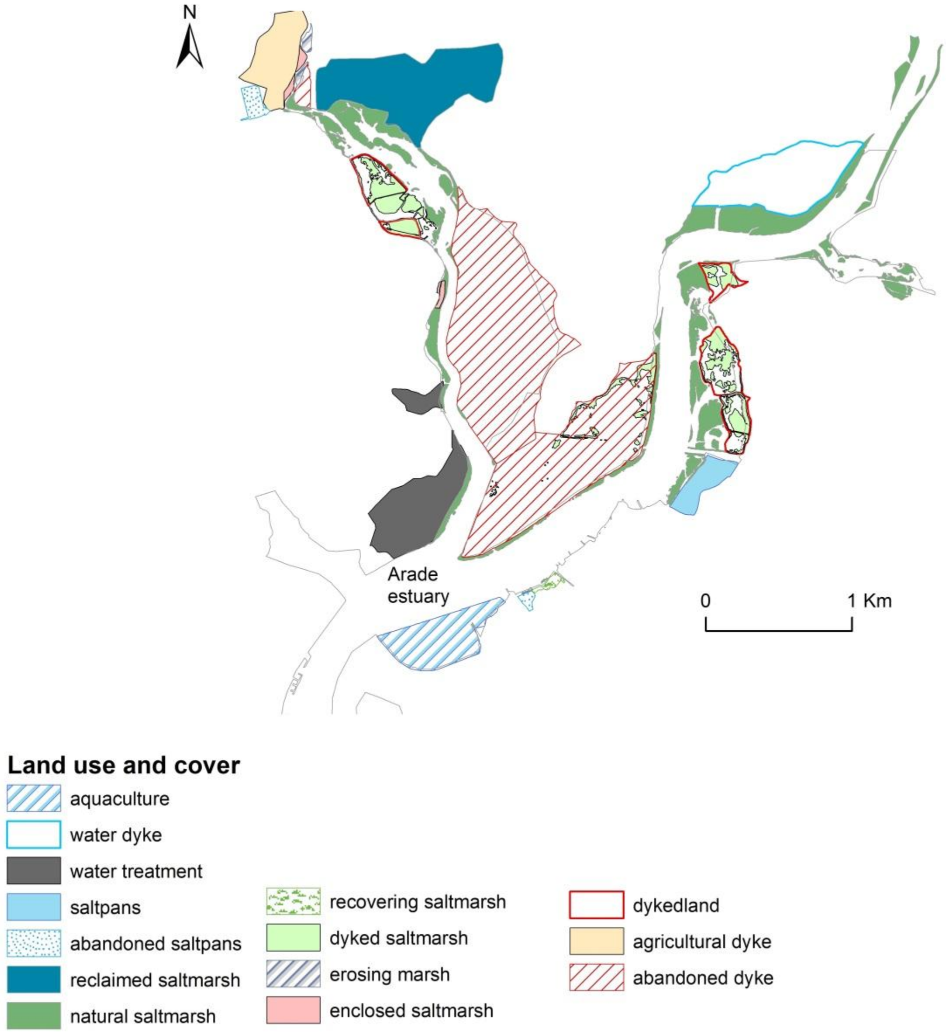


Fig. A6 Land use and cover in Arade and Boina, 2010

The following images are presented as they are published in: Almeida, D., Neto, C.; Costa, J.C. (2014) O processo de reclamação dos sapais da Laguna de Alvor (Portimão). In Formação e Ocupação de Litorais - nas margens do atlântico - Brasil / Portugal, Chapter: X, Publisher: Corbã Editora e Artes Gráficas Ltda, Editors: Silvia Dias Pereira, Joana Gaspar Freitas, Sergio Bergamaschi, Maria Antonieta C. Rodrigues, pp.170-184. ISBN: 978-85-98460-20-8

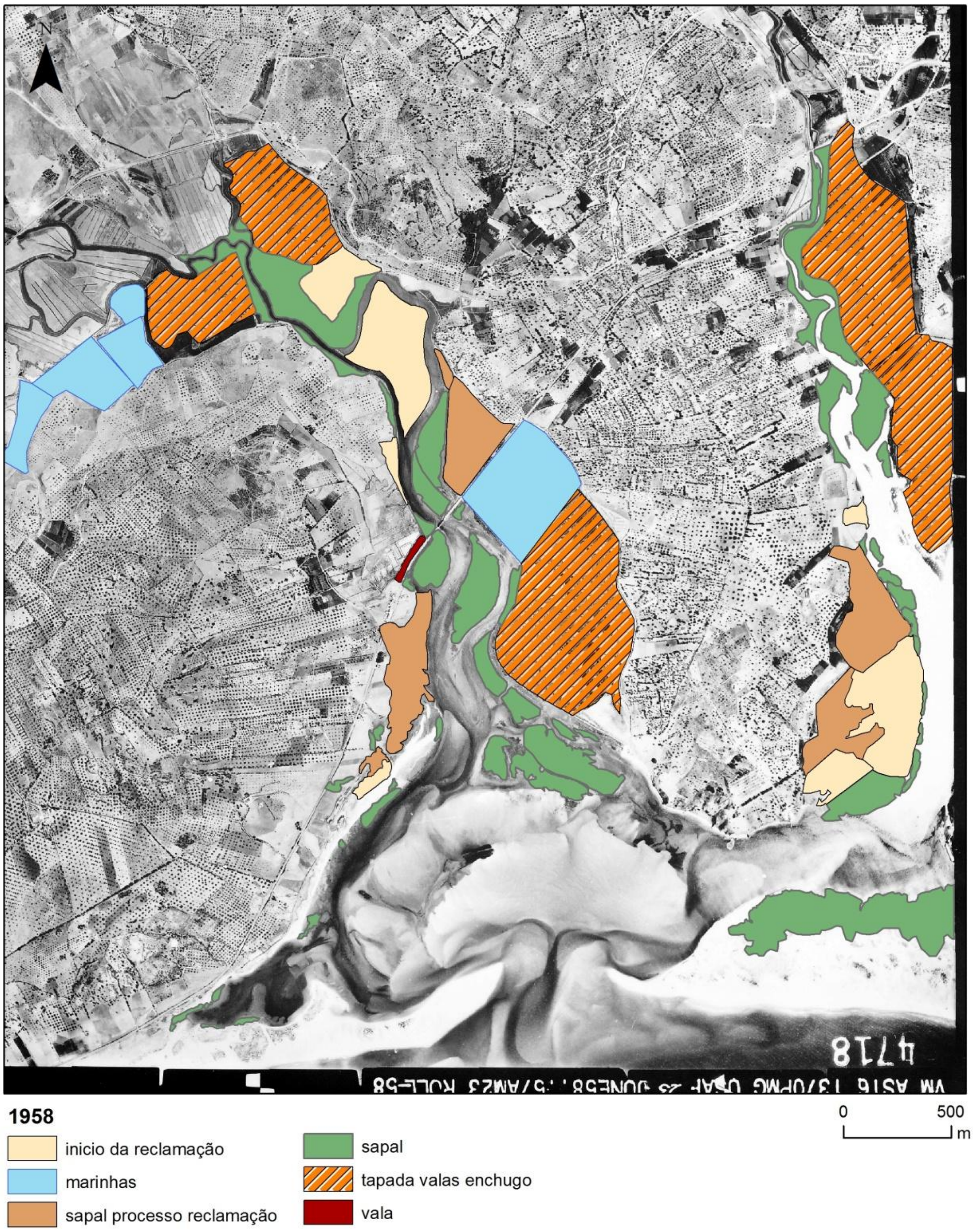


Fig. A7 Land use and cover in Alvor, 1958 (aerial photograph of 1958, CEG)

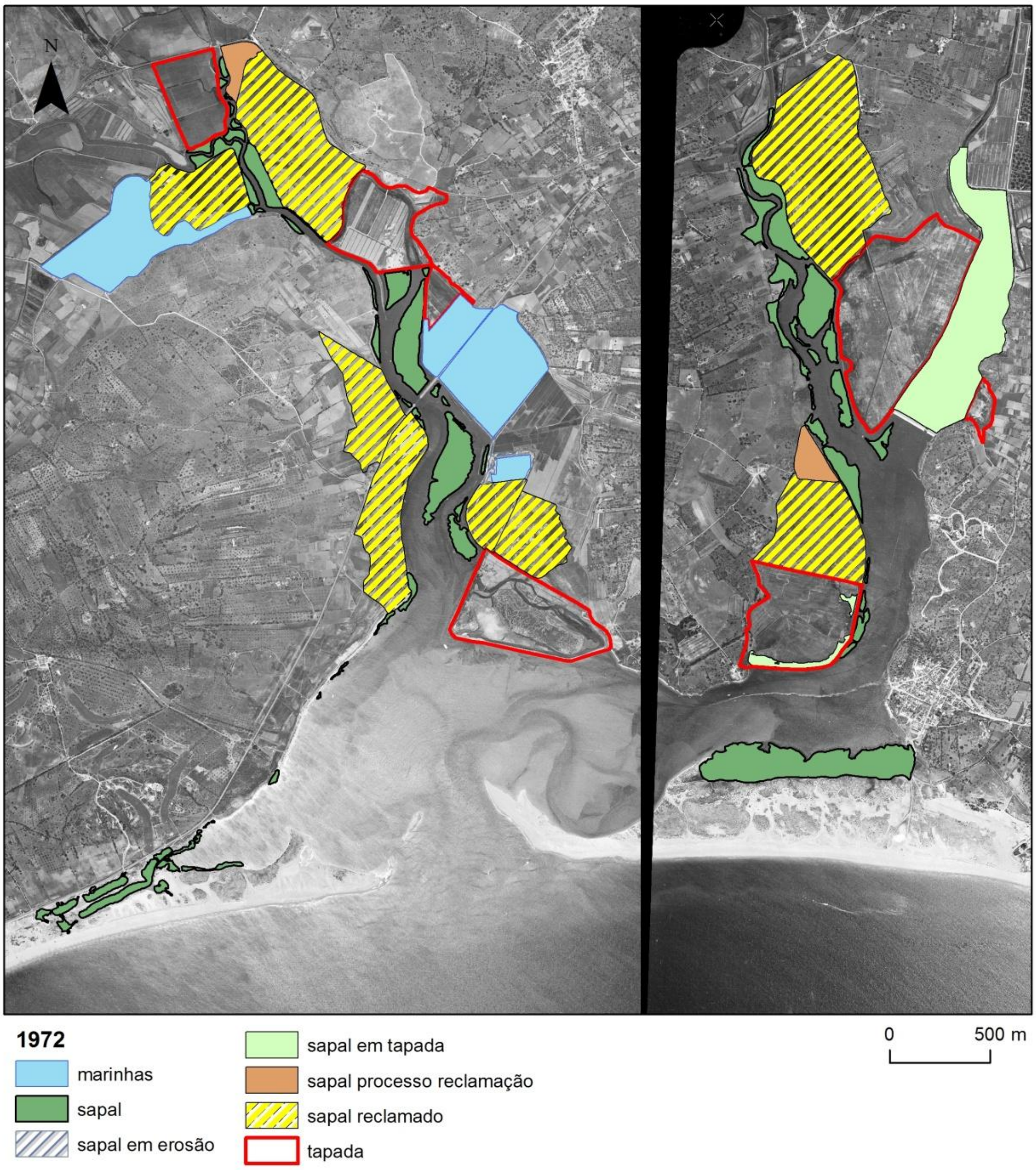


Fig. A8 Land use and cover in Alvor, 1972 (aerial photograph of 1972, CEG)

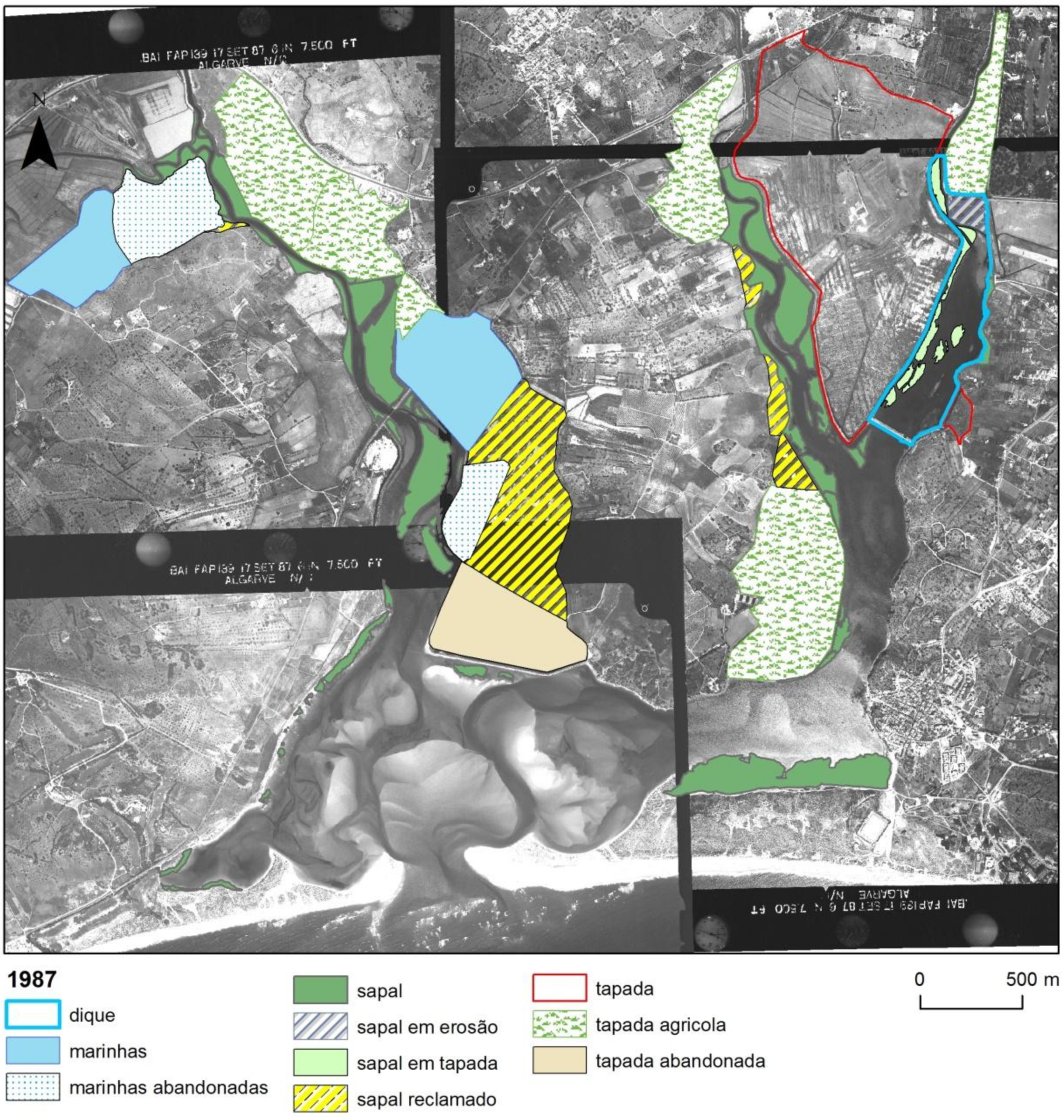


Fig. A9 Land use and cover in Alvor, 1987 (aerial photograph of 1987, CEG)

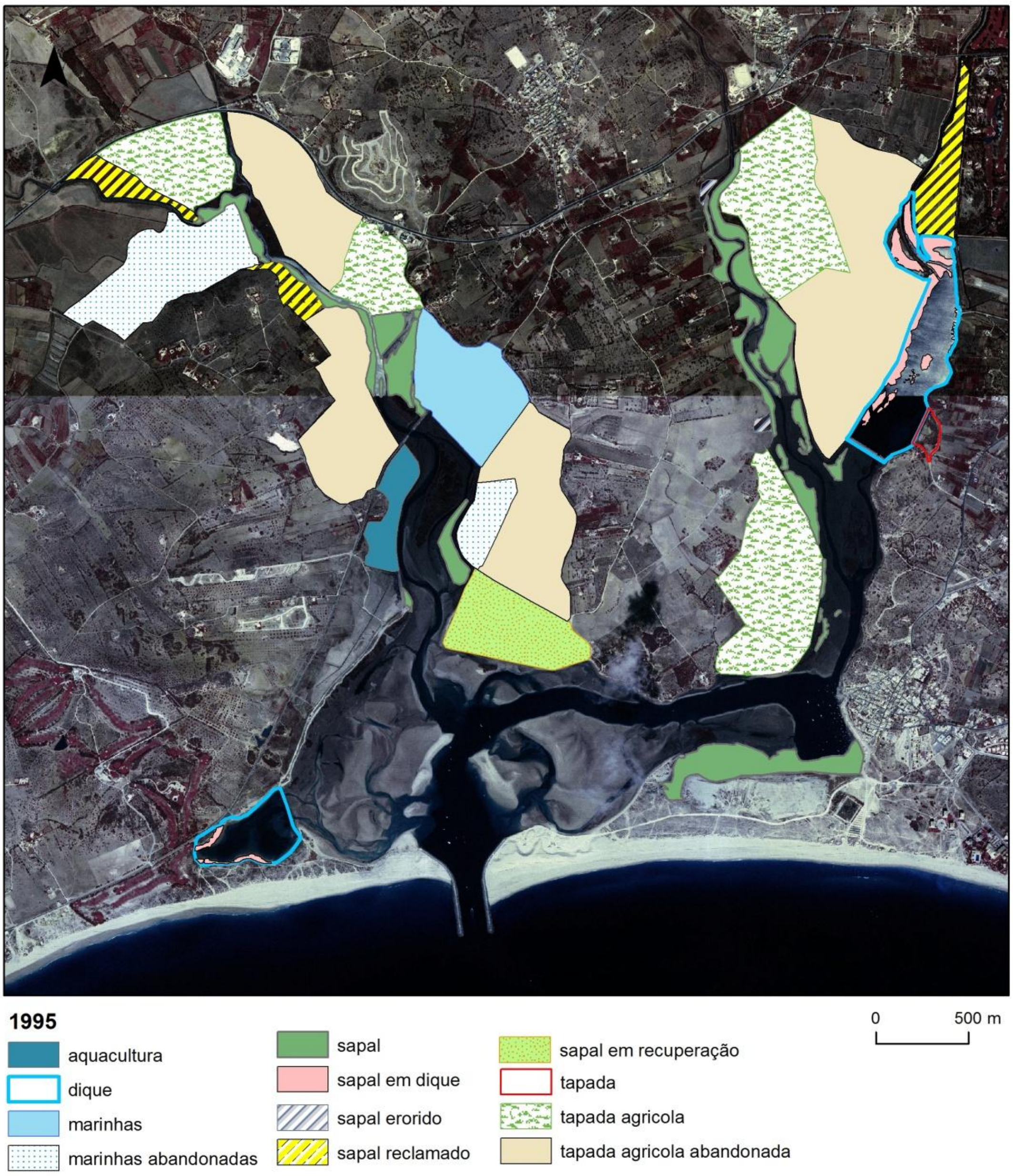


Fig. A10 Land use and cover in Alvor, 1995 (ortophotomap, 1995)

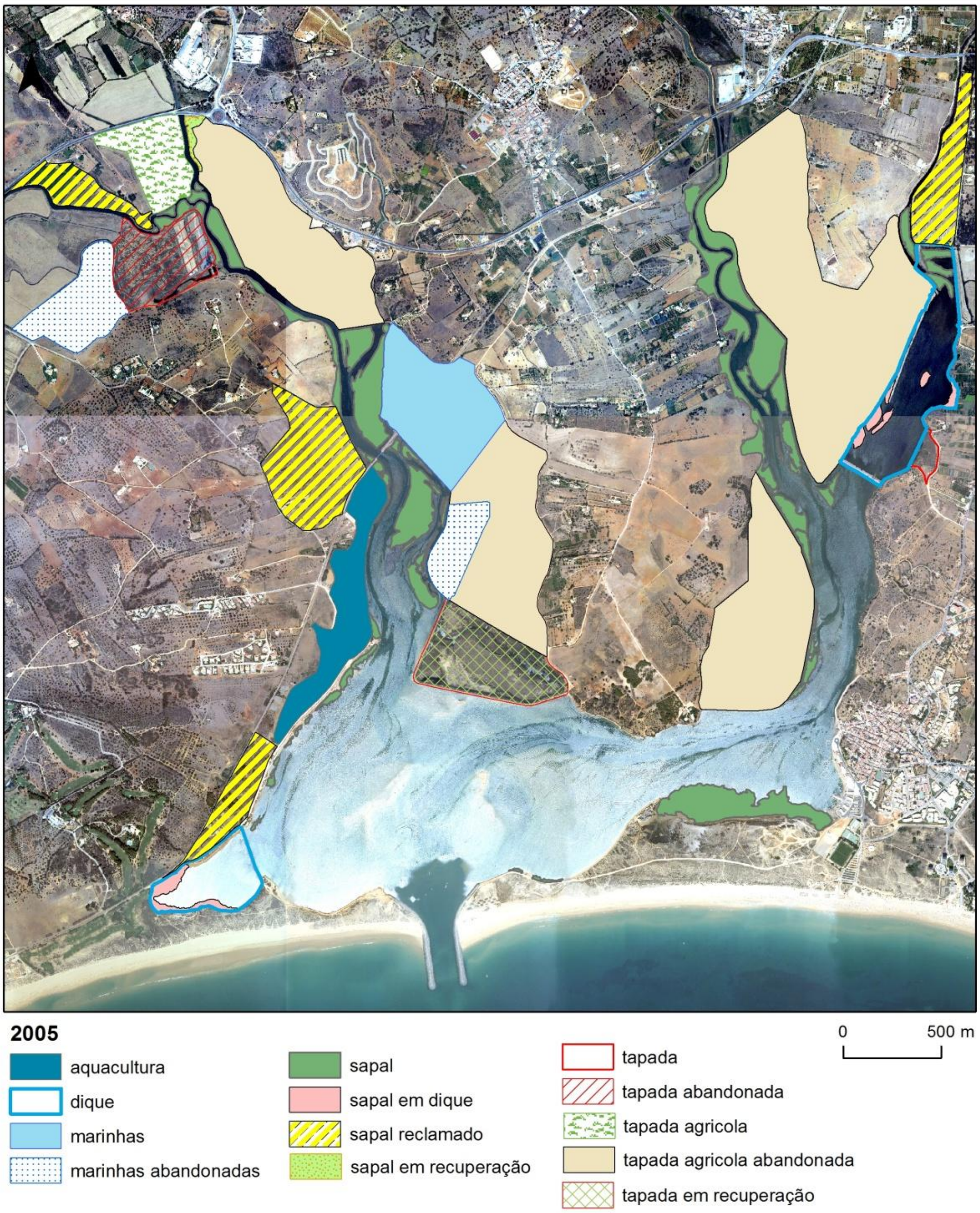


Fig. A11 Land use and cover in Alvor, 2005 (ortophotomap, 2005)

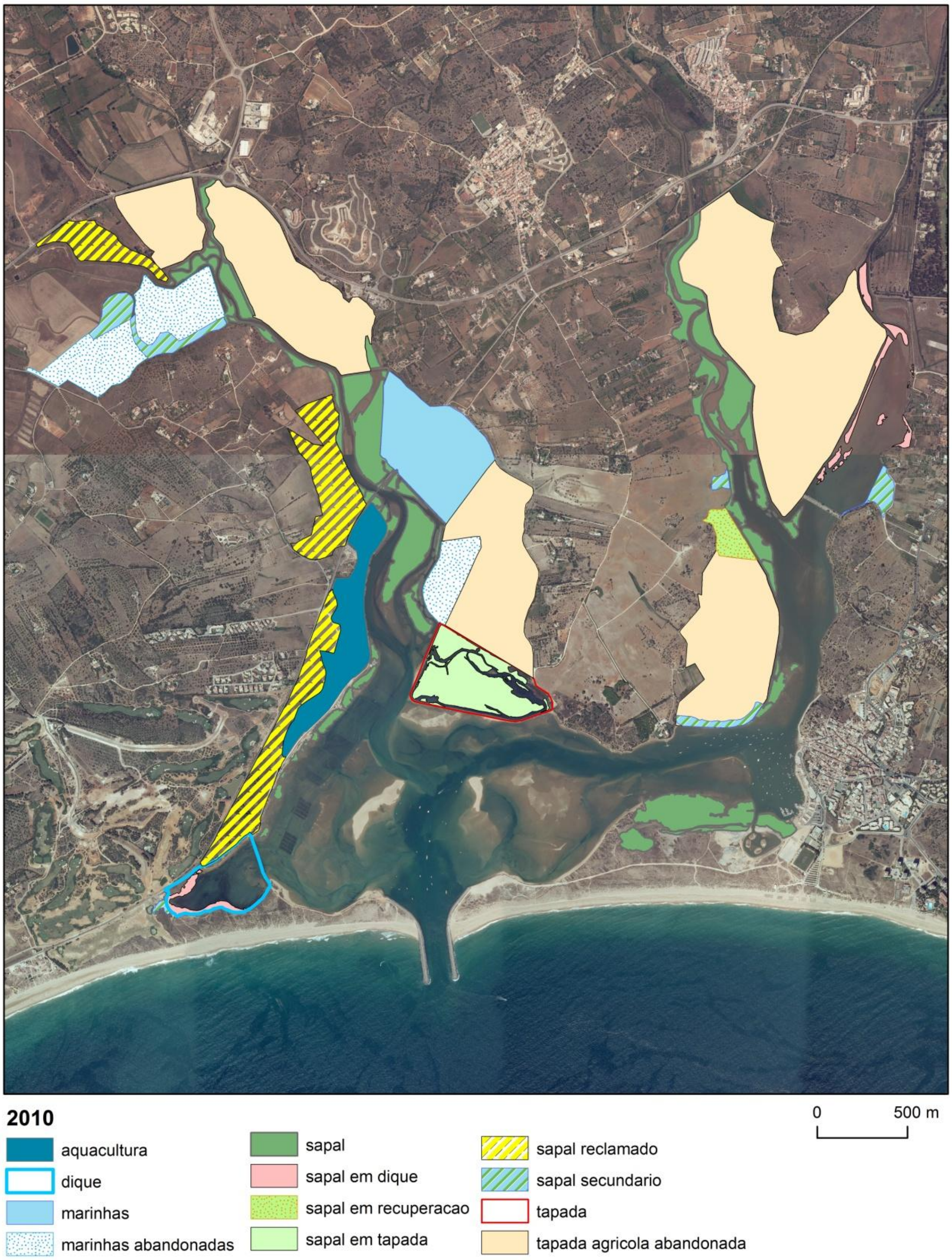


Fig. A12 Land use and cover in Alvor, 2010 (ortophotomap, 2010)

APPENDIX B - **Hydrodynamics, sedimentation and marsh morphology**

Ternary diagram

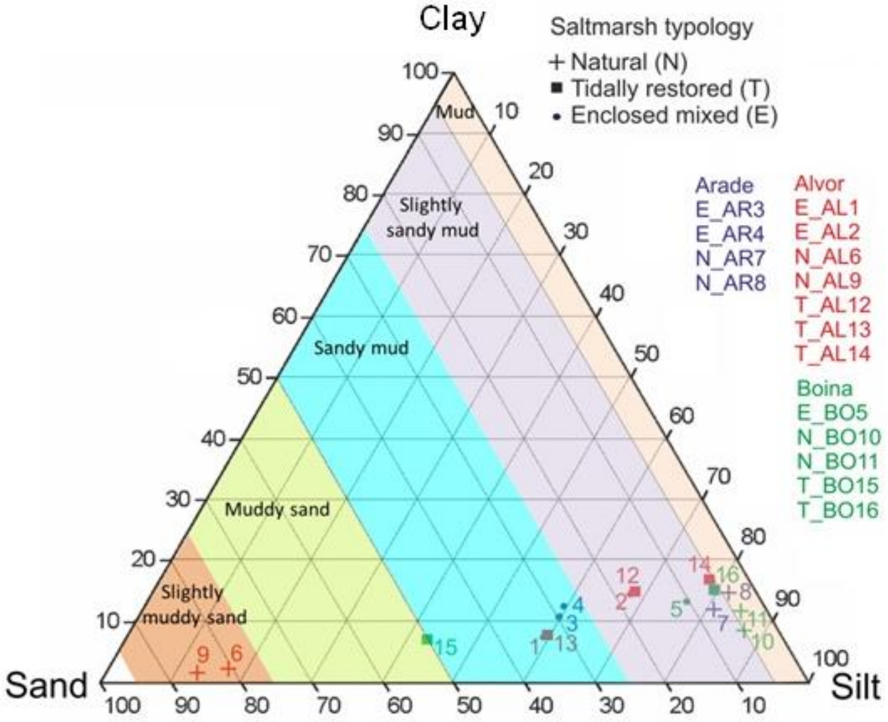


Figure B1. Ternary diagram showing the percentage of sand, silt and clay of all sediment samples.

Suspended Sediments Concentration

The suspended sediments concentration is related with sediment supply, vital to vegetation-sedimentation cycle, despite of other important interactions, like the sea level (hydro period). It is used to describe the dry mass of sediment that is suspended in a defined volume of water, during a defined period. Despite the measurements made within the scope of this work, results weren´t published.

This process contributed to the understanding of the sediment deposition onto the marsh surface, because it accounts the particles in the water column that flood the marshes expressed in g/l (Nolte et al. 2013). It was used the bottle method (Nolte et al., 2013) in one tide period. It is a pre-event method with a low cost and medium labour implicated, however its estimated accuracy is low, it was applied as a model parameter. This method was taken in the 14th of December 2013 beginning at the first low tide at 6.15 am, and ending as the next low tide at 6.30 pm. using a 1.5 litres bottle. Sampling site was chosen by its accessibility and because it marks the edge of saltmarsh appearance in the Arade River mouth. The sampling was collected manually, which encompasses some uncertainty about disturbances in the sediment layer (Nolte et al., 2013).

Samples were taken into laboratory for decantation. After this process, all water was removed and the sediments were passed through a paper filter using a *Kitasato balon*. The paper filter was dried at 60ºC and at the end of the process, sediments were weighted.

Results

Table B1. Suspended sediments concentration at Arade River

Tide aspects/height	hour	suspended sediments
		g/1.5l
low tide 1.01	**6:15**	0.20
	7:15	0.20
	8:15	0.60
	9:15	0.15
	10:15	0.20
	11:15	0.10
high tide 3.00	**12:24**	0.20
	13:24	0.20
	14:24	0.20
	15:24	0.10
	16:24	0.30
	17:24	0.20
low tide 0.98	**18:30**	0.20
total of 19.5 l (13hours of tide)		2.85

Suspended sediments (SS) results are presented in table B1 and totalize 1.9g/l measured in a whole tidal cycle: it can be noticed that is an increasing SS in the flooding tide ate 8:15h, where SS are of 0.6g/1.5l, reducing drastically in the next hour (9:15h), marking the highest concentration of sediments coming with the tide. Cumulatively, flooding period totalize more than a half of the total suspended sediments (0.83g/l), and the ebb tide characterized by lower volume of sediments in suspension – as close to the next low tide, there is an increase of SS (0.3g/1.5). These results have been integrated in the hydrodynamic model of Arade (see Figures B1 to B3), developed by Teles et al., 2013.

The hydrodynamic model for Arade River (Teles et al., 2013) shows higher velocity with the incoming tide, especially with higher incidence in the central river channel. Tide influence is spread upstream Fontes (marks the northeast part of the study area) with a maximum velocity of 0.25m/s, while between Mexilhoeira da Carregação and E.N.125, the velocity reaches 0.50m/s. The incoming tide plays a significant role in sediment supply in the Boina creek segment; Figure 2 shows that velocity is the highest (0.50-0.75m/s) at the entrance of Boina, influenced by a riprap at this location. Nevertheless, velocity downgrades to a maximum of 0.50m/s along the creek. Regarding salinity, both Boina creek and the Arade´s middle segment concentrate 27.5-35g/kg, downgrading from Fontes forward to 20-27.5g/kg, and the salinity concentration diminishes upstream Arade river /see Fig.B3 and B4).

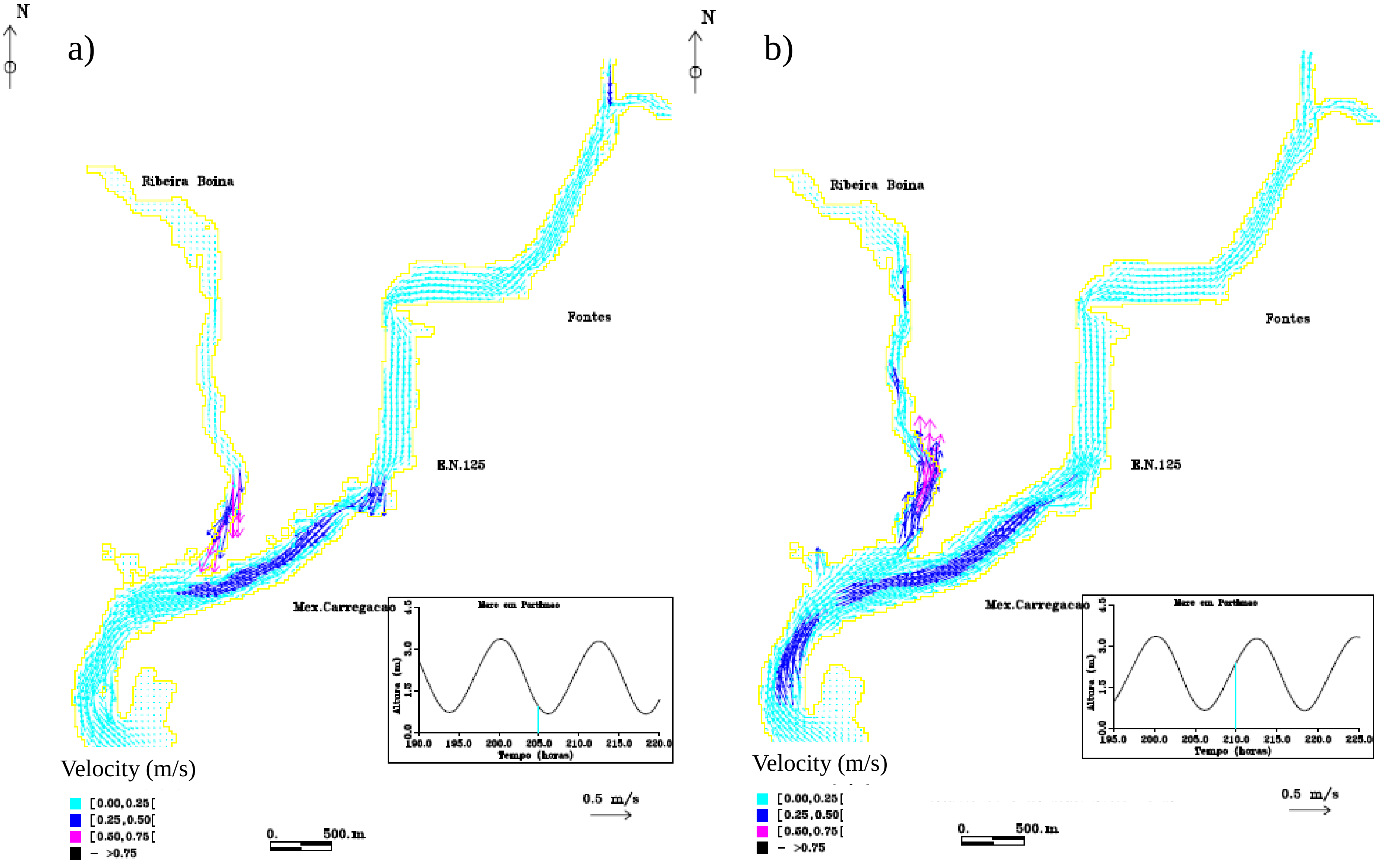


Fig.B2 Tide propagation model of Arade River and Boina creek – Velocity field

a) Ebb tide - simulation period: 205h | 0.5 m/s b) Flooding tide - simulation period: 210h | 0.5 m/s

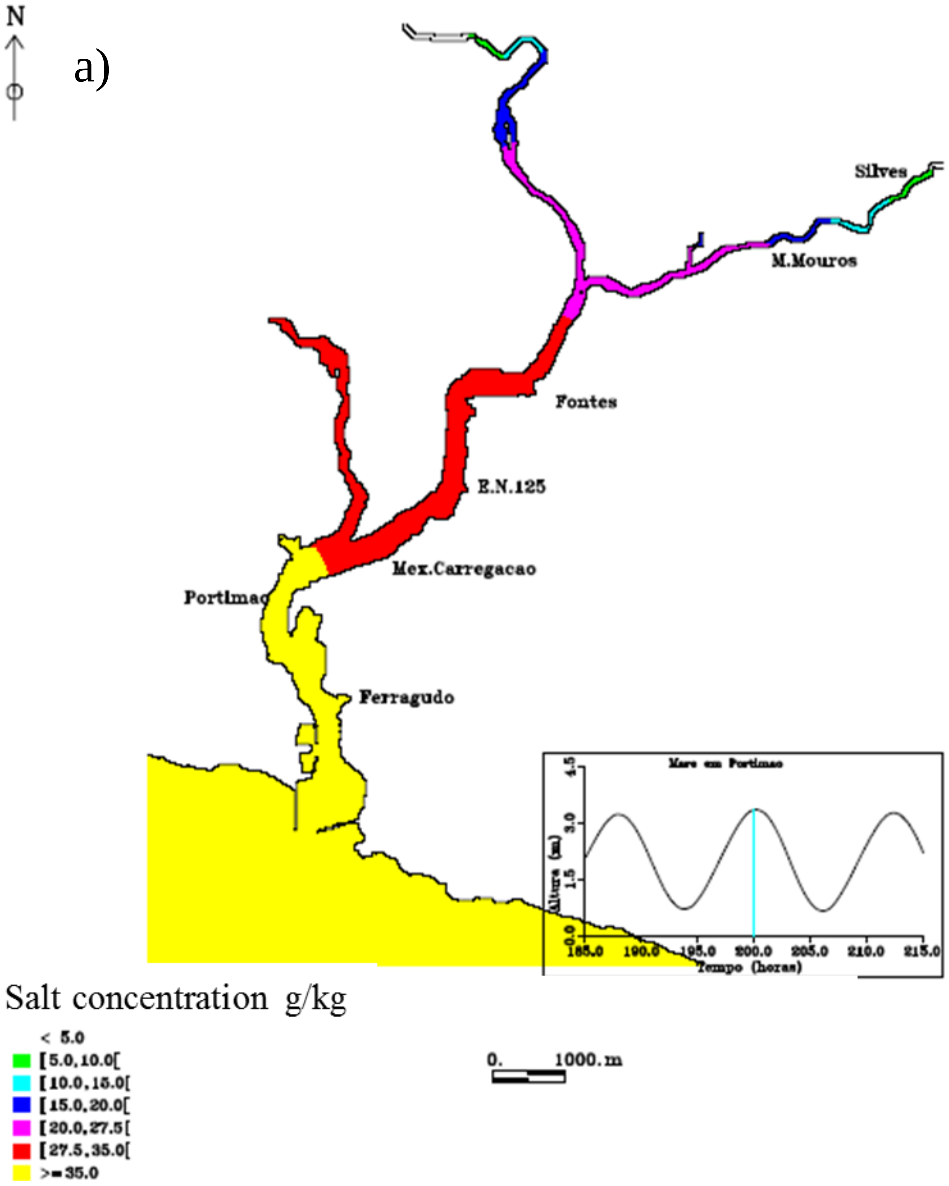


Fig.B3 Tide propagation model of Arade River and Boina creek – salinity field 200h/205h

a) Simulation period: 200 hours b) Simulation period: 205 hours

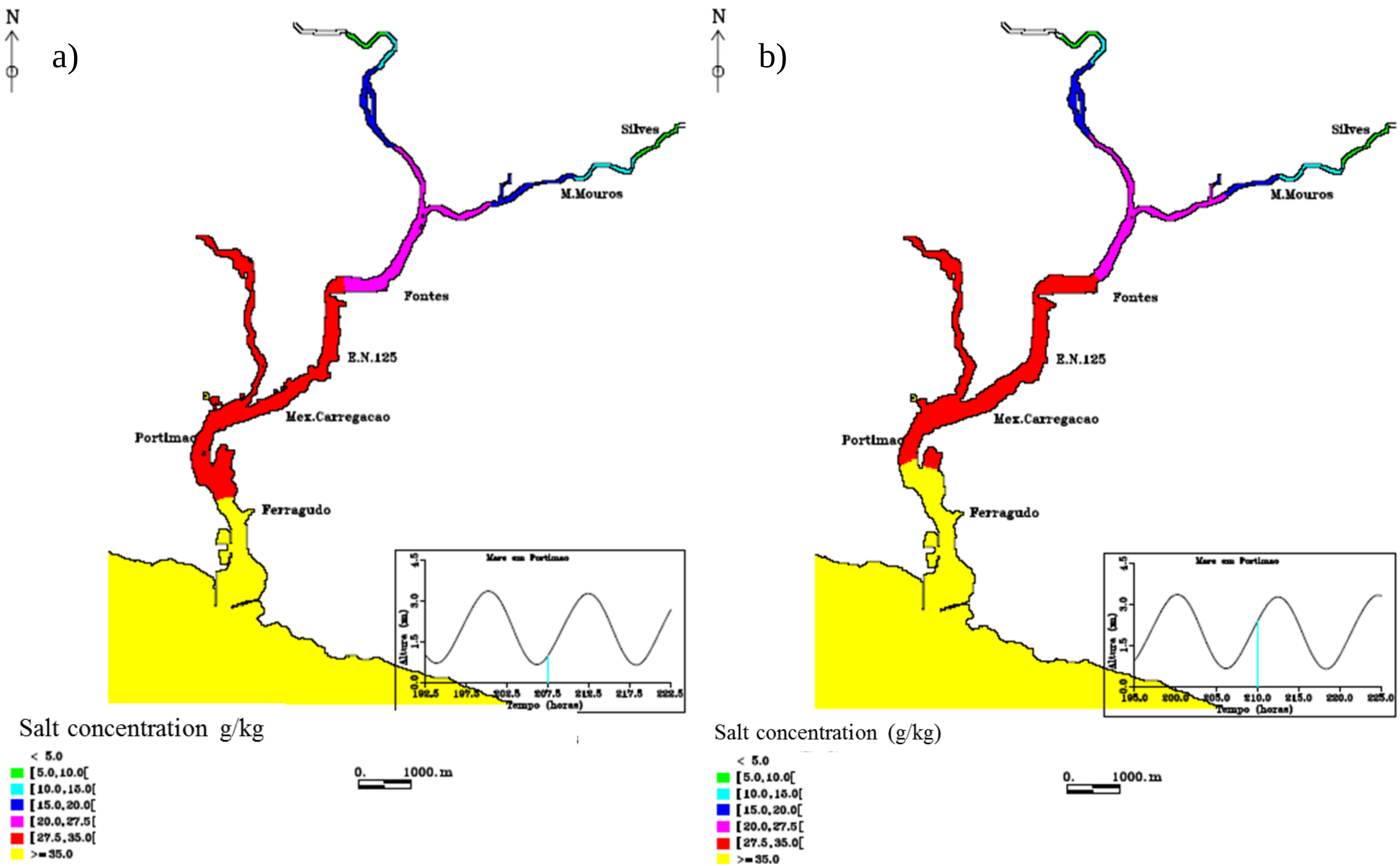


Fig.B4 Tide propagation model of Arade River and Boina creek – salinity field

a) Simulation period: 207.5 hours b) Simulation period: 210 hours

APPENDIX C – **Landscape metrics calculations**

Table C1. Landscape metrics calculations for Arade 1972

			ARADE 1972				
Patch type	NumP	ED	Divers	AWMSI	AWMPFD	MPI	MNN
low	15	87.07	5.92	1.23	1.67	7.26	121.01
low_med	9	369.03	65.82	1.69	1.45	331.81	180.39
med_low	1	25.98	1.47	2.16	1.54	0.00	47.25
med_high	2	19.63	0.66	1.74	1.61	13.97	11.94
high	2	12.83	0.35	1.66	1.68	2.12	6.75
t_low	21	124.16	13.27	1.36	1.72	5.75	150.02
t_low_med	6	97.54	6.70	1.61	1.53	11.98	152.60
t_med_low	2	26.07	0.82	2.05	1.63	0.02	52.37
t_med	5	54.82	2.36	1.66	1.60	0.07	63.34
t_med_high	4	69.76	1.29	3.05	1.80	0.08	123.09
t_high_med	2	46.25	1.34	2.73	1.66	29.40	103.81
mean	*6.27*	*84.83*	*9.09*	*1.90*	*1.63*	*36.59*	*92.05*
st.deviation	*6.09*	*96.03*	*18.32*	*0.53*	*0.09*	*93.74*	*56.30*

Table C2. Landscape metrics calculations for Arade 2010

			ARADE 2010				
Patch type	NumP	ED	Divers	MSI	MPFD	MPI	MNN
low	18	346.57	34.76	1.73	1.60	187.32	169.97
low_med	5	135.51	15.00	1.82	1.46	10.38	270.79
med	2	12.77	0.34	1.67	1.62	0.00	6.76
med_high	1	7.37	0.24	1.67	1.60	0.00	10.26
high	2	38.53	1.47	2.46	1.61	62.04	219.18
t_low	52	319.95	21.29	1.30	1.84	303.65	79.26
t_low_med	8	190.68	18.26	2.01	1.52	1443.82	108.74
t_med	4	57.88	1.18	3.00	1.79	16.78	76.89
t_med_high	3	16.37	0.34	1.79	1.72	4.25	3.52
t_high	9	202.27	5.81	3.13	1.71	11.29	114.40
mean	*10.40*	*132.79*	*9.87*	*2.06*	*1.65*	*203.95*	*105.98*
std.deviation	*14.66*	*121.29*	*11.33*	*0.57*	*0.11*	*424.30*	*86.73*

Table C3. Landscape metrics calculations for Boina 1972

BOINA 1972

Patch type	NumP	ED	Divers	AWMSI	AWMPFD	MPI	MNN
low	17	119.68	6.03	1.43	1.67	25.00	108.35
low_med	5	131.13	12.06	1.88	1.48	1.06	116.89
med_low	3	97.49	5.55	2.76	1.66	0.87	51.60
med	6	89.75	2.53	2.52	1.70	50.16	129.20
med_high	2	70.27	2.77	3.05	1.60	0.00	118.90
high	1	11.03	0.31	2.08	1.66	0.00	5.26
t_low	15	142.77	7.71	1.57	1.61	216.25	121.22
t_low_med	3	98.94	15.92	2.02	1.63	0.19	142.88
t_med_low	6	258.56	24.49	2.69	1.56	330.45	102.35
t_med	5	81.91	3.94	2.04	1.59	3.48	114.74
t_med_high	2	95.88	2.08	4.77	1.78	186.64	217.14
t_high	2	28.65	0.70	2.69	1.76	0.02	129.83
t_agro	1	47.92	15.91	1.27	1.30	0.00	43.23
mean	*5.23*	*98.00*	*7.69*	*2.37*	*1.61*	*62.62*	*107.81*
std.deviation	*4.90*	*59.05*	*7.04*	*0.87*	*0.12*	*104.87*	*49.95*

Table C4. Landscape metrics calculations for Boina 2010

BOINA 2010

Patch type	NumP	ED	Divers	AWMSI	AWMPFD	MPI	MNN
low	20	250.83	14.71	1.68	1.62	277.40	133.82
med_low	2	143.08	14.20	3.04	1.59	0.02	20.89
med_high	1	15.24	0.46	2.21	1.65	0.00	137.22
t_low	25	318.84	22.62	1.60	1.69	377.77	99.43
t_low_med	4	161.34	27.61	1.75	1.48	41.61	168.33
t_med_low	17	325.51	14.95	2.13	1.64	924.09	50.32
t_med	2	55.75	1.46	3.21	1.71	0.06	105.87
t_med_high	3	44.42	1.09	2.41	1.70	0.02	89.56
t_high	4	95.94	2.73	2.82	1.69	5.35	119.85
mean	*8.67*	*156.77*	*11.09*	*2.32*	*1.64*	*180.70*	*102.81*
std.deviation	*8.74*	*110.73*	*9.54*	*0.56*	*0.07*	*294.67*	*42.68*

Table C5. Landscape metrics calculations for Alvor 1972

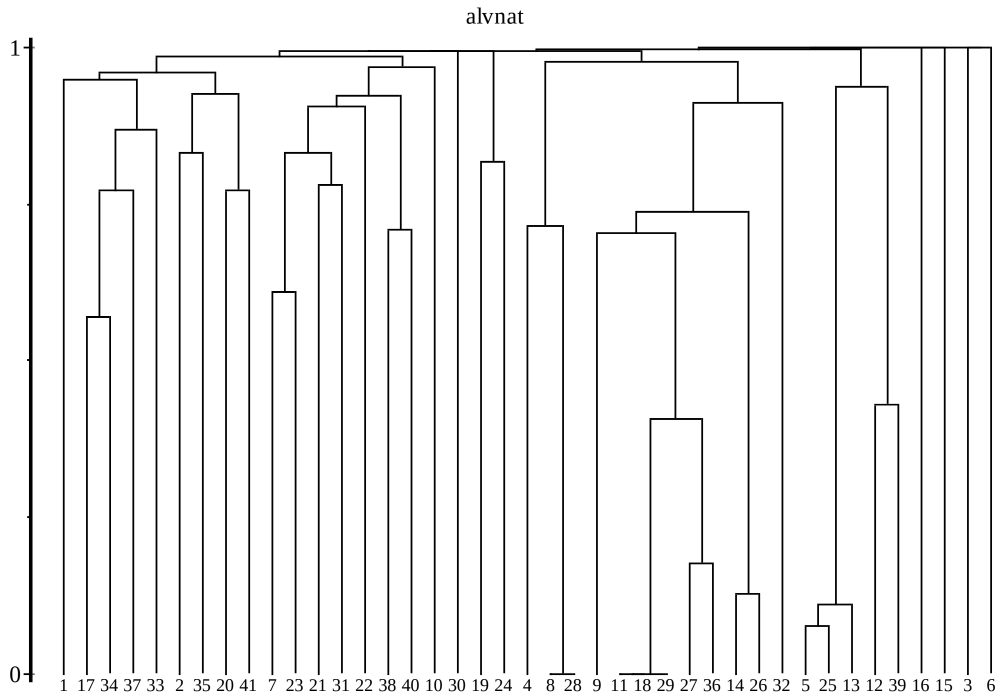
			ALVOR 1972				
Patch type	NumP	ED	Divers	AWMSI	AWMPFD	MPI	MNN
low	9	36.68	126.12	1.43	1.33	73.70	87.06
low_med	3	36.95	68.18	2.40	1.46	6.86	53.30
t_low	26	79.61	355.30	2.05	1.46	79.28	77.99
t_low_med	4	22.94	253.23	2.19	1.51	0.30	51.89
t_med	7	53.37	130.30	2.18	1.46	169.21	64.18
t_med_low	3	29.07	57.07	2.11	1.41	0.05	62.74
t_high	1	8.17	416.74	3.85	1.71	0.00	2.53
e_low_med	5	62.69	75.46	2.80	1.48	1.85	32.13
e_med	3	21.11	172.81	2.35	1.50	1.53	107.09
e_med_high	3	39.05	84.79	2.75	1.50	0.06	112.50
e_agricult	3	94.33	5.28	1.75	1.30	0.26	125.57
mean	*6.09*	*44.00*	*158.66*	*2.35*	*1.47*	*30.28*	*70.63*
std.deviation	*6.64*	*24.94*	*124.57*	*0.61*	*0.10*	*52.50*	*34.77*

Table C6. Landscape metrics calculations for Alvor 2010

			ALVOR 2010				
Patch type	NumP	ED	Divers	AWMSI	AWMPFD	MPI	MNN
low	22	120.20	111.28	1.56	1.55	704.50	68.18
med_low	1	12.77	123.20	2.81	1.55	0.00	21.58
med	2	19.81	136.54	2.81	1.61	0.00	12.61
med_high	3	23.59	197.53	2.16	1.59	0.11	9.67
high_med	2	52.31	59.79	3.77	1.56	0.03	24.00
high	3	38.44	135.09	3.13	1.62	58.68	34.84
t_med	1	23.36	30.12	2.54	1.45	0.00	94.64
t_high	1	3.21	745.70	1.74	1.56	0.00	30.04
e_waterchannel	4	76.97	209.44	5.42	1.76	1.83	48.32
e_low	1	4.43	218.28	1.30	1.40	0.00	26.73
e_med_low	1	10.72	67.78	1.75	1.41	0.00	14.48
e_med	10	147.16	28.76	1.66	1.58	408.37	117.03
e_med_high	13	109.80	54.67	1.81	1.46	1560.71	97.29
e_high	6	75.62	124.85	2.82	1.60	0.87	53.69
mean	*5.00*	*51.31*	*160.22*	*2.52*	*1.55*	*195.36*	*46.65*
std.deviation	*5.89*	*45.48*	*173.24*	*1.06*	*0.09*	*428.29*	*33.65*

APPENDIX D – **Ecological classification**

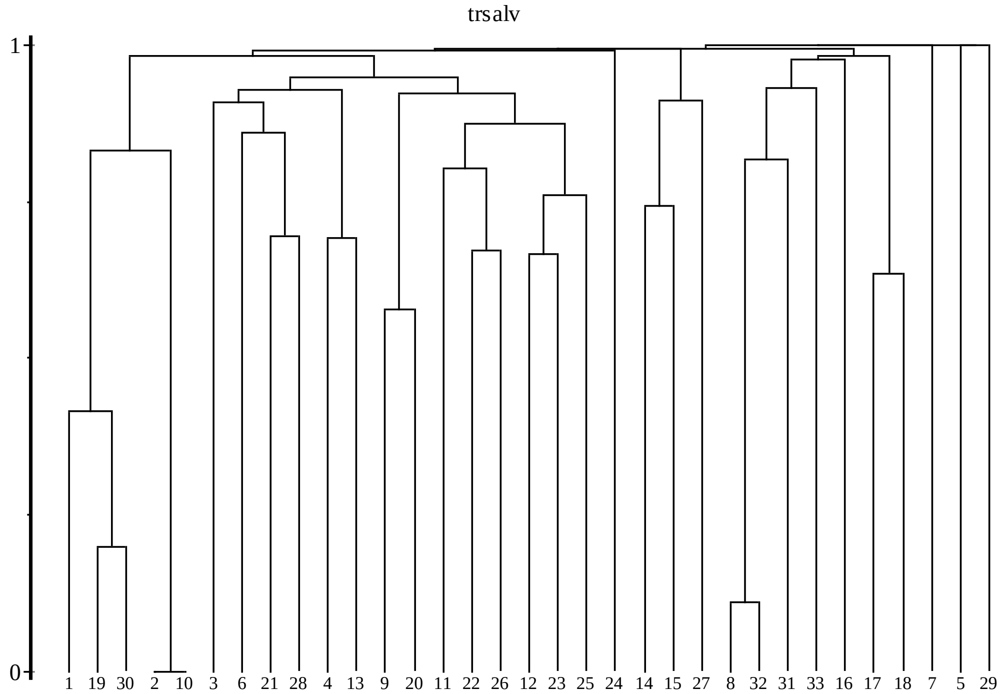
Figure D1. Unweighted Pair Group Method with Arithmetic Mean (UPGMA) using Bray-Curtis coefficient for natural saltmarshes of Alvor and list of species in order.



Alvor_natural saltmarshes

1 *Artemisia crithmifolia*
2 *Arthrocnemum macrostachyum*
3 *Asparagus albus*
4 *Atriplex halimus*
5 *Atriplex patula*
6 *Bolboschoenus glaucus*
7 *Bromus lanceolatus*
8 *Calendula arvensis*
9 *Carpobrotus edulis*
10 *Cistanche philypaea*
11 *Cotula coronopifolia*
12 *Elytrigia juncea*
13 *Elytrigia elongata*
14 *Emex spinosa*
15 *Ferula tingitana*
16 *Frankenia laevis*
17 *Halimione portulacoides*
18 *Hypochaeris radicata*
19 *Inula critmoides*
20 *Limoniastrum monopetalum*
21 *Limonium algarvense*
22 *Myriolimion diffusum*
23 *Limonium lanceolatum*
24 *Limonium vulgare*
25 *Lotus creticus*
26 *Medicargo polymorpha*
27 *Mellilotus segetalis*
28 *Oxalis pes-caprae*
29 *Polypogon maritimus*
30 *Puccinellia iberica*
31 *Puccinellia maritima*
32 *Salsola vermiculata*
33 *Sarcocornia perennis* ssp.*alpini*
34 *Sarcocornia perennis* ssp. *perennis*
35 *Sarcocornia pruinosa*
36 *Scorpiurus vermiculata*
37 *Spartina maritima*
38 *Spergularia media*
39 *Sporobolus pungens*
40 *Suaeda albescens*
41 *Suaeda vera*

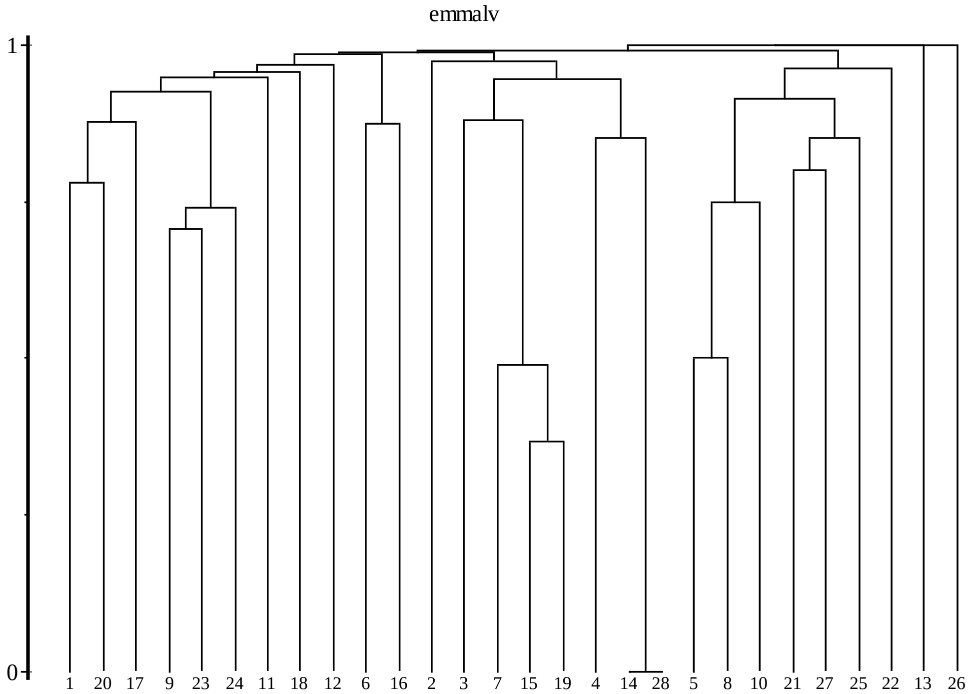
Figure D2. Unweighted Pair Group Method with Arithmetic Mean (UPGMA) using Bray-Curtis coefficient for tidally restored saltmarshes of Alvor and list of species in order.



Alvor_tidally restored saltmarshes

1 *Anagallis arvensis*
2 *Artemisia crithmifolia*
3 *Arthrocnemum macrostachyum*
4 *Atriplex halimus*
5 *Bolboschoenus glaucus*
6 *Bromus lanceolatus*
7 *Carpobrotus edulis*
8 *Cistanche philypaea*
9 *Cotula coronopifolia*
10 *Emex spinosa*
11 *Frankenia leavis*
12 *Halimione portulacoides*
13 *Hypochaeris radicata*
14 *Juncus acutus*
15 *Juncus maritimus*
16 *Limoniastrum monopetalum*
17 *Limonium algarvense*
18 *Limonium lanceolatum*
19 *Medicago polymorpha*
20 *Mesembryanthemum nodiflorum*
21 *Polypogon maritimus*
22 *Puccinellia iberica*
23 *Salicornia ramosíssima*
24 *Salsola vermiculata*
25 *Sarcocornia alpini*
26 *Sarcocornia pruinosa*
27 *Sarcocornia perennis* ssp. *perennis*
28 *Sonchus maritimus*
29 *Spartina maritima*
30 *Spergularia bocconei*
31 *Spergularia media*
32 *Suaeda albescens*
33 *Suaeda vera*

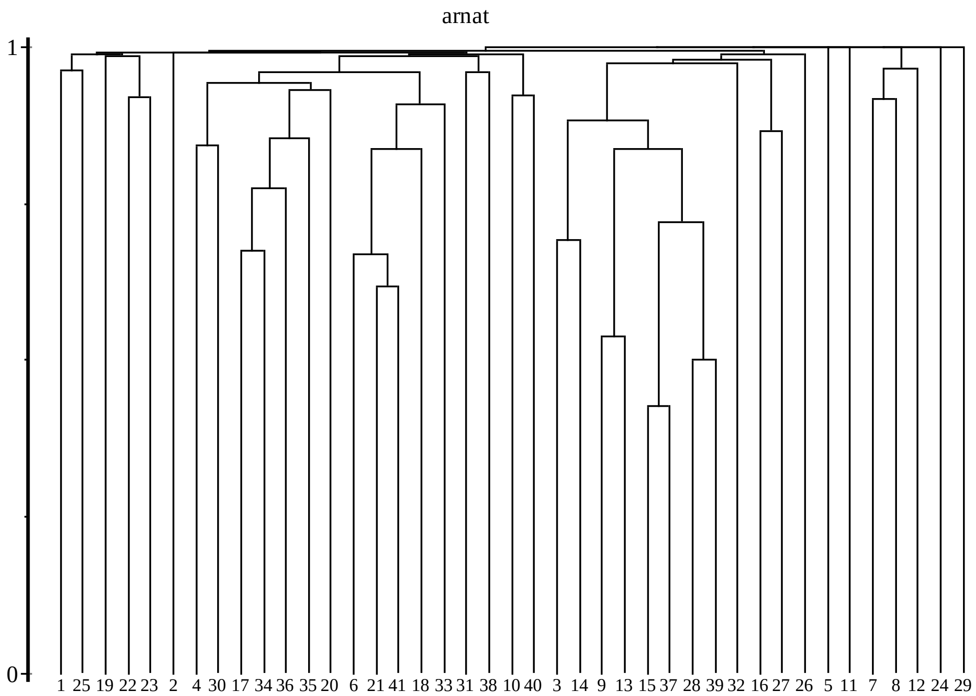
Figure D3. Unweighted Pair Group Method with Arithmetic Mean (UPGMA) using Bray-Curtis coefficient for enclosed mix marshes of Alvor and list of species in order.



Alvor_enclosed mix marshes

1	*Arthrocnemum macrostachyum*	15	*Melilotus segetalis*
2	*Atriplex halimus*	16	*Polycnemum arvence*
3	*Bolboschoenus glaucus*	17	*Polypogon maritimus*
4	*Brachyponium phoenicoides*	18	*Puccinellia ibérica*
5	*Bromus lanceolatus*	19	*Puccinellia maritima*
6	*Carpobrotus edulis*	20	*Salicornia ramosíssima*
7	*Cotula coronopifolia*	21	*Salsola vermiculata*
8	*Cynara cardunculus*	22	*Sarcocornia perennis* ssp. *perennis*
9	*Halimione portulacoides*	23	*Sarcocornia perennis* ssp. *alpini*
10	*Hypochaeris radicata*	24	*Sarcocornia pruinosa*
11	*Juncus acutus*	25	*Sonchus maritimus*
12	*Juncus maritimus*	26	*Suaeda albescens*
13	*Medicago polymorpha*	27	*Suaeda vera*
14	*Melica minuta*	28	*Taraxacum officinale*

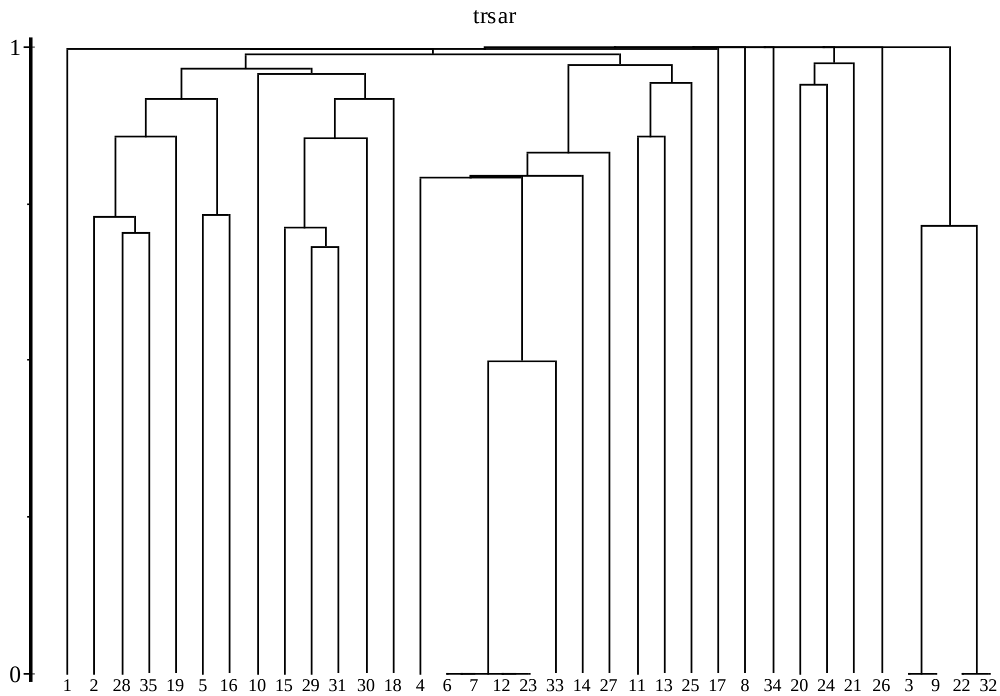
Figure D4. Unweighted Pair Group Method with Arithmetic Mean (UPGMA) using Bray-Curtis coefficient for natural saltmarshes of Arade and list of species in order.



Arade_natural saltmarshes

1 *Artemisia crithmifolia*
2 *Artemisia crithmifolia*
3 *Artemisia gallica*
4 *Arthrocnemum macrostachyum*
5 *Aster tripolium* ssp.*pannonicus*
6 *Atriplex halimus*
7 *Atriplex prostrata*
8 *Bolboschoenus maritimus* var. *maritimus*
9 *Bromus lanceolatus*
10 *Carpobrotus edulis*
11 *Cistanche philypaea*
12 *Cotula coronopifolia*
13 *Digitaria sanguinalis*
14 *Elytrigia enlongatus*
15 *Elytrigia juncea*
16 *Frankenia laevis*
17 *Halimione portulacoides*
18 *Inula crithmoides*
19 *Juncus acutus*
20 *Juncus maritimus*
21 *Limoniastrum monopetalum*
22 *Limonium algarvense*
23 *Limonium lanceolatum*
24 *Limonium narbonense*
25 *Limonium vulgare*
26 *Medicago polymorpha*
27 *Oxalis pes-caprae*
28 *Polypogon maritimus*
29 *Puccinellia iberica*
30 *Puccinellia maritima*
31 *Salicornia ramosissima*
32 *Salsola soda*
33 *Salsola vermiculata*
34 *Sarcocornia perennis* ssp..*alpini*
35 *Sarcocornia perennis* ssp. *perennis*
36 *Sarcocornia pruinosa*
37 *Sonchus maritimus*
38 *Spartina maritima*
39 *Spergularia bocconei*
40 *Suaeda albescens*
41 *Suaeda vera*

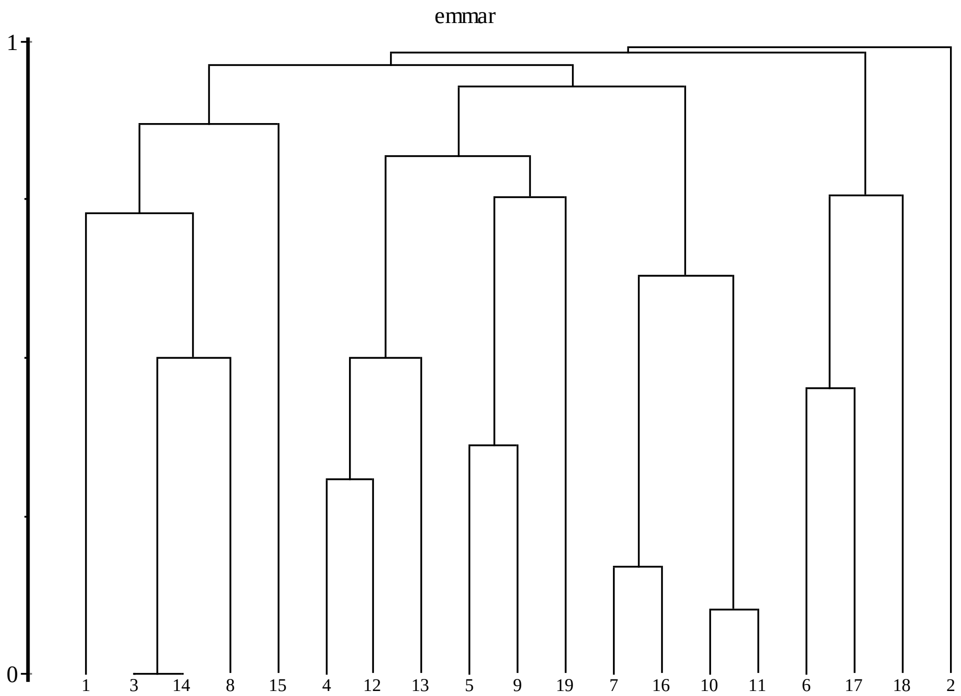
Figure D5. Unweighted Pair Group Method with Arithmetic Mean (UPGMA) using Bray-Curtis coefficient for tidally restored saltmarshes of Arade and list of species in order.



Arade_tidally restored saltmarshes

1 *Artemisia gallica*
2 *Arthrocnemum macrostachyum*
3 *Aspargos albus*
4 *Aster tripolium* ssp. pannonicus
5 *Atriplex halimus*
6 *Atriplex patula*
7 *Atriplex prostrata*
8 *Bolboschoenus maritimus* var. *compactus*
9 *Sedum sediforma*
10 *Cistanche phylipaea*
11 *Elytrigia elongata*
12 *Emex spinosa*
13 *Ferula tingitana*
14 *Frankenia leavis*
15 *Halimione portulacoides*
16 *Inula crithmoides*
17 *Juncus acutus*
18 *Juncus maritimus*
19 *Limoniastrum monopetalum*
20 *Limonium algarvense*
21 *Limonium lanceolatum*
22 *Medicargo polymorpha*
23 *Melilotus segetalis*
24 *Oxalis pes-caprae*
25 *Puccinellia iberica*
26 *Puccinellia maritima*
27 *Salicornia ramosíssima*
28 *Salsola vermiculata*
29 *Sarcocornia perennis* ssp. *alpini*
30 *Sarcocornia pruinosa*
31 *Sarcocornia perennis* ssp. *perennis*
32 *Scorpiurus vermiculata*
33 *Sonchus maritimus*
34 *Spartina maritima*
35 *Suaeda vera*

Figure D6. Unweighted Pair Group Method with Arithmetic Mean (UPGMA) using Bray-Curtis coefficient for enclosed mix marshes of Arade and list of species in order.



Arade_enclosed mix marshes

1 *Arthrocnemum macrostachyum*
2 *Artemisia gallica*
3 *Atriplex halimus*
4 *Elytrigia elongata*
5 *Elytrigia juncea*
6 *Halimione portulacoides*
7 *Inula critmoides*
8 *Juncus acutus*
9 *Limoniastrum monopetalum*
10 *Limonium algarvense*
11 *Limonium vulgare*
12 *Puccinellia iberica*
13 *Salicornia ramosissima*
14 *Salsola soda*
15 *Salsola vermiculata*
16 *Sarcocornia pruinosae*
17 *Sarcocornia perennis* ssp. *perennis*
18 *Sarcocornia perennis* ssp.*alpini*
19 *Suaeda vera*