The application of landscape ecology techniques for managing disturbed Mediterranean coastal seascapes.

By

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6 June 2014

The work presented in this thesis was conducted at the Insititut de Ciència i Tecnologia Ambientals (ICTA).This thesis should be cited as: Nowell, M.S. (2014). The application of landscape ecology techniques for managing disturbed Mediterranean coastal seascapes. PhD Thesis, Universitat Autònoma de Barcelona, Barcelona, Spain.

Summary

Anthropogenic pressure is the preeminent threat to the biodiversity, resilience and the ecological functioning of the coastal and marine environment. Developing effective responses to this multifaceted threat is a significant challenge for managers and decision-makers due to the complexities of poorly understood marine ecosystems. The ecologically meaningful interpretation of spatial data using seascape ecology techniques has the potential to be a powerful conservation tool. The emerging discipline of seascape ecology explores the causes and consequences of spatial patterns in the marine environment. In this dissertation, seascape ecology techniques are used to understand the ecological consequences of anthropogenic disturbance on the spatial patterns of coastal Mediterranean seascapes through three studies.

In the first study, the relationship between spatial patterns and biodiversity was explored. Quantification of seascape structure using spatial pattern metrics showed that species richness can be conserved by protecting the diversity of habitats in the seascape. The results also highlighted the importance of patch complexity for increased species richness. Using spatial metrics and multivariate analysis, the effects of anthropogenic pressures on seascape structure was determined. This approach allowed for the influence of different disturbance variables on specific components of the seascape to be assessed. The results indicate that land-based pollution is causing fragmentation of the seascape and has the largest influence on the composition of the mosaic of habitats. In the final study, the effect of these disturbance variables on ecosystem service delivery was determined using carbon stocks as an indicator ecosystem service. Land-based pollution emerged as an important driver of seascape structure and seagrass carbon capture. The analysis emphasized the importance of habitat context within the seascape for ecosystem service delivery.

I conclude that seascape quantification techniques provide valuable information on the causes and consequences of spatial patterns in coastal Mediterranean seascapes. Quantifying seascape structure using spatial pattern metrics is an effective and consistent technique for the ecologically meaningful evaluation of spatial data at the scales required for management. The approaches presented in this dissertation are valuable and informative tools for conservation planning.

Resumen de la tesis

La presión antropogénica es la amenaza preeminente para la biodiversidad, la resiliencia y el funcionamiento ecológico de los medios costero y marino. El desarrollo de respuestas efectivas a esta amenaza multifacética es un desafío importante para los gestores y responsables de la toma de decisiones, debido a la complejidad de los ecosistemas marinos poco conocidos. La interpretación ecológica fidedigna de los datos espaciales utilizando técnicas de ecología del paisaje marino tiene el potencial de ser una poderosa herramienta de gestión. Esta disciplina emergente explora las causas y consecuencias de los patrones espaciales en el medio marino. En esta tesis, las técnicas de ecología del paisaje marino se utilizan para entender las consecuencias ecológicas de las perturbaciones antropogénicas sobre los patrones espaciales de los paisajes marinos mediterráneos costeros a través de tres estudios.

En el primer estudio se exploró la relación entre los patrones espaciales y la biodiversidad. La cuantificación de la estructura del paisaje marino mediante una métrica de patrones espaciales mostró que la riqueza de las especies se puede conservar mediante la protección de la diversidad de hábitats en el paisaje marino. Los resultados también pusieron de relieve la importancia de la de la complejidad en la distribución en cuanto a la riqueza de especies. Se determinaron los efectos de las presiones antropogénicas sobre la estructura del paisaje marino utilizando las métricas espaciales y el análisis multivariante.

Este enfoque permitió evaluar la influencia de las diferentes variables de perturbación sobre los componentes específicos del paisaje marino. Los resultados indican que la contaminación de origen terrestre está provocando la fragmentación del paisaje marino y tiene la mayor influencia en la composición del mosaico de hábitats. En el último estudio , se determinó el efecto de estas perturbaciones en los servicios que proporcionan los ecosistemas utilizando los stocks de carbono como indicadores, en particular en las praderas de fanerógamas. En análisis mostró la importancia del hábitat en el paisaje marino en cuanto a los servicios de los ecosistemas.

Concluimos que las técnicas de cuantificación del paisaje marino proporcionan información valiosa sobre las causas y consecuencias de los patrones espaciales de los ambientes mediterráneos costeros. La cuantificación de la estructura del paisaje marino con métricas de patrones espaciales es una técnica efectiva y consistente para la evaluación ecológica fidedigna de los datos espaciales a las escalas necesarias para la gestión. Los enfoques presentados en esta tesis son herramientas e instrumentos informativos valiosos para los planes de conservación.

Acknowledgements

I would like to extend my gratitude to the PEGASO Project for funding my PhD (EU FP7 PEGASO Project is funded under the call ENV.2009.2.2.1.4 Integrated Coastal Zone Management). I am also exceedingly grateful for the opportunity to participate in a FP 7 project and for all of the people I had the absolute pleasure of meeting and working alongside.

In terms of academic support, Bruce Talbot has been the fundamental force in the completion of this dissertation. His effort in reviewing chapters, tireless feedback on ideas and moral support is truly appreciated. Thank you for hosting me in your office at Skog og Landskap for three critical thesis-forming weeks. I am exceedingly grateful to Marcial Bardolet for the permission to use the Posidonia LIFE cartography and for providing me with the management plans, bathymetry and other data. Thank you so much to Gloria Salgado for translating the summary into Spanish and for your unbelievable support and kindness. Sharing an office with you was an absolute highlight of these three years. Luca Salvati was very helpful in developing the Principal Components Analysis approach used in chapter 4. Thank you also to Andreas Brunner and Jesse Kalwij for the feedback and sound-boarding.

My family has been an endless font of support, motivation and love during these past three years. To my mother, Leanne Nowell, your devotion to my cause is unrivalled! I can never thank you enough for all that you do. My father, David Nowell, sponsored my education and foundation in the fascinating field of conservation ecology and for that, I am not sure there are enough 'thank yous' in the world. Brandon and Tyrone go beyond the calls of brotherhood and have been the most wonderful sources of energy and enthusiasm on dark nights. Thank you very much to Simi for the memorable adventures along the way. I love that my grandmother, Heather Talbot, has had the end in mind since I began this challenge. She has never ceased to motivate me along the way. To Roy, Vanessa, Nards (& Frank ;-)), Bean, Jayne, Sue, my grandparents, Gavin, Heidi, Tim, Jarri, Helle, Christine, Budda and the rest of my amazing family – THANK YOU!

To my friends, my goodness, I am so grateful for your kind words, crazy words and downright hilarious words of support. Barbs, Wisey and Woof, you kids set the cool bar pretty damn high. Anellie, Brittness, Tina, Mopani and Cassie – total legends! Squeak (Captain Awesome), Van and Uncle Greg – what an honour to have embarked on a crazy cycle through Croatia with you chaps. I want to be like you when I grow up. Cinzy, ti ringrazio di tutto. Non potevo chiedere per una conquilina migliore. A special thank you to Lorenzo for protecting me from greedy kings. A huge thank you to my friends who have kept me sane with coffee and cava.

It takes a village.

Candidate profile

Megan Nowell completed her BSc and MSc degrees in Conservation Ecology at Stellenbosch University, South Africa. After completing her undergraduate degree, Megan worked at a NGO in southern India determining carbon sequestered by timber species. She also worked at the South African National Parks GIS-Hub as a GIS mapper for invasive vegetation. This introduction to the use of remote sensing and spatial analysis technologies in conservation planning, combined with her experience in ecosystem service quantification inspired her MSc thesis in the restoration of natural capital, entitled, 'Determining the hydrological benefits of clearing invasive alien vegetation on the Agulhas Plain, South Africa.'. This thesis allowed her to develop her skills in remote sensing, spatial analysis, GIS and hydrological modelling.

In 2011, Megan began her PhD at the Universitat Autònoma de Barcelona in the Environmental Science and Technology Institute (ICTA). She has worked hard to compensate for her lack of background in marine ecology, but her passion for the subject is undeniable. In addition to writing her PhD thesis, Megan contributed to the EU FP7 PEGASO Project with her reports on seascape ecology tools for ecosystem accounting and marine spatial planning. She hopes to continue developing her skills in seascape ecology, spatial analysis and conservation planning.

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CHAPTER 1

General introduction

"The living ocean drives planetary chemistry, governs climate and weather, and otherwise provides the cornerstone of the life-support system for all creatures on our planet, from deep-sea starfish to desert sagebush. That's why the ocean matters. If the sea is sick, we'll feel it. If it dies, we die. Our future and the state of the oceans are one." – Sylvia Earle, Sea Change: A message of the oceans, 1995.

Abstract

The heavily impacted Mediterranean coastline is home to some of the most productive and diverse ecosystems in the world. The complexity of these ecosystems, coupled with a limited understanding of their functioning and a lack of data at the scales needed for management, hinder their protection. The bottom-up approach of ecosystem-based management is gaining momentum as an effective method of protecting ecosystem functioning by conserving the mosaic of habitats in seascapes. This approach forms the basis of many marine policies, protocols and directives, however an understanding of the ecological consequences of the spatial patterns caused by anthropogenic activities is still missing. The emerging field of seascape ecology has the potential to bridge this knowledge gap using techniques developed for landscape ecology to explore the drivers of spatial patterns and their ecological consequences. In this dissertation, the application of landscape ecology techniques to address this knowledge gap and enhance the management of Mediterranean coastal seascapes is explored. The following chapter provides the background information and justification for this study. A summary of the landscape ecology concepts, principles and techniques with reference to seascape ecology is given within the context of the management responses to the heavily threatened Mediterranean Sea.

1.1. Context

Coastal and marine ecosystems are among the most diverse, productive and economically valuable in the world (Costanza et al. 1997; Suchanek 1994). These ecosystems provide essential ecosystem goods and services such as nutrient cycling, disturbance regulation, control of water quality, food production, habitat provision, erosion prevention and nursery grounds (Barbier 2012). Ecosystem services are increasingly recognized as vital to society and of significant economic value, however they are also severely threatened by disturbance and anthropogenic activities (Barbier et al. 2011; Daily et al. 1997).

With more than 67% of the global population residing within 60 km of the coast and an even greater number dependent on it for resources, the most significant threats to coastal and marine ecosystems result from human pressure (Barbier et al. 2008; Burkhard et al. 2010; Gray 1997). These threats include i) habitat degradation, fragmentation and destruction, ii) pollution, iii) over-fishing and the exploitation of marine resources, iv) invasive alien species introductions, and v) climate change (Burkhard et al. 2010; Gray 1997; Worm et al. 2006).

Anthropogenic pressures threaten the resilience of coastal ecosystems and their ability to deliver ecosystem services (Burkhard et al. 2010; Coll et al. 2010; Worm et al. 2006). The most serious threat to marine biodiversity is habitat loss, but degradation and fragmentation of habitats also leads to significant losses of species diversity (Gray 1997). Marine organisms, the functions they provide and the transfer of energy within marine ecosystems depend on a healthy mosaic of habitats.

Historically, the coast of the Mediterranean Sea has been one of the most densely populated regions on Earth (Airoldi et al. 2008). Urbanisation of the coastline is reaching a climax along the Europeans shores, while unprecedented population expansion and development is occurring along the southern and eastern coastline (Lejeusne et al. 2010). The pollution, exploitation of marine resources, overfishing, habitat degradation and invasive species introductions, in conjunction with the runaway coastal development make the Mediterranean one of the most heavily impacted seas in the world (Lejeusne et al. 2010). The Mediterranean basin is also host to an exceptionally high level of biodiversity and endemism that is significantly threatened (Médail & Quézel 1999; Pergent et al. 2012). Never has it been more critical to address the problem than now.

1.2. The ecosystem-based approach to management and conservation

Despite the Mediterranean Sea's long history of use, cooperative protection measures at the regional sea level have been implemented relatively recently. One of the most influential policies to this day is the Barcelona Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean Policy. This policy was built on the Mediterranean Action Plan (MAP), which was adopted in 1975 with the objectives of controlling pollution and achieving sustainable management and development in the coastal zone. Please see Appendix C for a more detailed description of Mediterranean Sea policies.

Since then, a number of additional policies and strategies have contributed to the protection of Mediterranean coastal and marine resources. These include the Habitats Directive, the Water Framework Directive (WFD)(2000), the Marine Strategy Framework Directive (MSFD)(2008) and the EU Biodiversity Strategy (2011). A common element in these policies is the ecosystem-based approach to management.

Ecosystem-based management (EBM) is gaining momentum as an effective approach to the conservation of biodiversity and ecological processes in marine and coastal management. The EBM approach seeks to reconcile conflicts between development and conservation by integrating ecological, social and economic sectors in decision-making (Barbier et al. 2008). From the ecological perspective, EBM focuses on the conservation of the ecosystem structure in order to protect the species, interactions and services provided by the ecosystem (Granek et al. 2010; Simberloff 1998).

Marine organisms are intrinsically linked to the structure of the seascape (Grober-Dunsmore et al. 2009). The spatial patterns of the seascape have been shown to influence the movement of energy, materials and organisms (Pickett & Cadenasso 1995). Ecological processes are dependent on these movements and interactions within the mosaic of patches in a seascape (Ernoult et al. 2003; Pickett & Cadenasso 1995). While EBM is based on the concept that protecting the mosaic will protect ecological processes and functioning, the lack of knowledge on the ecological consequences of spatial patterns in seascapes is a notable void in our understanding (Pittman et al. 2011). Furthermore, information on the effect of human activities on spatial patterning and ecological processes is of the utmost importance for effective management and planning of the coastal environment (Boström et al. 2011; Pittman et al. 2011; Wedding et al. 2011).

1.3. Landscape ecology

Landscape ecology explores the relationships between the spatial patterns of a landscape and ecological processes (Wedding et al. 2011). The central goals are to understand i) the spatial relationships among ecosystems or landscape elements, ii) the flow of energy, material, information and organisms in landscape mosaics, and iii) the ecological dynamics of the landscape over time (Turner et al. 2001; Wu 2013). The distinguishing features of this discipline are that it focuses primarily on the importance of the spatial configuration of landscape elements, and that typically, much larger spatial extents are studied than in other fields of ecology (Turner et al. 2001).

The discipline of landscape ecology originated in the 1960s when geography and holistic ecology were merged together with landscape management and planning, landscape architecture, economics and sociology (Turner et al. 2001; Wiens et al. 1993). It was not until the 1980s when technological advances made large scale spatial data available to ecologists and geographers that landscape ecology firmly took root as a field of study (Turner et al. 2001; Wu 2013).

Landscape ecology explores the dynamics of natural, semi-natural, agricultural and urban landscapes at multiple scales (Wu 2013). A landscape is defined as a "spatially heterogeneous area characterized by a mosaic of patches that differ in size, shape, contents and history" (Wu 2013). Two main perceptions of landscapes exist. In the first model, the landscape is perceived as a patch matrix, where certain focal patches exist within a matrix of ecologically neutral landscape elements (Wiens et al. 1993). For example, seagrass patches can be considered to be focal patches within a matrix of sediment (Boström et al. 2011). The patch matrix model is based on the theory of island biogeography, which was first proposed by MacArthur and Wilson in the 1960s. This theory explores the factors (e.g. patch size, degree of isolation and habitat suitability) that determine species richness on 'islands', as explained in detail by MacArthur and Wilson (1967). The patch matrix model has been the dominant perspective since it was first proposed largely because of its simplicity, however this may also be considered a disadvantage. The patch-matrix model has been criticised for over-simplifying how organisms interact with landscape patterns (McGarigal 2006). It is, nevertheless, an effective model for studying fragmentation (Boström et al. 2006; Boström et al. 2011; McGarigal 2006).

The second landscape model, the patch mosaic model, has recently emerged as a strong contender to the patch matrix model (Boström et al. 2011). This model views the landscape as a mosaic of different patch types whose composition and spatial configuration influence the ecological functioning of the entire landscape (Wiens et al. 1993). The context of focal patches within the mosaic is of utmost importance (Wedding et al. 2011). An advantage of the patch mosaic model is its realistic representation of the perception and interaction of organisms with their environment. The level of detail required to understand how these organisms interact and respond to landscape pattern is a drawback which has hindered the development the of model-specific methods used to quantify the spatial geometry of landscapes (McGarigal 2006).

Landscape spatial heterogeneity drives a range of ecological functions such as population structure, community composition, metapopulation dynamics and ecological processes (Pickett & Cadenasso 1995). These higher level processes, as well as organism perception and behaviour is directly related to the spatial structuring of the landscape at multiple levels (Johnson et al. 1992; McGarigal 2006). Landscape structure refers to the patterns of the landscape and is determined by the size, shape, arrangement and distribution of individual landscape elements (Walz 2011). Quantifying the spatial structure of a landscape can provide valuable information on the relationships between spatial patterns and ecological functioning and processes (Turner 1989; Uuemaa et al. 2009).

Spatial pattern metrics are used to describe landscape structure mathematically (Walz 2011). Spatial metrics are usually formulas or algorithms that are used to quantify the composition and configuration of the landscape (McGarigal 2006). The composition refers to the variety and abundance of patch types. Composition metrics include the i) proportional abundance of each class, ii) richness, iii) diversity, and iv) evenness. These metrics are only relevant at the landscape level. Spatial configuration, on the other hand, reflects the arrangement, position and orientation of patches and includes metrics representing i) patch area, ii) patch edge, iii) patch shape complexity, iv) core area, v) contrast, vi) aggregation, vii) subdivision, and viii) isolation. These metrics are normally calculated at the patch level (e.g. a seagrass meadow), but also provide information at the class level (e.g. all seagrass meadows in a specific area) and landscape level (e.g. all benthic habitats in a specific area). A detailed description of the metrics used in this study is given in Appendix A.

1.4. Seascape ecology

The marine counterpart to landscape ecology, seascape ecology, has only emerged in the last two decades (Boström et al. 2011). In the past, aquatic ecosystems were considered to be components of the broader landscape, rather than landscapes in their own right (Boström et al. 2011). Although a conceptual and analytical framework for seascape ecology has yet to be developed, the central concepts and analytical approaches of landscape ecology have been suggested to be equally applicable to aquatic benthic environments (Boström et al. 2011; Pittman et al. 2011; Wedding et al. 2011).

A seascape is defined as a "spatially heterogeneous area of coastal environment that can be perceived as a mosaic of patches, a spatial gradient, or some other geometric patterning quantified from either benthic or pelagic environments" (Boström et al. 2011). Marine organisms, like their terrestrial counterparts, utilise the mosaic of patches for foraging and movement, rarely showing a strong preference for a single habitat type (Boström et al. 2006; McGarigal 2006). From the perspective of the organism, habitat use is proportionate to the fitness it confers to the organism (McGarigal 2006). Movement of an organism through a mosaic of habitats is a function of the resistance of intervening habitats (McGarigal 2006).

A key difference between organism movement in marine and terrestrial mosaics, is that the marine realm has higher functional connectivity (Worm et al. 2006). This means that organisms have a greater capacity for dispersal and therefore habitat fragmentation in the seascape may affect marine organisms differently than it would species in terrestrial environments

(Meynecke et al. 2008; Montefalcone et al. 2010). While low levels of fragmentation may not be detrimental, the isolation of patches can reduce the connectivity of marine ecosystems significantly (Meynecke et al. 2008). A disruption of connectivity negatively impacts population dynamics, community structure and has been found to increase the susceptibility to invasion by alien species (Montefalcone et al. 2010).

As in landscape ecology, the structure of a seascape is determined by the size, shape, arrangement and distribution of individual patches and the spatial heterogeneity of the habitats present (Walz 2011). Since biodiversity is intrinsically linked to the habitats on which they depend for their existence, the structure of the seascape determines the occurrence and distribution of species from the local to the global level (Ernoult et al. 2003; Walz 2011).

The application of landscape ecology concepts and techniques to the seascape has been explored for coastal environments with particular success in shallow-water benthic ecosystems (Boström et al. 2011). Over the past 30 years, only 23 papers have been published on the quantification of seascape structure using spatial pattern metrics (Wedding et al. 2011). Of these studies, 7 explored seascape mosaics consisting of multiple habitat types, as I do in this dissertation. The majority of seascape studies assessed ecological patterns (Wedding et al. 2011). For example, fish assemblage attributes and density were predicted based on seascape structure by Pittman et al. (2007). Similarly, spatial metrics were used to link geomorphic coastal features to nearshore fisheries production (Meynecke et al. 2008). The fine-scale study by Garrabou et al. (1998) was the first application of landscape indices to benthic communities in the Mediterranean. In a subsequent study, Garrabou et al. (2002) used landscape pattern indices to explore the effect of depth on the structure of rocky benthic communities over a two year time period.

To my knowledge, no studies have used spatial pattern metrics to explore the effect of anthropogenic disturbance on seascape structure and the consequences on an ecosystem service at class level. The studies conducted in this dissertation seek to explore the relationships between spatial patterns and ecological processes in coastal Mediterranean seascape.

1.5. Goals and objectives

The objective of this study was to explore the relationships between the effects of anthropogenic pressures on spatial patterns in Mediterranean coastal seascapes and their ecological consequences using landscape ecology techniques. The aim is to assess the application of seascape ecology techniques to inform ecosystem-based management of disturbed coastal Mediterranean seascapes.

In order to achieve the objective, four goals needed to be met:

1. Firstly, the relevance of landscape ecology techniques for Mediterranean coastal seascapes was ascertained.
2. The second goal was to determine how anthropogenic disturbance affected seascape structure.
3. Understanding the relationship between ecosystem service delivery and seascape structure was the third goal.
4. The final goal was to determine the effect of disturbance on ecosystem service delivery.

1.6. Thesis structure

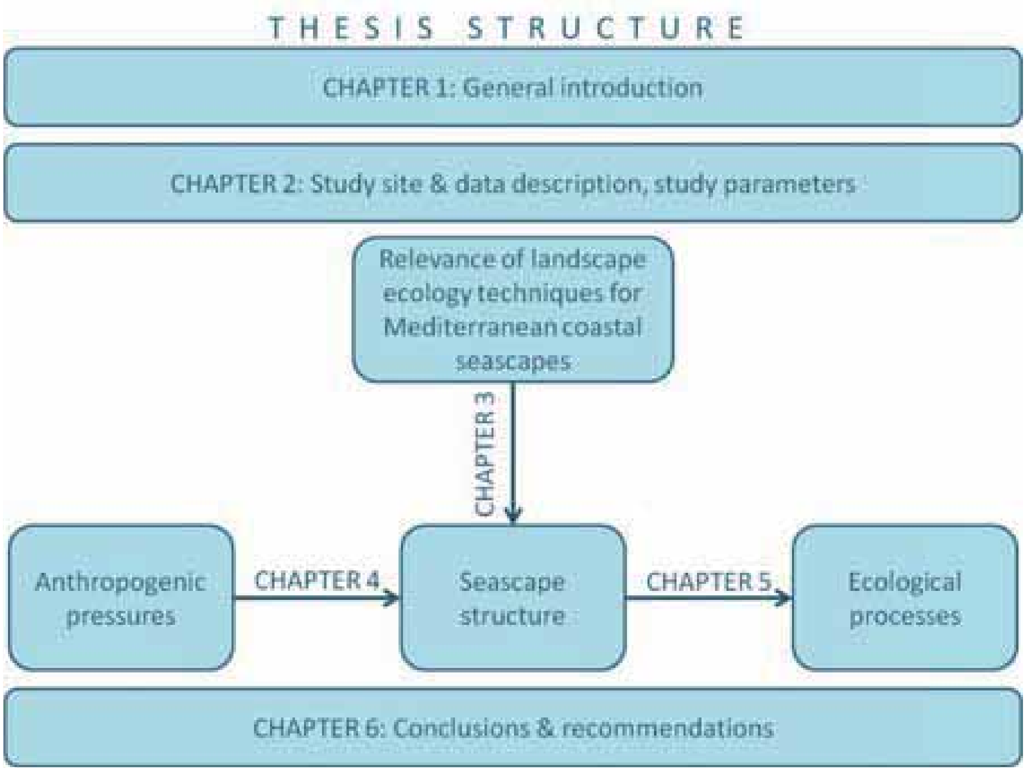

Figure 1.1: The thesis consists of 6 chapters, of which chapters 3-5 are presented as manuscripts for publication.

As illustrated in Figure 1.1, this dissertation is composed of six chapters. The first chapter provides the background information and justification for this study and is intended to give a general overview rather than a detailed introduction, which is included in each chapter.

Chapter 2 was dedicated to a detailed description of the seven study sites and the data used in this study. Information regarding the geography, threats and management of each site is given. The study parameters are also described. A site level comparison of the data is then performed. This descriptive analysis explores the composition of the seascapes in terms of habitat diversity and the configuration of patches. Understanding the composition and configuration of the seascapes is a necessary backdrop for the detecting patterns in seascape structure in the following chapters.

Chapter 3 is a fundamental chapter in that the landscape ecology techniques used in the other chapters are tested here. The application and ecological relevance of using spatial pattern metrics to quantify seascape structure and link it to biodiversity patterns is explored through multivariate analysis.

This technique is then used to identify how selected disturbance variables influence seascape structure in chapter 4. The effects of eight variables representing typical forms of disturbance on spatial patterns are determined using multivariate analysis techniques. I thank Dr. Luca Salvati for his invaluable help in designing a suitable statistical approach.

In chapter 5, the relationship between seascape structure and ecosystem service delivery is explored. The influence of anthropogenic disturbance on seagrass carbon stocks as an indicator ecosystem service was determined for coastal Mediterranean seascapes.

The final chapter discusses the main findings, management recommendations, global implications and possibilities for future research.

Chapters 3, 4 and 5 are laid out as manuscripts for publication. For this reason, a certain degree of repetition in the method section should be expected.

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CHAPTER 2

Study site and data description

Abstract

As an area of exceptional biodiversity and endemism that is also highly threatened, the Balearic Islands in the north western Mediterranean Sea have an elevated conservation value. Seven study sites in the Balearic Islands were chosen for the research conducted in this dissertation. This chapter provides a detailed description of the study sites, the parameters of the study and a descriptive analysis of the data.

2.1. Study site selection

The Mediterranean Sea is the largest and deepest enclosed basin on Earth (Coll et al. 2010). It is a marine biodiversity hotspot with more than 17 000 species identified (Coll et al. 2010). This biodiversity is generally concentrated in coastal zones with 90% of the plant species and 75% of the fish species between 0 and 50 m deep (RAC/SPA 2010). Around a quarter of the species found in the Mediterranean are endemic (Pergent et al. 2012). The high level of biodiversity is due to the geomorphology, environmental conditions and the complicated geological history of the basin (Boudouresque 2004; Lejeusne et al. 2010). This biodiversity tends to decrease from the north-western part of the Mediterranean to the south-eastern part (Coll et al. 2010).

The Mediterranean Sea has a negative water balance (i.e. the evaporation exceeds the inputs from rivers) and water from the less saline Atlantic Ocean, the Red Sea and the Black Sea flow into the basin (Boudouresque 2004). Evaporation is higher in the eastern half of the basin which results in a salinity gradient that increases from west to east (Coll et al. 2010). The semi-diurnal tides are small in amplitude (30-40 cm), with the exception of the Adriatic Sea and the Gulf of Gabès. The low quantity and nutrient-poor runoff from rivers results in the sea being oligotrophic, however enrichment may occur in coastal areas from currents, municipal sewage or temporal thermoclines (Coll et al. 2010).

Figure 2.1: The Mediterranean Sea is a biodiversity hotpot that is faced with increasing human pressures in the coastal zone. The Human Influence Index for Europe shows the highest scores in red and the lowest in blue (WCS 2005).

Study sites were chosen in the western Mediterranean because this area has the highest species richness and is also heavily threatened making it a hotspot of ecological importance (Coll et al. 2010). Spain's Balearic Islands were selected due to the presence of a mosaic of habitats representative of typical Mediterranean seascapes, the high conservation interest in the area and the availability of accurate and fine scale benthic habitat maps. As part of the

Baetic-Rifan complex, the Balearic Islands are also a hotspot within the Mediterranean Basin in that it is a highly threatened area with exceptionally high biodiversity and endemism (Médail & Quézel 1999). The location of the study sites made for an interesting comparative study between different disturbance intensities, diversity of habitats and protection levels. Seascapes in the coastal zone were chosen for two reasons. Firstly, the coastal zone has been identified as the ideal place to develop, apply and test spatial pattern metrics (Wedding et al. 2011), and secondly, the land-sea interface is where disturbance has the greatest effect (Harris 2011).

The Balearic Islands are located 175 km east of the Iberian Peninsula in the north-western Mediterranean Sea. The Balearic Islands are an autonomous community that consists of three main islands and a number of smaller islands. The main islands are home to around 1 million permanent inhabitants but this number can increase by as much as 300% in the peak season. Historically, the Balearic Islands relied on agriculture and cattle ranching as the primary economy, however a massive shift towards the tourism sector has occurred in recent decades (Morales-Nin et al. 2005). Tourism related pressures such as coastal development and the seasonal increase and exploitation of natural resources is a significant source of disturbance (Morales-Nin et al. 2005). Agricultural runoff, shipping (container ships, ferries, and recreational vessels), trawling and commercial fishing constitute other important forms of disturbance (Box et al. 2007; Diedrich et al. 2010). The area is also of interest for oil exploration.

Seven study sites were chosen (Figure 2.2) in the coastal zones of the islands of Mallorca, Ibiza and Formentera. The study sites are all sites of community importance (SCI), which affords them the same minimum level of protection (Table 2.1). Six of the seven sites are also protected for birds under the Habitats Directive (SPA), Costa de Llevant being the exception. Only two of the sites are IUCN protected areas. The Cabrera Archipelago is an IUCN Category II and Es Vedra is an IUCN Category IV protected area. As there was no variance between the SCI ranking, only the IUCN and SPA protection types were included as variables in the multivariate analysis.

Table 2.1: The protection status and the year it was issued are given for each study site.

Study site	SCI	SPA	IUCN	Year
Muntanyes d'Artà	✓	✓		2000
Cap de Barbaria	✓	✓		2006
Cabrera Archipelago	✓	✓	Cat. II	1991
Costa de Llevant	✓			2000
La Mola	✓	✓		2000
Llots de Ponent	✓	✓		2000
Es Vedra	✓	✓	Cat. IV	2000

Table 2.2: The presence/absence of the 26 habitat classes is given for the study sites.

Benthic habitat	Description	Habitat code	Arta	Barbaria	Cabrera	Levant	Mola	Ponent	Vedra
Rock	Rocky areas	1	x						
Sediment	Fine	2			x	x			
	Fine, well calibrated	3	x	x				x	
	Coarse, with current	4	x	x	x	x	x	x	x
	Poorly calibrated	5		x	x		x	x	x
Algae	Sciaphilic, dispersed	6			x				
	Sciaphilic	7			x			x	x
	Photophilic, dispersed	8		x	x				
	Photophilic	9		x	x	x	x	x	x
Coralligenous	Continuous	10			x				
	Dispersed	11			x			x	
Cymodocea nodosa	Continuous	12	x	x	x	x		x	
	Dispersed	13	x	x	x				
	Mixed, Caulerpa prolifera	14			x				
Detritus	Continuous	15	x	x	x	x	x	x	x
Posidonia oceanica	Continuous	16	x	x	x	x	x	x	x
	Isolated	17	x	x	x		x	x	x
	Sink holes	18	x		x				
	Degraded	19	x	x	x			x	
	Rocks	20	x	x	x		x	x	x
	Channels	21	x	x					
	Mixed, algae	22				x			
	Mixed, precoralligenous	23					x		
Precoralligenous	Continuous	24		x	x		x	x	x
	Dispersed	25			x		x		
Caulerpa prolifera	Continuous	26				x			
Count			12	14	20	8	10	13	9

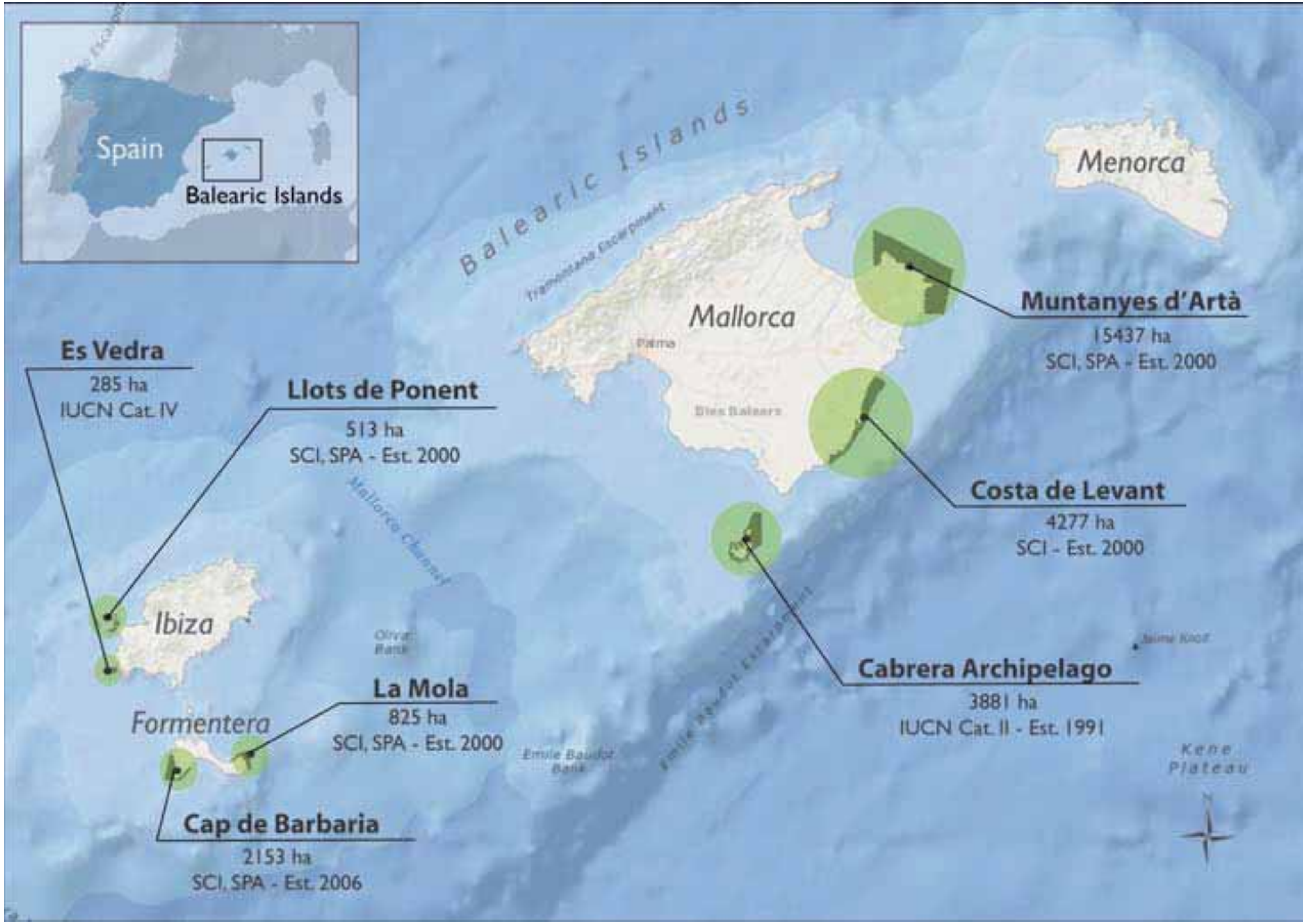


Figure 2.2: Seven study sites were chosen on the islands of Mallorca, Ibiza and Formentera.

2.2. Study site descriptions

2.2.1. Muntanyes d'Artà

The Muntanyes d'Artà seascape (Figure 2.3) is a Site of Community Importance (SCI) and a Specially Protected Area under the Birds Directive (SPA). The 15 437 ha area has been formally protected since 2000. Located at the north-eastern tip of Mallorca Island, the mountainous coastline takes the brunt of the Tramontana and Levante winds and is subject to currents. While very scenic, this coast has been relatively well preserved as a result of the few access points to the public.

The average depth of the seascape is 32.14 m, with a maximum depth of 50 m. The Muntanyes d'Artà seascape is a good example of marine coastal communities of the island of Mallorca with expansive *Posidonia oceanica* seagrass meadows, rocky areas rich in species and a number of underwater caves. The seagrass meadows are mostly established on rocky substrate where they compete with photophilic algae, but do also grow in sandier sediments where the currents are less intense. The result is a patchwork of sand and rock seagrass and photophilic algae, interspersed with sciaphilic algae in the more rugged terrain and in deeper areas. *Posidonia oceanica* is present up to depths of around 35 m which marks the transition from the littoral to the circalittoral zone. Detritus is abundant below this depth

The coastal population of 6148 inhabitants, increases significantly during the summer as most of the urbanized area consists of summer homes. The main human activities in the area are water sports. Anchoring and trawling is prohibited from Morro d'Aubarca to Cap des Freu due to the presence of four underwater cables that meet at Cala Mesquida. A commercial port and fishing harbour is located at Cala Ratjada in the south-eastern part of the SCI.

The management plan of the SCI includes proposals for regulating anchoring boats and assessing the impacts of invasive algae (*Womersleyella setacea, Lophocladia lallemandii and Caulerpa racemosa*) on the seagrass meadows. There is a need for impact studies and species specific management plans.

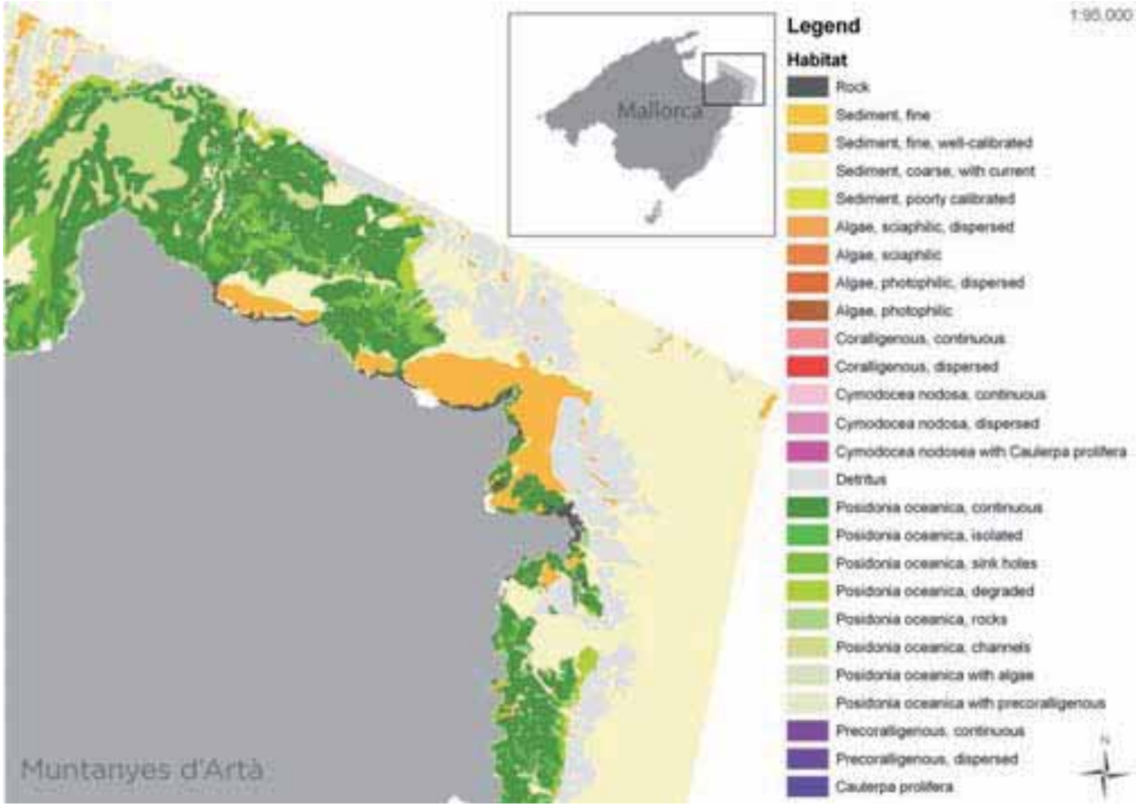


Figure 2.3: The Muntanyes d'Arta seascape is dominated by Posidonia oceanica *meadows.*

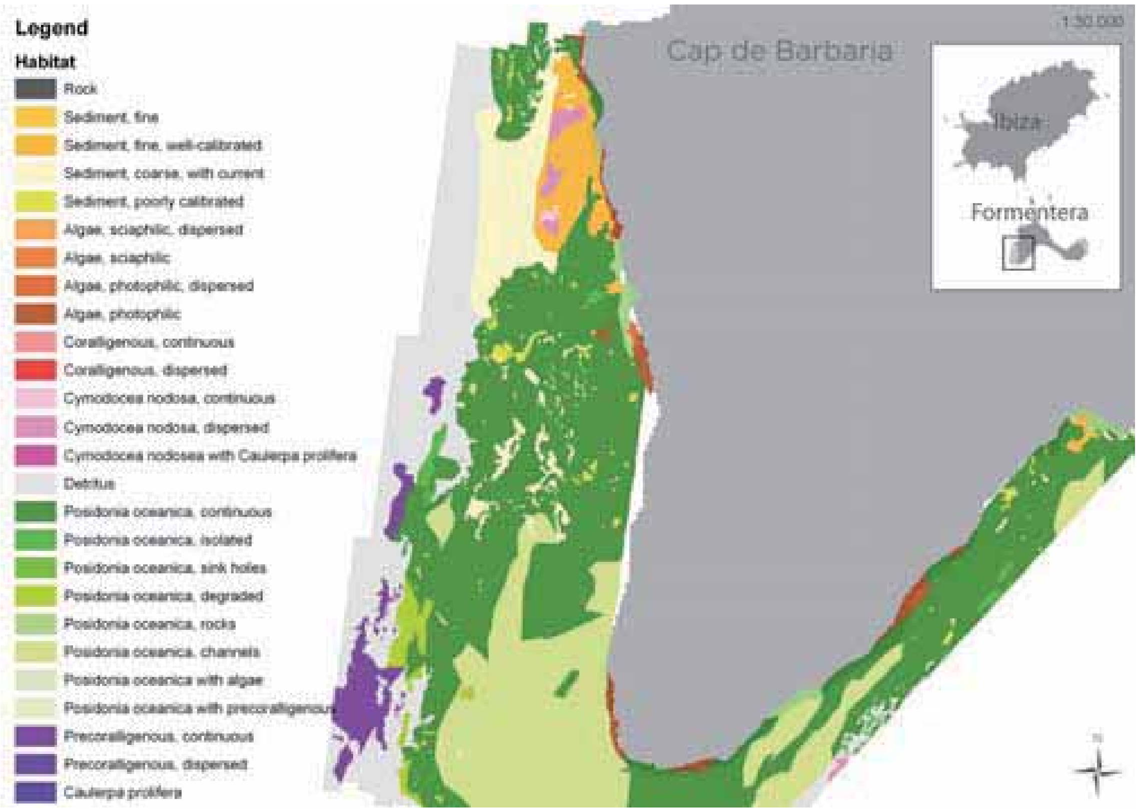


Figure 2.4: The Cap de Barbaria seascape consists of 15 types of habitats.

2.2.2. Cap de Barbaria

Located at the south-western tip of Formentera Island, Cap de Barbaria (Figure 2.4) was declared a SCI and SPA in 2006. Most of the coastline consists of steep cliffs, contrasting with the gently sloping morphology of the seabed. The average depth is 19 m with a maximum depth of 40 m. The 2153 ha seascape is home to large areas of sparse seagrass meadows and a number of patches of precoralligenous communities. The seagrass meadows are not very dense as a result of growing on bedrock and patches alternate with photophilic algae or rocky areas.

While Formentera has the lowest human pressure of the Balearic Islands, the resident population of the island (5377 inhabitants) can increase to as many as 16 500 people during the summer months. The Cap de Barbaria area is uninhabited and the nearest marina is La Savina in the north of the island. No significant human activities take place in this area and the benthic habitats are considered to be in a good condition. For this reason, no proposed actions are included in the management plan.

2.2.3. Cabrera Archipelago

The Cabrera Archipelago is situated off the southern tip of Mallorca (Figure 2.5) and consists of 19 small islands and islets covering around 10 000 hectares, of which nearly 9 000 hectares are marine environment. The average depth of the seascape is 32 m with areas reaching a maximum depth of 85 m.

Human activities have been limited around the archipelago since 1916 when it became a military zone. The Cabrera Archipelago was declared a National Park (IUCN Category II) in 1991 and a Specially Protected Area of Marine Importance in 2003 under the Barcelona Convention. The archipelago has been protected in order to preserve the large-scale ecological processes and diverse array of coastal and marine habitats. This area is rich in habitats with significant *Posidonia oceanica* and *Cymodocea nodosa* meadows as well as a number of important benthic habitats, including coralligenous and precoralligenous communities.

Damage from bottom trawling has been reported to the north and east of the archipelago resulting in a proposal to extend the national park (Oceana 2007; Oceana 2012).

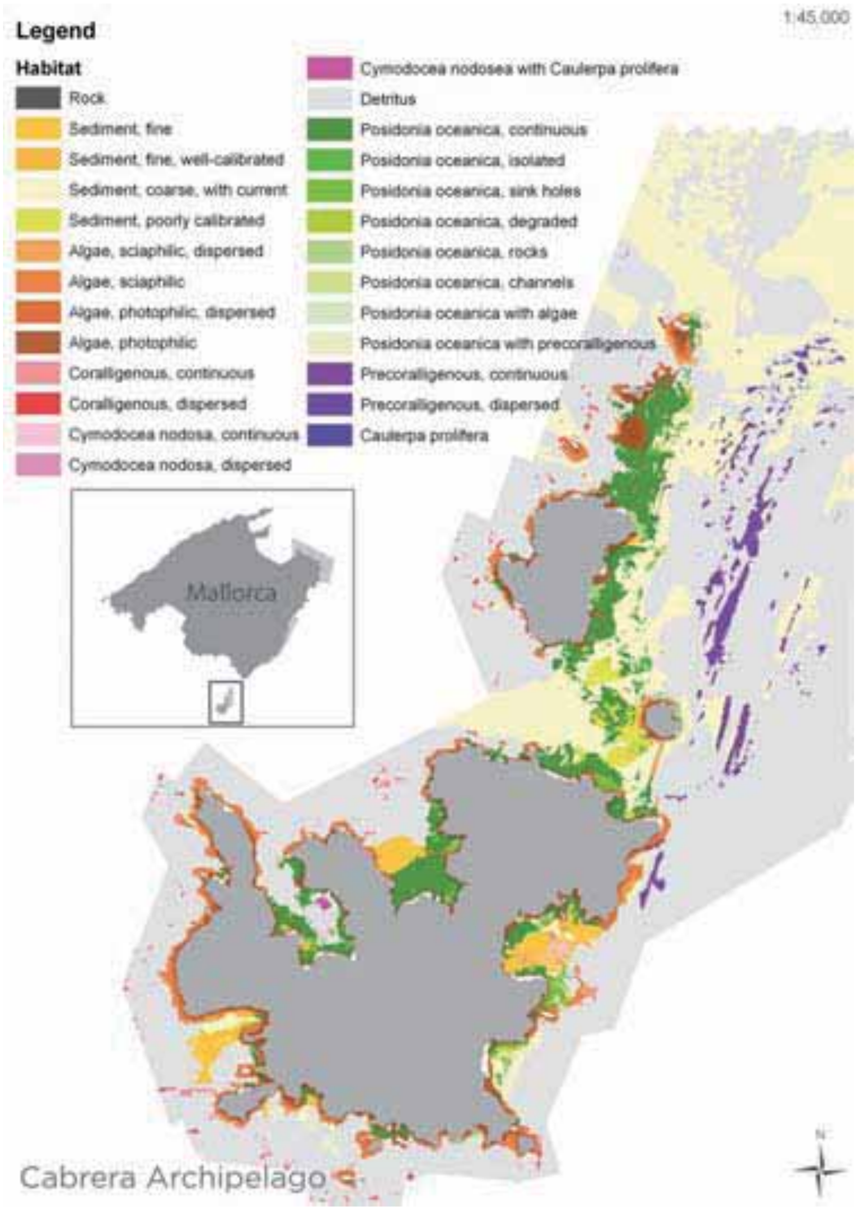


Figure 2.5: The Cabrera Archipelago is situated south of the island of Mallorca and has the highest habitat richness of the study sites.

2.2.4. Costa de Llevant

Costa de Llevant is a 4277 ha SCI on the east coast of Mallorca Island (Figure 2.6). The area has been formally protected since 2000 however it is one of the main tourist areas on the island of Mallorca. The coastline consists of moderately high cliffs with many small coves and beaches, often resulting from small streams and gullies.

The morphology of the seabed is generally steep until a depth of between 10 and 20 m, after which the slope is gentle. The area has a mean depth of 19 m and maximum of 40 m. Expansive meadows of *Posidonia oceanica* are present in the north of the SCI, however the area is not exceptionally rich in habitats. The seagrass meadows rarely exceed a depth of 30 m probably as a result of limited gravel and rock suitable for establishment. Trawling at these depths also degrades the meadows. The bay of Porto Colom is a sandy area entirely covered by *Caulerpa prolifera*, making it a prized fishing area.

In the last 50 years, the coastline has become heavily urbanized. The SCI lies along the coast of three municipalities, namely Manacor (35 908 inhabitants), Santanyi (10 523 inhabitants) and Felanitx (14 123 inhabitants). Four ports are located within close proximity of the SCI. Many water sports take place within the area, but the main threat to benthic habitats is from boat anchoring in the vicinity of coves and inhabited areas and trawling. Invasive species (*Lophocladia lallemandii, Caulerpa taxifolia and Womersleyella setacea*) also threaten habitats especially in the shallow rocky areas.

The management plan of the Costa de Llevant SCI proposes actions to increase regulatory measures to reduce the impact of boats and ban anchorage in several bays and coves. Surveillance of the invasive species present is also a priority. The existing fisheries legislation prohibits trawling below depths of 50 m, however the entire SCI is excluded and evidence of habitat degradation as a result of trawling is a concern.

2.2.5. La Mola

La Mola is situated on the eastern side of Formentera Island (Figure 2.7). The seascape is 825 ha and has been a SCI and SPA since 2000. The calcareous cliffs drop down to between 5 and 15 m under the sea surface, below which the rocky seabed has a gentle slope with an average depth of 22 m and a maximum depth of 50 m.

Posidonia oceanica meadows and precoralligenous communities are the dominant habitats. The seagrass meadows are limited to the rocky substrate with detritus abundant in the deeper areas of the SCI.

The lack of human settlements or ports within the SCI means that water sports and boating seldom occur. This area is considered pristine with very low human pressure, with the exception of fishing. The main threat is the invasion of exotic algae, such as *Lophocladia lallemandii, Acrothamnion preissii and Womersleyella setacea.*

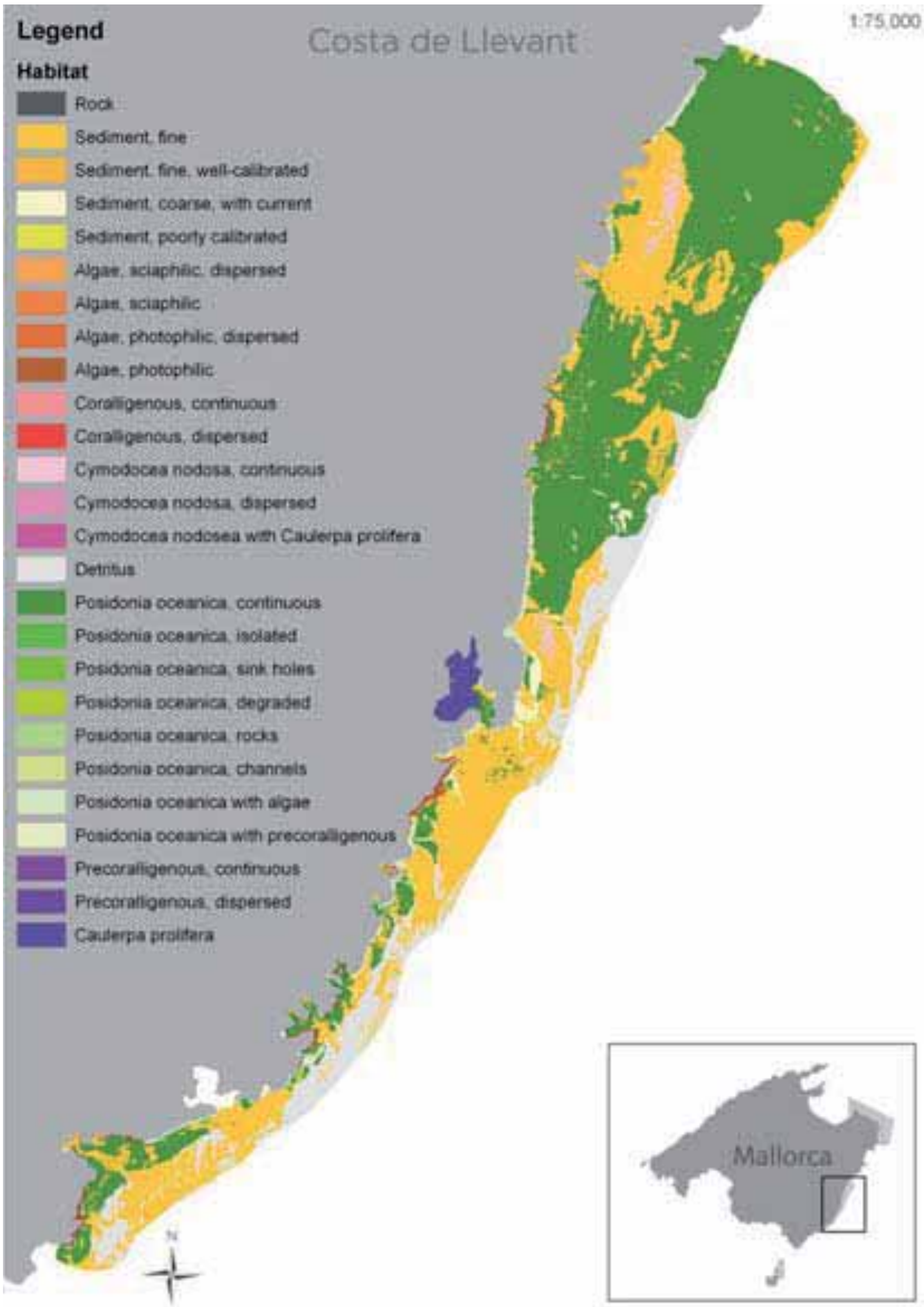


Figure 2.6: Costa de Llevant is a long, narrow seascape with large Posidonia oceanica *meadows in the north of the site.*

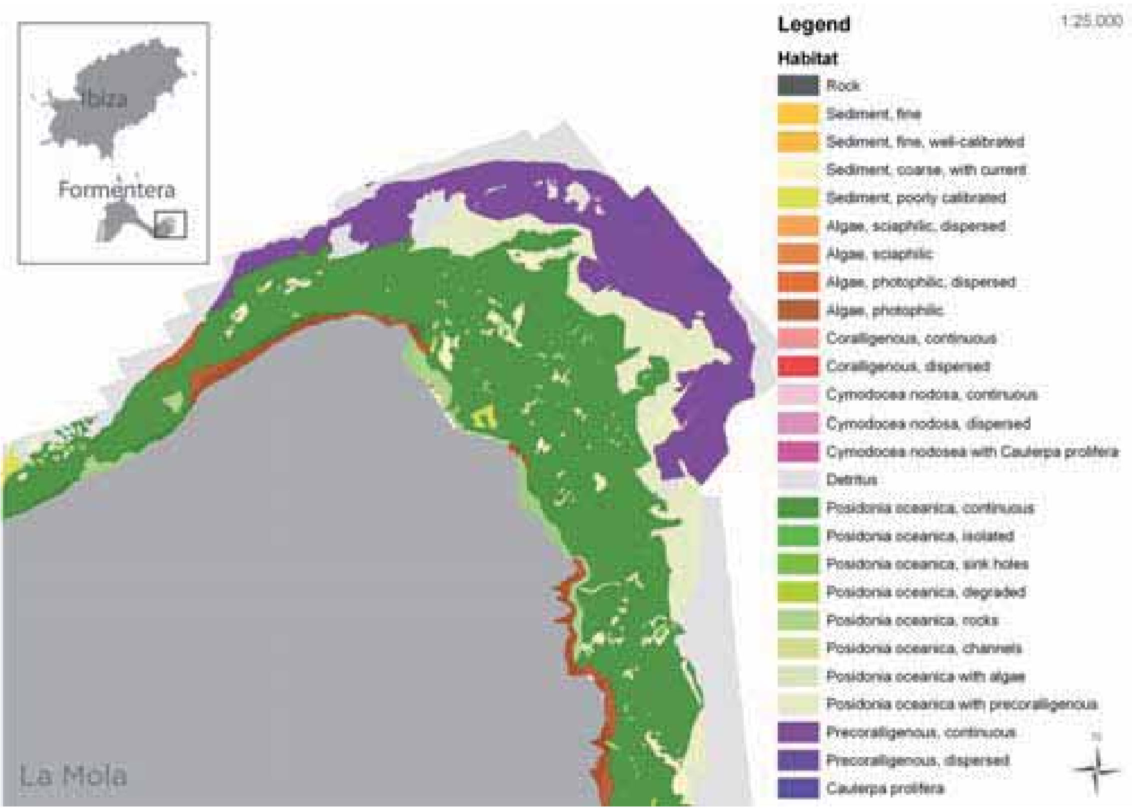


Figure 2.7: The seascape of La Mola contains a large patch of precoralligenous communities as well as continuous Posidonia oceanica meadows.

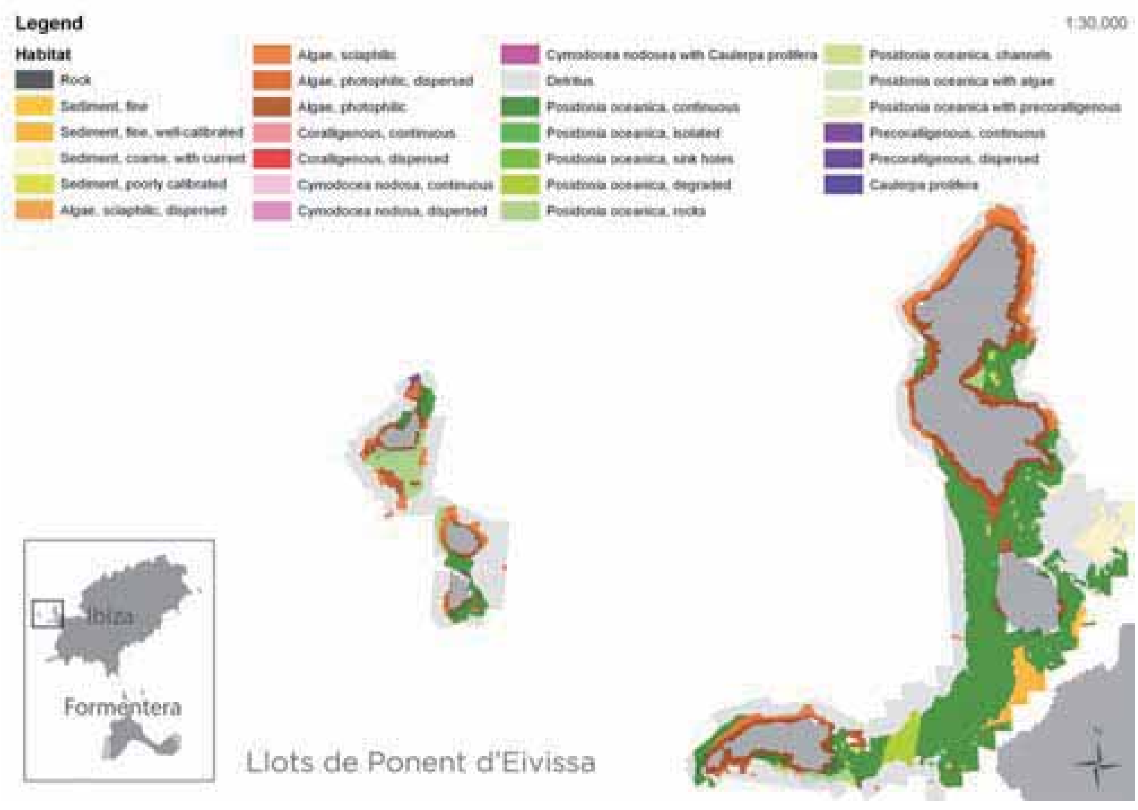


Figure 2.8: The Illots de Ponent seascape includes several Posidonia oceanica *patches. The islands are rimmed by algae.*

2.2.6. Illots de Ponent d'Eivissa

Llots de Ponent d'Eivissa consists of six uninhabited islands and a dozen smaller islets and reefs on the west coast of Ibiza (Figure 2.8). The area is a SCI and SPA that has been protected since 2000. The seascape is 513 ha in size and is relatively shallow with a mean depth of 13 m. The seabed is mostly sandy with boulders on a gentle slope. The deepest point within the study area is 40 m, however areas as deep as 80 m are present on the western side of the islands where the slope is steeper.

Seagrass meadows are sparse in area and are mostly present on the eastern coast of the islands. The SCI is rich in algae which surround most of the islands. The location of the islands in open water means that the habitats are exposed to strong currents.

The water quality is considered to be good although the pressure from human activities is moderately high due to the proximity to Sant Antoni de Portmany and commercial shipping routes. More than 16 000 inhabitants live in the area and this number can easily double during the summer months, increasing the human pressure on the SCI. Recreational boating is common in the area as are other water sports, including diving. The management plan for the SCI highlights assessing the impact of diving on gorgonians as a priority as well as monitoring the risks and evolution of invasive species in the area (*Lophocladia lallemandi*).

2.2.7. Es Vedrà and Es Vedranell

The islands of Es Vedrà and Es Vedranell are located on the west coast of the island of Ibiza and are uninhabited (Figure 2.9). A 285 hectare area surrounding the rocky outcrops was declared a SCI and SPA in 2006. The high number of terrestrial species that are endemic to the islands contributed to the designation of the site as an IUCN Category IV protected area.

The cliffs of the islands continue below the sea surface, dropping by as much as 40-50m on the southern coasts, which makes the bathymetric gradient particularly impressive. The deepest point of the seascape is 50 m with an average depth of 17 m. The good water quality and strong currents allow many rare suspension feeders to live on the rocky bottoms, alternating with seagrass meadows (above 30 m), gravel and detritus. Some boulders and sedimentary bottoms are present to the north of the island where the slope is not as steep and seagrass meadows can grow.

Despite not being inhabited, the islands have a moderate human pressure due to sport fishing and deep-sea diving. Assessing this pressure is the main priority identified in the management plan for the SCI.

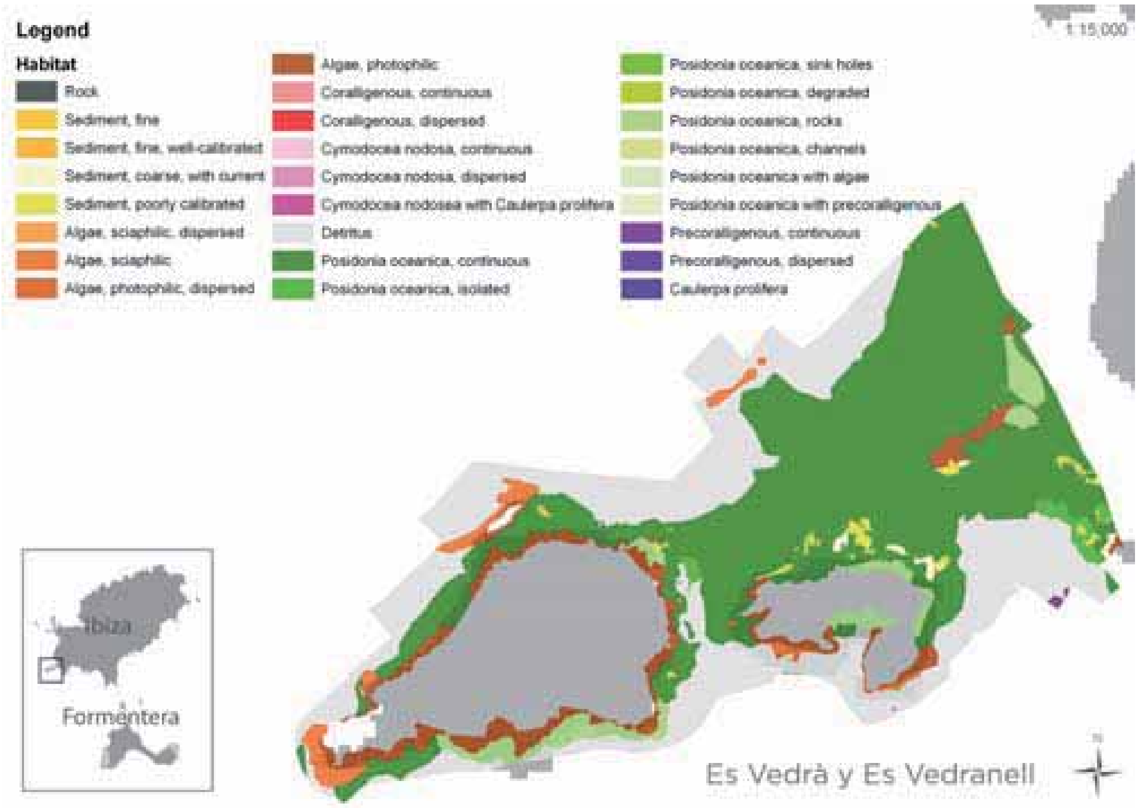


Figure 2.9: Es Vedra and Es Vedranell are flanked by continuous Posidonia oceanica *meadows on the northern sides of the islands where the slope is not as steep.*

2.3. Study parameters

2.4.1. Seascape boundaries

In this study, I use the definition that a seascape is a "spatially heterogeneous area of coastal environment that can be perceived as a mosaic of patches, a spatial gradient, or some other geometric patterning quantified from either benthic or pelagic environments" (Boström et al. 2011). This mosaic of habitats is potentially continuous. Defining the boundaries of the seascape is a subjective decision and depends on the phenomenon being studied (McGarigal 2006). In this case, the causes and consequences of spatial patterns in the context of management are explored. Therefore, the seascape limits are defined as the boundaries of the management unit (i.e. the boundaries of the Site of Community Importance) in combination with a depth limit of 40 m. Habitats below 40 m in depth could not be accurately classified using the side-scan sonar technique, but as the focal habitats for the study very rarely occur in depths greater than this, this depth limit did not hinder the study.

As management boundaries and side-scan sonar mapping produce linear boundaries, study sites were chosen where the intersection between focal habitats and the study site boundaries was minimal. An example is given in figure 2.10. Santa Eulalia was one of nine SCIs in the Balearic Islands that were excluded from the study because the perimeter of the focal habitats was delineated by the seascape boundary rather than the natural shape of the patch. SCIs were also excluded where the habitat classification system differed significantly.

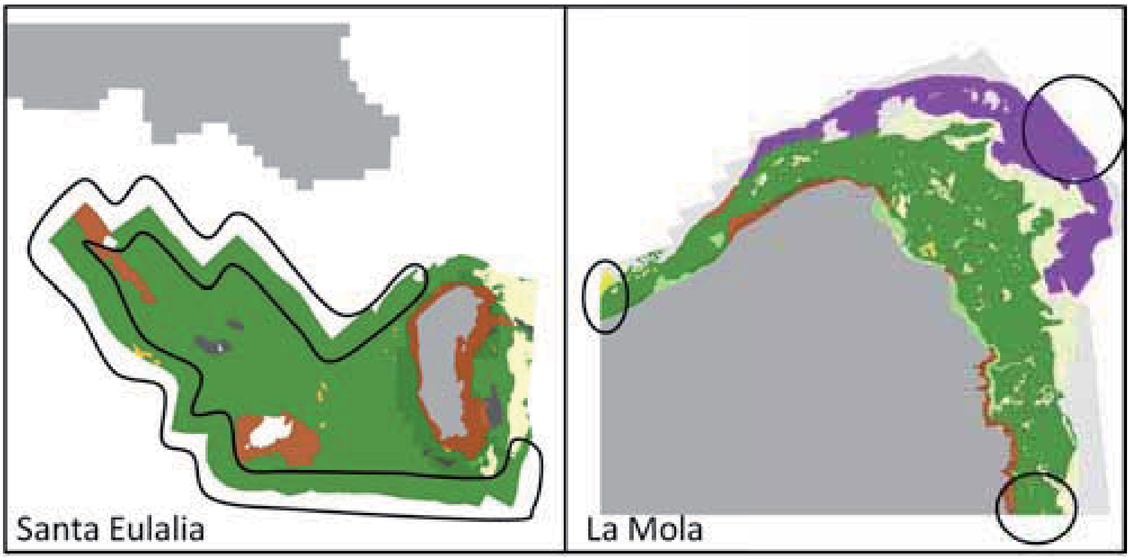


Figure 2.10: Two examples of seascape boundaries are given. The Santa Eulalia study site was excluded because most of the focal habitats (in green) intersected the boundary (highlighted in black). Study sites like La Mola were included where intersections were minimal.

2.4.2. Focal habitats and the seascape model

Two landscape models were discussed in chapter 1 (section 1.3). In the first, the landscape or in this case, seascape, is perceived as a patch-matrix. Focal habitats are set within a matrix of ecologically neutral habitats, such as sediment. While this model has been used to study fragmentation of seagrass meadows, it tends to over-simplify the seascape (Boström et al. 2006; McGarigal 2006). The 3-dimensional nature of the seascape means that species are able to use multiple habitats in their home range and therefore a broader approach that incorporates the mosaic of available habitats would be more applicable (Meynecke et al. 2008). Furthermore, studies have shown that different habitats are able to interact synergistically in ecological processes (Nagelkerken 2009). This ecological connectivity highlights the importance of habitat context (Grober-Dunsmore et al. 2007). To avoid excluding this level of complexity in the study, the patch-mosaic perspective was adopted. In this model the context of focal patches within the mosaic is of utmost importance (Wedding et al. 2011).

The focal habitats were selected based on two criteria. Firstly, the focal habitats were listed as ecologically significant for conservation. These habitats support a range of biodiversity and fundamental ecosystem processes. The second criterion was that the habitats should be representative of typical Mediterranean coastal ecosystems. The focal habitats used in this study were algae, *Cymodocea nodosa*, *Posidonia oceanica*, coralligenous communities, precoralligenous communities and *Caulerpa prolifera*.

2.4.3. Scale

Scale is one of the most important considerations in landscape and seascape ecology. Scale refers to "the spatial or temporal dimension of an object or a process" (Turner et al. 2001). Although inextricably linked and often confused, I distinguish between the scale of the study and the hierarchical level. The scale of a study is characterized by grain and extent (Turner et al. 2001). Grain size refers to the spatial resolution of the data. Patterns cannot be detected below the grain. The upper limit of the resolution of a study is set by the extent, which is the size of the study area (Wiens 1989). The grain and extent influence the response of spatial metrics and therewith the patterns and processes that can be identified (Schindler et al. 2013; Wiens 1989).

The hierarchical level, on the other hand, refers to the level of organization on an ecological level (Turner et al. 2001). An example would be the organisation of individual organisms into populations, which form communities that in turn make up ecosystems. There is no 'correct' scale for studies in landscape/seascape ecology, but the objectives of the study and the hierarchical level being studied guide scale selection (Turner et al. 2001; Wiens 1989). The seascape cartography used in this dissertation was produced at a scale of 1:1 000. This resolution was sufficient for observing patterns at the patch, class and seascape level. The class level was used as the focal hierarchical level, however spatial pattern metrics were calculated at all levels and compared statistically. The class level metrics performed the best and showed the most statistically significant relationships with the variables used in the chapters. This was especially true for the disturbance variables (mostly at a spatial resolution of 1 km^2) to which the class level metrics responded the best.

2.4. Data exploration

2.4.1. Geographical profiles

The western Mediterranean is relatively stable environmentally, both in terms of temperature and salinity below 200 m, however the seas around the Balearic Islands are distinct due to their proximity to the Algerian Basin, which is a reservoir for Atlantic waters. The Balearic Basin to the north of the islands experiences a circulation of water masses as a result of cold Mediterranean waters flowing from the north along the continental shelf-break and warm modified Atlantic waters flowing from the south (Moranta et al. 1998).

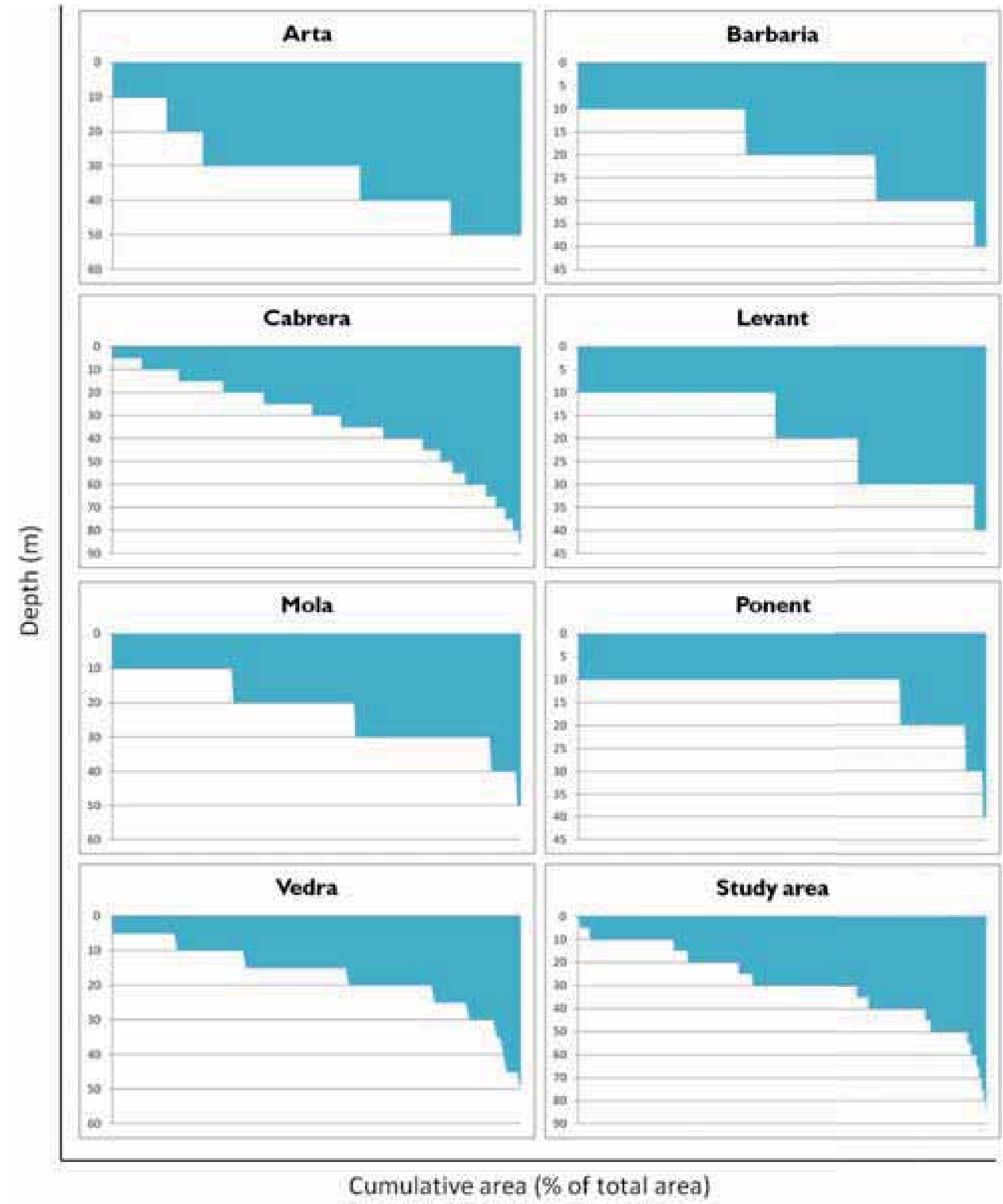


Figure 2.11: The depth profiles for the study sites are given with the depth (in metres) of the habitats on the y-axis and the cumulative area on the x-axis.

2.4.2. Habitat type and abundance

The Simpson's diversity index was used to compare the habitat richness, diversity and evenness at each of the study sites (Figure 2.12). The seascapes of Muntanyes d'Artà and Cap de Barbaria showed similar patterns of moderate habitat richness, but high diversity and evenness. Both sites have almost equal proportions of *Posidonia oceanica* meadows and sediment (Figure 2.13). Interestingly, the Cabrera Archipelago had the highest habitat richness, but also the lowest evenness due to detritus dominating the seascape. This means that there is a high diversity of habitats, but their relative abundance is low. The Costa de Llevant seascape is fairly evenly distributed in terms of habitat richness and diversity. *Posidonia oceanica* meadows are also abundant in this site. This is the only seascape that contains the invasive *Caulerpa prolifera* algae. La Mola also has an abundance of *Posidonia oceanica*, however the diversity index is higher as a result of more evenly distributed habitats. Precoralligenous communities are a notable component of this site. The seascapes of Llots de Ponent and Es Vedra may appear to be similar in composition in Figure 2.13, however Llots de Ponent shows a higher habitat richness in the Simpson's diversity index. Several under-represented habitats cause the low evenness score. The Es Vedra seascape is dominated by *Posidonia oceanica* meadows with a relatively low habitat richness, diversity and evenness.

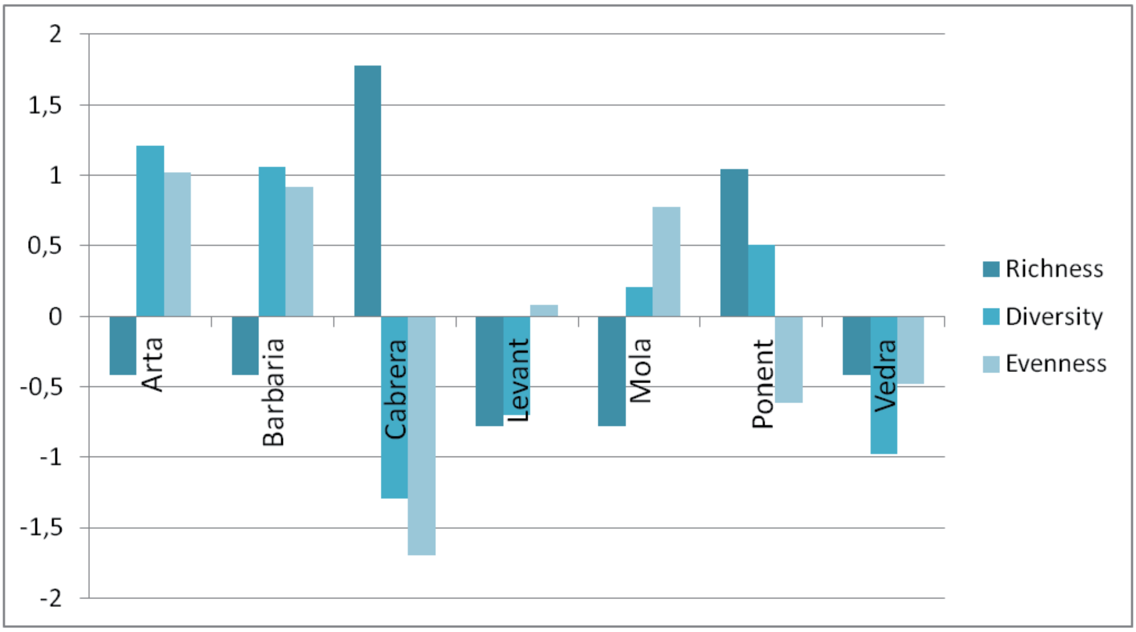


Figure 2.12: Simpson's diversity, richness and evenness indices were calculated for the types of habitats present in each study site.

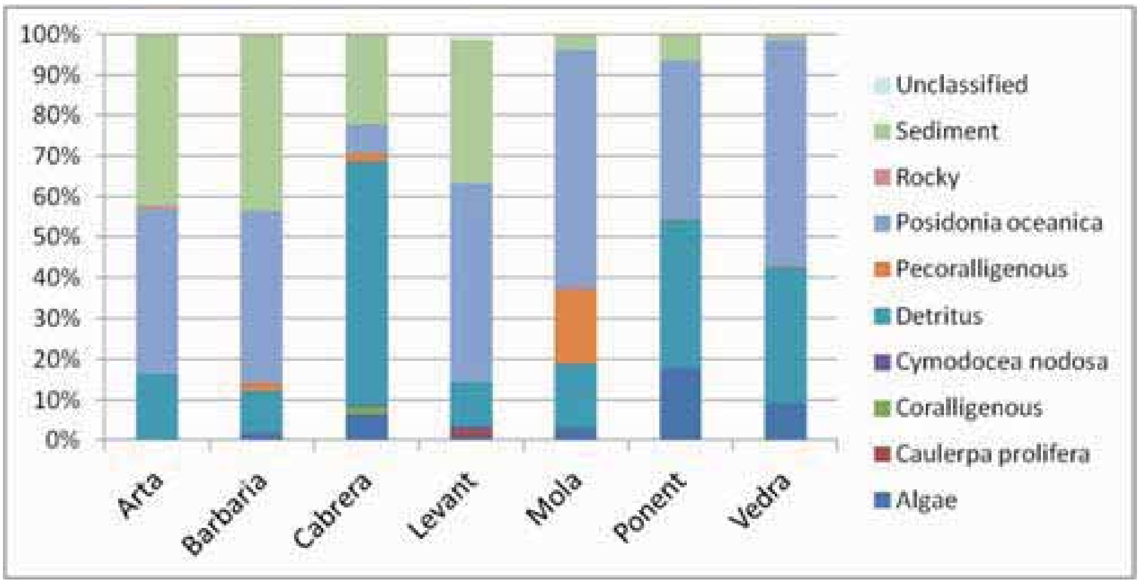


Figure 2.13: The relative abundance of the habitat types present in each study site is given.

2.4.3. Patch size and number

The initial data analysis showed a strongly positive skewed distribution of patch size. The skewness was 56.3 with a kurtosis of 3822.4. As these values are substantial, the dataset was divided into groups to enhance the detection of patterns. Table 2.3 shows that the number of patches decreases with increased patch size.

Table 2.3: The number of patches in each area group (m^2) is given for the study sites.

Area group (m^2)	Arta	Barbaria	Cabrera	Levant	Mola	Ponent	Vedra
<5000	3353	334	2240	1015	201	233	87
5000-25 000	707	39	219	129	28	49	23
25 000-50 000	146	13	43	35	9	5	2
50 000-100 000	129	3	32	20	5	11	4
100 000-200 000	40	3	14	19	1	6	1
200 000-500 000	44	6	12	9	2	4	2
500 000-1 000 000	17	1	7	3	3	0	0
>1 000 000	15	6	7	6	2	1	1
Total patches	4451	405	2574	1236	251	309	120

Patches smaller than 5000 m^2 represent 80% of the total observations. Histograms of patches within this first group showed that all study sites had a large number of very small patches and few larger patches (Figure 2.14). The distribution of patch size (<5000 m^2) can be explained by the Weibull function. The probability density function is given as:

$$f(x) = \frac{c}{b} \times [(x-q)/b]^{c-1} \times e^{\{-[(x-q)/b]\}^c}$$

Where, b is the scale parameter of the distribution, c is the shape parameter of the distribution, q is the location parameter of the distribution, and e is the base of the natural logarithm. The parameters of each Weibull distribution are given in Table 2.4. The Weibull function tends to slightly overestimate the patch size for very small patches, but as it intersects the middle of the first histogram, this is probably a result of class width selection and the function is considered a good fit for all study sites.

Table 2.4: The scale, shape and location (y-intercept) of the Weibull function if given for the density distribution (in red) in Figure 2.14.

Study site	Scale	Shape	Location
a) Muntanyes d'Artà	968,7706	0,7025	0
b) Cap de Barbaria	546,9324	0,6048	0
c) Cabrera Archipelago	543,2405	0,6281	0
d) Costa de Llevant	945,8875	0,9031	0
e) la Mola	605,2819	0,7006	0
f) Llots de Ponent	542,8189	0,5067	0
g) Es Vedra	832,5480	0,5767	0
h) All study sites	766,0190	0,6662	0

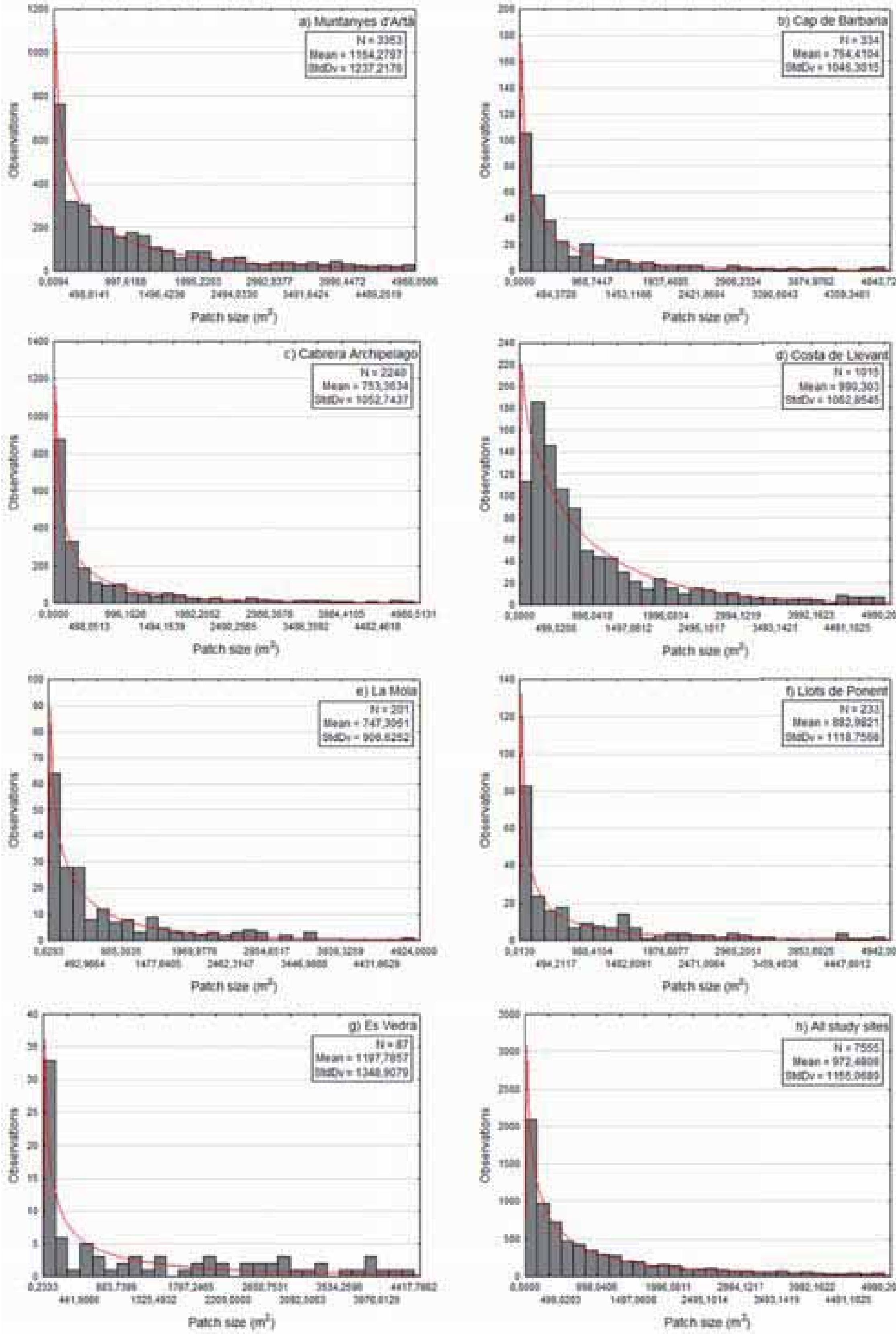
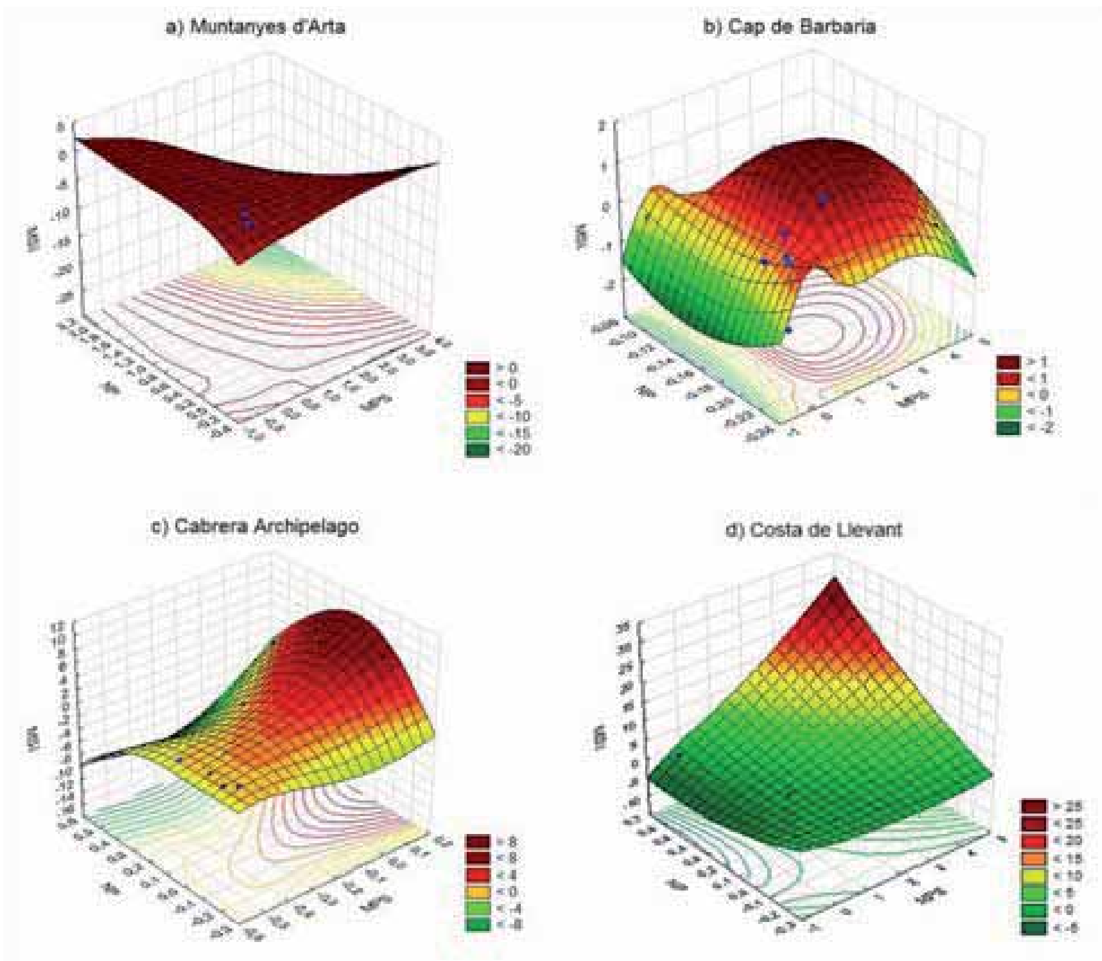


Figure 2.14: The histograms of patches smaller that 5000 m² show a strongly skewed distribution as a result of a large number of very small patches. The Weibull distribution function is given in red.

2.4.4. Spatial heterogeneity

Three metrics were used to describe the spatial heterogeneity of the seascapes, namely mean patch size (MPS), the number of patches (NP) and the mean shape index (MSI). The average size and quantity of patches describes the configuration of habitats within the seascape, while the shape index provides information on the complexity. The surface plct of the Muntanyes d'Artà seascape (Figure 2.15a) shows a high number of compact patches (low complexity) that are relatively large in size. Cap de Barbaria (Figure 2.15b) consists of a moderately well distributed seascape with shape complexity peaking for patches larger than the mean. The shape complexity is very low for many small patches in the Cabrera Archipelago (Figure 2.15c), but increases substantially with patch size. The highest levels of patch shape complexity were found in the Costa de Llevant study site. The surface plot shows many large patches with very high scores for complexity (Figure 2.15d). The seascape of La Mola shows the trend of shape complexity increasing with patch size (Figure 2.15e). There is however a significant dip with many average sized patches of very low complexity. The Llots de Ponent study site shows the opposite trend, with shape complexity decreasing with mean patch size (Figure 2.15f). Patch size, number and complexity are well distributed in the final study site, Es Vedra, showing a steady increase in complexity with mean patch size.



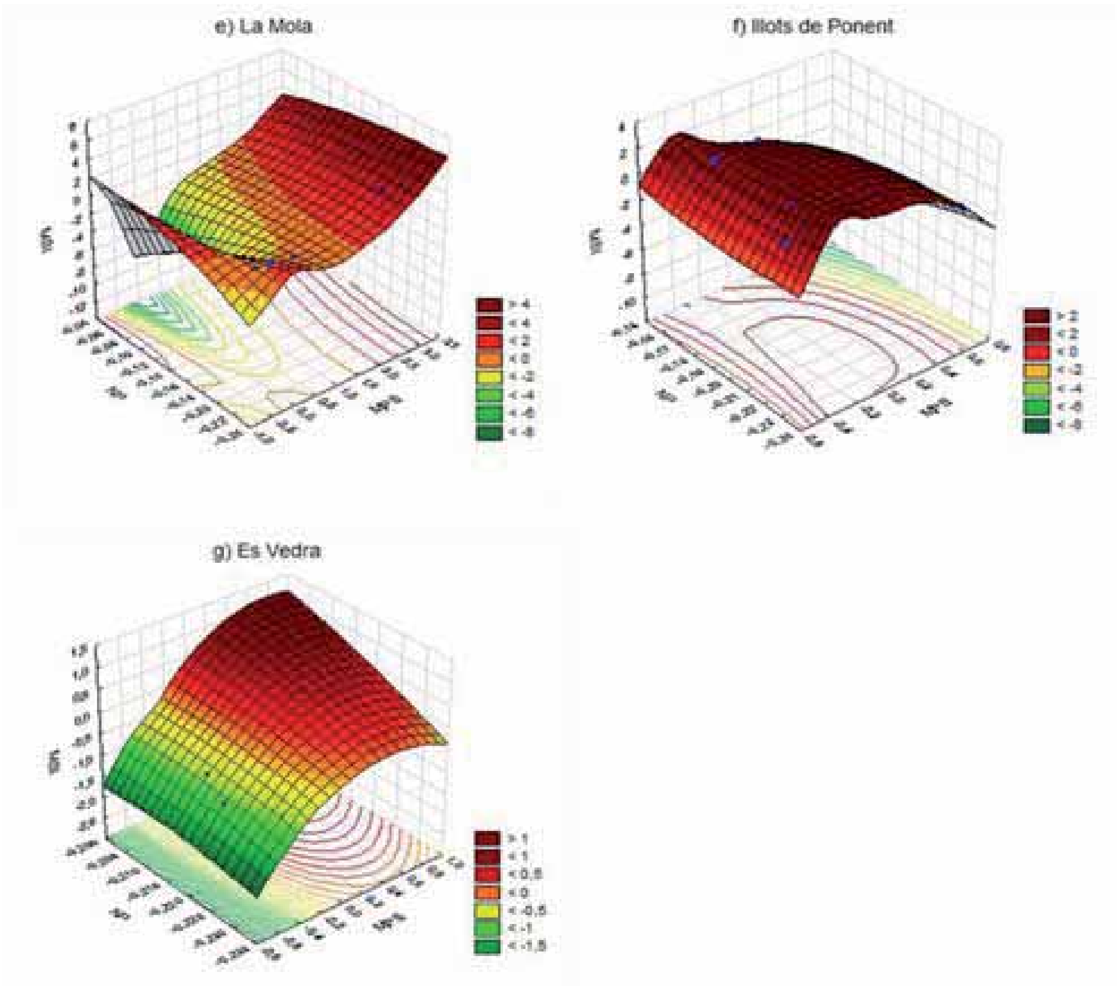


Figure 2.15: The 3D surface plots of the standardized mean shape index (MSI), number of patches (NP) and mean patch size (MPS) shows the variation in spatial heterogeneity between study sites.

2.5. Summary

The study sites chosen for this dissertation vary considerably in habitat diversity, richness and evenness, yet they share a similar pattern in how these habitats are arranged spatially. All seven study sites have very high numbers of small patches. The general trend is that these patches increase in shape complexity with increased size. The commonalities in the spatial structure of the seascapes make them comparable, while the differences in habitat composition, protection levels and disturbance highlight interesting questions that are explored in the following chapters.

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CHAPTER 3

Quantifying seascape structure

Abstract

Landscape ecology principles and techniques have recently been applied to seascapes to better understand the relationships between spatial patterns and ecological processes. Originally developed for terrestrial environments, the relevance of landscape ecology concepts, spatial pattern metrics and statistical methods has yet to be assessed for coastal and marine environments. In this chapter, the ability of spatial pattern metrics to provide relevant and meaningful information about the biodiversity of coastal Mediterranean seascapes was explored. The results indicate that the conservation of species richness requires the protection of the abundance of different types of habitats. The shape complexity of habitat patches also contributes to species richness. Quantifying the composition, configuration and complexity of the seascape using spatial metrics is a cost and time effective technique for evaluating large spatial datasets and providing information at a scale relevant for management.

3.1. Introduction

Coastal and marine ecosystems are among the most heavily threatened and poorly understood environments globally (Granek et al. 2010). This is especially true of the Mediterranean Sea, which is proportionally the most heavily impacted sea in the world and a globally recognised biodiversity hotspot (Coll et al. 2010). Ecosystem-based management seeks to reconcile the conflicts between human activities and conservation (Barbier et al. 2008). It is based on the concept that the complexity of interactions in marine ecosystems can be protected by conserving the mosaic of habitats (Granek et al. 2010). Since biodiversity is intrinsically linked to the habitats on which they depend for their existence, the structure of the seascape determines the occurrence and distribution of species from the local to the global level (Ernoult et al. 2003; Healey & Hovel 2004; Walz 2011). Seascape structure also plays a significant role in the ecological functioning and processes of ecosystems.

While the protection of seascape structure is essential for maintaining key ecological processes, information on how spatial patterns in the seascape influence biodiversity is lacking (Dauber et al. 2003). Understanding the ecological consequences of spatial patterns caused by human activities is of the utmost importance for effective management and planning of the coastal environment (Boström et al. 2011; Pittman et al. 2011; Wedding et al. 2011).

Landscape ecology concepts and techniques are increasingly applied to the coastal marine environment to better understand the causes and consequences of spatial patterning within the seascape (Boström et al. 2011; Wedding et al. 2011). This multidisciplinary field can be used to explore the links between spatial structure, ecological function and seascape change (Gustafson 1998). The structure of the seascape determines the movement of energy, matter, species and the functioning of ecological processes (Ernoult et al. 2003; Pickett & Cadenasso 1995). Marine organisms are intrinsically linked to the arrangement of the mosaic of habitats in seascapes (Grober-Dunsmore et al. 2009). The structure of a seascape is determined by the size, shape, arrangement and distribution of individual patches and the spatial heterogeneity of the habitats present (Walz 2011).

Seascape structure can be quantified using spatial pattern metrics. Spatial metrics are usually formulas or algorithms that are used to quantify: (a) the landscape/seascape composition (the abundance and diversity of habitats), (b) the spatial configuration (the spatial arrangement of habitat patches in the mosaic), and (c) the patch shape complexity (Wedding et al. 2011). While as of yet, no guidelines exist for utilising landscape metrics in the coastal and marine environment, the quantification of spatial patterns using landscape ecology approaches can provide seascape managers and decision-makers with a consistent method for monitoring changes and comparing seascape structure across temporal and spatial scales (Wedding et al. 2011).

Using landscape ecology techniques to better understand the relationship between spatial patterns and biodiversity could be a considerable benefit for the effective management coastal seascapes. The objective of this chapter is to test whether spatial pattern metrics can provide relevant and meaningful information about the composition of biodiversity in Mediterranean seascapes. Shallow-water coastal seascapes provide an opportunity to apply and test landscape metrics in the marine environment (Wedding et al. 2011). In this chapter I explore i) the influence of spatial patterns on habitat types, ii) the underlying factors driving seascape structure, iii) the correlations between specific aspects of seascape structure, habitat heterogeneity and species richness, and lastly iv) the spatial drivers of species richness are determined. As biodiversity is often used as an indicator of ecosystem health, exploring the applicability of spatial pattern metrics with reference to the consequences on biodiversity demonstrates the value of this tool in measuring changes in the health of coastal seascapes (Magurran 1988).

3.2. Method

3.2.1. Study site selection

Study sites on the island of Mallorca provide an interesting opportunity to test the application of seascape ecology in typical Mediterranean coastal ecosystems. The Balearic Islands have an elevated conservation value as an area of exceptional biodiversity and endemism that is also highly threatened (Médail & Quézel 1999). Not only was fine-scale benthic cartography available, but species inventories of the study sites provided an excellent opportunity to explore the consequence of spatial patterns on biodiversity. The majority of marine biodiversity occurs in the benthic zone, adding to the value of studying the spatial patterns of benthic habitats (Angel 1993). Coastal ecosystems are also suitable areas to test spatial pattern metrics in the marine environment because maps are usually of better quality and accuracy as a result of the relative ease of mapping in comparison with deep-sea environments (Wedding et al. 2011).

Three coastal study sites were selected on Mallorca (Figure 3.1). Muntanyes d'Artà is a 15 437 ha Site of Community Importance (SCI) and a Specially Protected Area for Birds (SPA) located at the north-eastern tip of the island. The seascape is a patchwork of sand and rock seagrass and photophilic algae, interspersed with sciaphilic algae in the more rugged terrain and in deeper areas. The Costa de Llevant study site is, as its name suggests, located on the eastern coast of Mallorca. It is a SCI 4 277 ha in size and is a major tourist area. Expansive meadows of *Posidonia oceanica* are present in the north of the SCI, however the area is not exceptionally rich in habitats, probably as a result of trawling. The bay of Porto Colom is a sandy area entirely covered by *Caulerpa prolifera*, making it a prized fishing area. The Cabrera Archipelago is a group of 19 small islands and islets covering around 10 000 hectares, of which nearly 9 000 hectares are marine environment. Located at the southern tip of Mallorca, the highly diverse seascape was declared a National Park (IUCN Category II) in 1991 and a Specially Protected Area of Marine Importance in 2003.

3.2.2. Benthic habitat data collection and processing

Detailed seascape maps of the study sites were obtained from the Posidonia LIFE project. The benthic habitats were mapped at a scale of 1:1,000 using a side-scan sonar technique for areas between 5 and 40 m deep and orthophotos for areas at depths between 0 and 5 m. The maps were downloaded from the Posidonia LIFE website (http://lifeposidonia.caib.es), converted from kml files to shapefiles and georeferenced. The shapefiles were projected using Transverse Mercator with WGS 84 as the coordinate system. The original data contained a total of 86 habitat classes for the eight study sites. Duplications and variations in the naming of habitats were removed leaving 26 unique benthic habitat classes (Table 3.1). The original names are available in Appendix B.

3.2.3. Habitat heterogeneity

Diversity is a central theme in ecology as it indicates the health of an ecological system (Magurran 1988). Measurements of diversity are based on the variety of organisms or habitat and their relative abundance. The Simpson's diversity indices were used to calculate habitat richness (R), diversity (D) and evenness (E). To avoid confusion, I refer to these three metrics as describing habitat heterogeneity.

Habitat richness is the number of benthic habitat types present in the study site (n). The diversity of the habitats reflects the abundance of each habitat type within the seascape. It was calculated as:

$$D = \frac{1}{\Sigma\left(\left({}^{CA}/_{TA}\right)^2\right)}$$

Where D is the habitat diversity, CA is the class area (m^2) and TA is the total area of the seascape. Simpson's D measures the probability that any two randomly selected habitat patches will be of the same type. Evenness described the relative abundance of habitat types and was calculated as D/n.

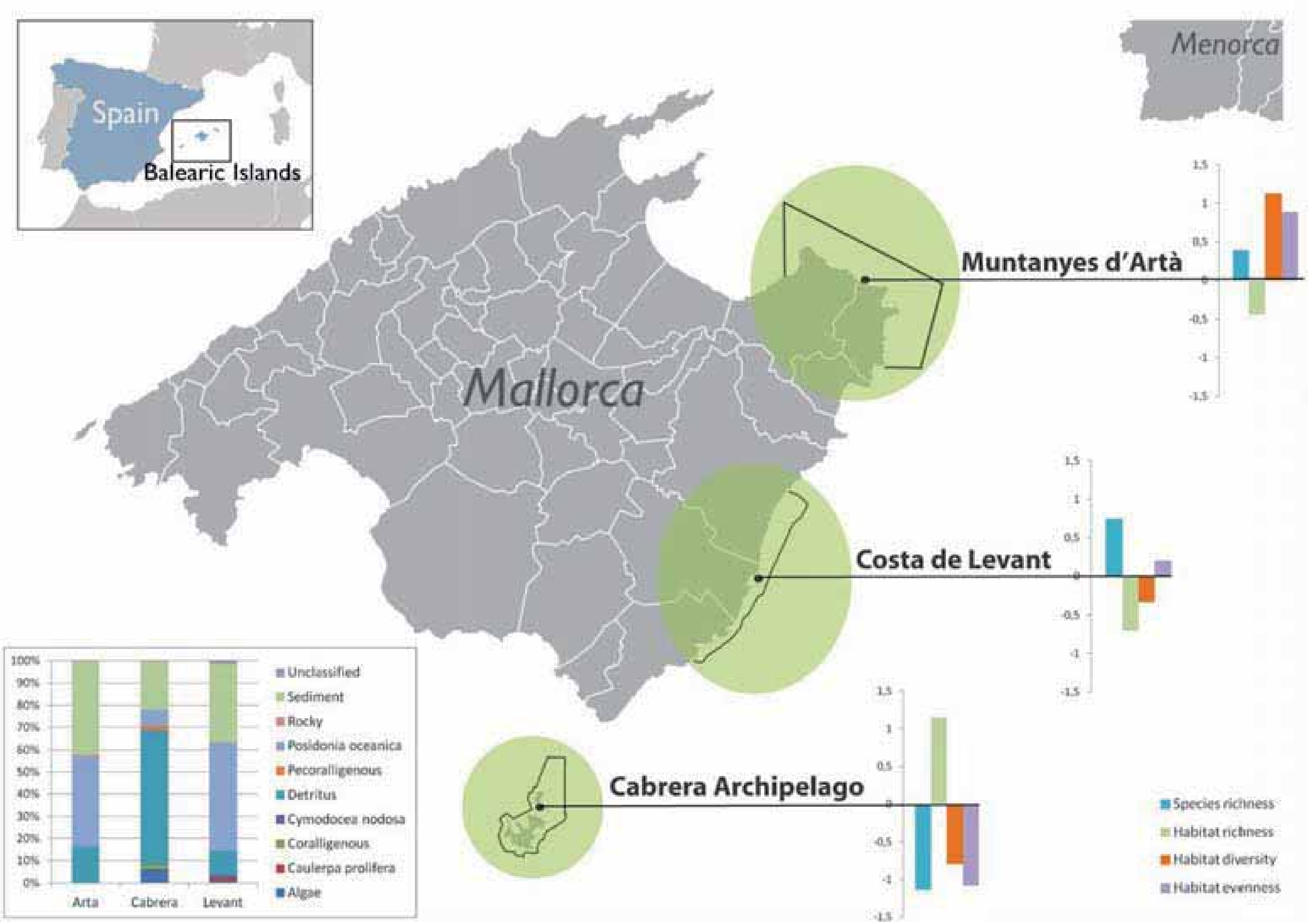

Figure 3.1: Three study sites on the island of Mallorca were used to explore the relationships between seascape structure, species richness and habitat diversity.

Table 3.1: The types of habitats present in the study sites is given with the habitat codes used in the data analysis.

Benthic habitat	Description	Habitat code	Arta (A)	Cabrera (C)	Levant (L)
Rock	Rocky areas	1	x		
Sediment	Fine	2		x	x
	Fine, well calibrated	3	x		
	Coarse, with current	4	x	x	x
	Poorly calibrated	5		x	
Algae	Sciaphilic, dispersed	6		x	
	Sciaphilic	7		x	
	Photophilic, dispersed	8		x	
	Photophilic	9		x	x
Coralligenous	Continuous	10		x	
	Dispersed	11		x	
Cymodocea nodosa	Continuous	12	x	x	x
	Dispersed	13	x	x	
	Mixed, Caulerpa prolifera	14		x	
Detritus	Continuous	15	x	x	x
Posidonia oceanica	Continuous	16	x	x	x
	Isolated	17	x	x	
	Sink holes	18	x	x	
	Degraded	19	x	x	
	Rocks	20	x	x	
	Channels	21	x		
	Mixed, algae	22			x
	Mixed, precoralligenous	23			
Precoralligenous	Continuous	24		x	
	Dispersed	25		x	
Caulerpa prolifera	Continuous	26			x
Count			12	20	8

3.2.4. Species richness

Species lists were compiled by the CEAB-CSIC and included in the management plans of the study sites. The species were grouped into phyla and included in the analysis to determine the relationship between seascape structure and species richness. The number of species in each phyla present in the study sites is given in Table 3.2. Full species lists are available in Appendix D.

Table 3.2: The number of species in each phylum is given for the study sites.

Phylum	Cabrera	Arta	Levant
Cyanobacteria	5	5	7
Green algae	20	21	23
Brown algae	35	43	30
Red algae	91	77	99
Phanerogams	2	2	2
Lichens	1	2	1
Foraminifera	1	1	1
Sponges	54	88	38
Cnidaria	22	27	43
Polychaete annelids	7	7	35
Echiura	1	1	1
Sipunculids	0	0	1
Molluscs	17	35	50
Bryozoans	19	68	50
Brachiopods	0	2	0
Arthropods	6	43	37
Echinoderms	13	8	26
Ascidians	0	0	15
Tunicates	17	13	0
Fish	82	87	103
Total	393	530	562

3.2.5. Spatial pattern metrics

Overlaying and intersecting large amounts of spatial data in order to calculate spatial pattern metrics requires GIS technologies (Walz 2011). Spatial pattern metrics were calculated at class level for the 26 habitat types present in the three study sites. The ^{V}LATE 1.1 (Tiede 2005) vector-based extension was used to calculate the spatial metrics in ArcGIS 9.3 (ESRI 2009). A total of 14 spatial metrics were extracted that quantify area, form, edge, diversity, subdivision and proximity (Table 3.3). The detailed descriptions and equations used to calculate the spatial metrics are given in Appendix A.

Table 3.3: A total of 14 spatial pattern metrics were calculated using vector-based software. The metrics provide information at patch (p), class (c) and seascape (s) level.

	Abbreviation	Spatial pattern metric	Level
Area	NP	Number of patches	c, s
	CA	Class area	p, c, s
	MPS	Mean patch size	c, s
	PSSD	Stand deviation of patch size	c, s
Form	MSI	Mean shape index	p, c, s
	MPAR	Mean perimeter to area ratio	p, c, s
	MFRACT	Mean fractal dimension	p, c, s
	TE	Total edge	c, s
	MPE	Mean patch edge	p, c, s
Diversity	PROP	Proportion	s
Subdivision	DIV	Division	c
	SPLIT	Splitting index	c
	MESH	Mesh	c
Proximity	MNN	Mean Nearest Neighbour	c

3.2.6. Data analysis

A number of statistical methods were used to explore the relationships between seascape structure and biodiversity. The statistical analyses were all performed using STATISTICA 8.0 statistical analysis software (StatSoft 2007). To answer the question of how the aspects of seascape structure (i.e. spatial pattern metrics) influence specific habitat types, a two-way joining cluster analysis was used. This is a technique that performs a cluster analysis on both the variables and the cases and was used to identify interesting relationships that would otherwise not be revealed in conventional clustering algorithms.

A Principal Components Analysis (PCA) was performed on the standardized data in order to identify any underlying, uncorrelated factors that influence seascape structure. Because PCA is an orthogonal transformation based on variance, class level metrics were used rather than seascape level metrics to increase variance in the analysis. Factors with an eigenvalue >1 were extracted and the variables and cases plotted on the factor plane to explore the correlations with the PCA axes. The spatial pattern metrics were projected as active variables with the Simpson's diversity indices and species richness data as supplementary variables.

The Spearman's rank correlation coefficient was then calculated to test the statistical relationship between the spatial pattern metrics and the variables describing habitat heterogeneity (i.e. Simpson's diversity indices) and species richness. A significance level of $p<0.01$ was used to identify statistically significant correlations among the data.

Finally, a stepwise multiple regression analysis was performed to determine if, and which seascape predictors were driving species richness. This technique was used to eliminate variables that did not contribute significantly to species richness, but whose interactions may confound results. Variables were grouped based on ecologically relevant drivers of species richness. The results of the PCA were used to group the spatial metrics into predictors of seascape composition (F1) and configuration (F2). The following predictors were included as the independent variables:

i. Factor 1 (F1): Factor 1 describes seascape composition and is composed of the number of patches (NP), class area (CA), total edge (TE), and the mesh metric (MESH).
ii. Factor 2 (F2): Factor 2 consists of the mean patch size (MPS), fractal dimension (MFRACT), mean patch edge (MPE), and the division (DIV) and splitting index (SPLIT). These metrics describe seascape configuration.
iii. Connectivity: The mean nearest neighbour (MNN) metric was used as an indicator of connectivity between patches of the same habitat type.
iv. Complexity: The complexity of the seascape also emerged as an important factor in the PCA. The predictor of complexity consisted of the mean shape index (MSI), mean perimeter:area ration (MPAR) and the fractal dimension (MFRACT).
v. Depth: The study by Garrabou et al. (Garrabou et al. 2002) showed that community structure and dynamics respond to the depth of patches. The average depth of the habitat class was used as the predictor.
vi. Habitat diversity (D): The diversity of the habitat takes into account the variety of habitats and their relative abundance. This index is often used as an indicator of ecosystem health.
vii. Habitat richness (R): Habitat richness is the variety of habitats available to species.
viii. Habitat evenness (E): Habitat evenness represents the relative abundance of habitat types.

The multiple regression was based on the assumptions that the data was normally distributed, that linear relationships exist between species richness and the predictors, and that variables are measured with minimal error. The residuals were tested for homoscedasticity.

3.3. Results

4.2.1. Seascape structure and habitat type

The two-way joining cluster analysis (Figure 3.2) identified which aspects of seascape structure were important for different habitat types. The results showed that the strongest positive relationships were in the Muntanyes d'Artà study site. The number of patches (NP), class area (CA), proportion, total edge (TE) and mesh metrics had a strong influence on coarse sediment (A4). These same metrics also influenced continuous *Posidonia oceanica* patches (A16). The same pattern was reflected for continuous *Posidonia oceanica* patches in the Costa de Llevant study site. Very large patches of *Posidonia oceanica* are present in both sites, which the cluster analysis may be responding to. Detritus, also forming large patches in Muntanyes d'Artà (A15) and the Cabrera Archipelago (C15), behaved in a similar way. Strong positive relationships were found between *Posidonia oceanica* patches growing on rocky bottoms (A20) and the patch complexity metrics (MSI and MPAR). A similar relationship exists between photophilic algae and patch complexity in the Cabrera Archipelago. Mean patch size (MPS) and mean patch edge (MPE) were strongly related to *Caulerpa prolifera* which is represented by a single large patch in the Costa de Llevant site (L26). Isolated *Posidonia oceanica* patches (A17 and C17) were positively correlated to the splitting index, which is a measure of fragmentation. A strong negative relationship was found between the division metric, mean shape index (MSI), fractal dimension (MFRACT) and continuous and mixed *Cymodocea nodosa* patches (A12 and C14, respectively). This pattern reflected the complex boundaries of these patches due to this species forming sparse meadows.

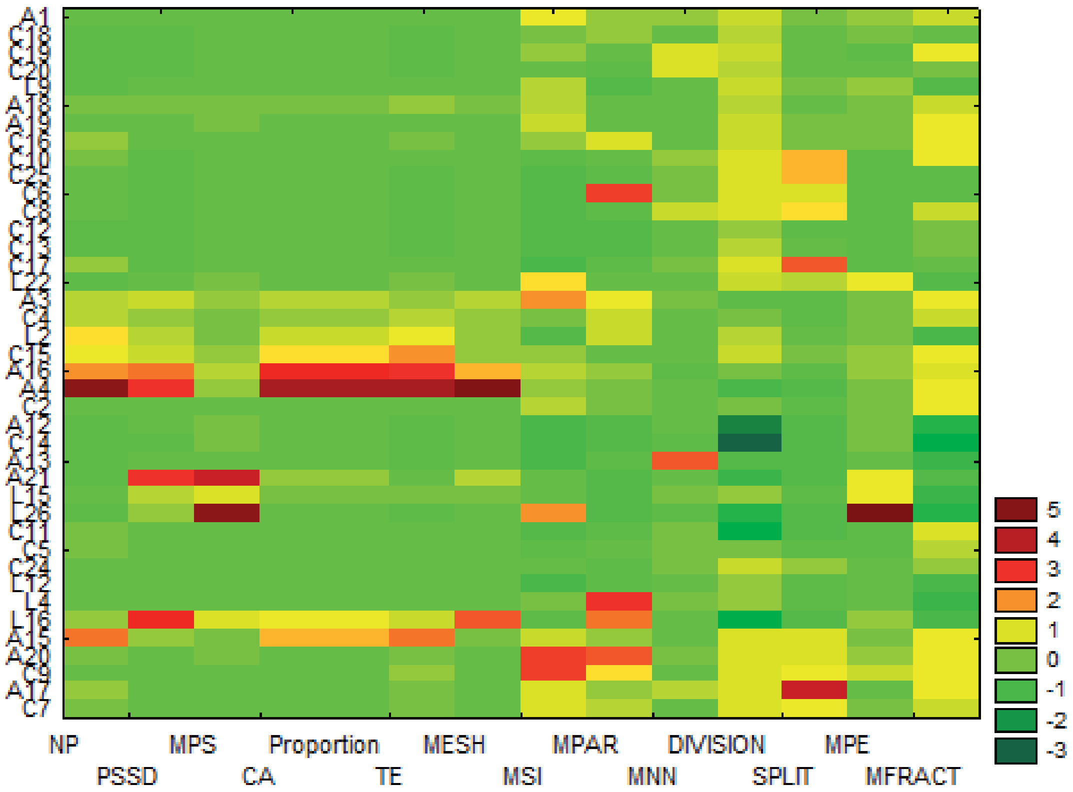

Figure 3.2: A two-way joining cluster analysis showed the importance of the individual aspects of seascape structure (x-axis) on cases (y-axis). The case code refers to the study site and the habitat type (numerical value). The corresponding habitat types are given in Table 3.1. The darker the colour, the stronger the relationship. Red means a positive relationship, while green shows negative relationships among the data.

4.2.2. Seascape structure and heterogeneity

The influence of seascape structure on the heterogeneity cf benthic habitats was determined using Principal Components Analysis. The PCA showed that four factors with eigenvalues >1 account for 80% of the variance in the data (Table 3.4). The majority of the spatial metrics describing seascape structure were correlated to either factor 1 or 2.

Table 3.4: Four factors with an eigenvalue >1 were included in the analysis (highlighted). These factors account for 80.3% of the variance.

	Eigenvalue	% Total	Cumulative	Cumulative
1	**5,505351**	**39,32394**	**5,50535**	**39,3239**
2	**2,875815**	**20,54154**	**8,38117**	**59,8655**
3	**1,855674**	**13,25481**	**10,23684**	**73,1203**
4	**1,005337**	**7,18098**	**11,24218**	**80,3013**
5	0,916108	6,54363	12,15829	86,8449
6	0,677908	4,84220	12,83619	91,6871
7	0,443519	3,16799	13,27971	94,8551
8	0,330401	2,36001	13,61011	97,2151
9	0,197775	1,41268	13,80789	98,6278
10	0,126012	0,90008	13,93390	99,5279
11	0,048331	0,34522	13,98223	99,8731
12	0,014097	0,10070	13,99633	99,9738
13	0,003671	0,02622	14,00000	100,0000
14	0,000000	0,00000	14,00000	100,0000

The factor loadings in Table 3.5 showed that factor 1 is strongly correlated to the number of patches (NP), class area (CA), the proportion (PROP) of each class, the total edge (TE) and the MESH metric. These metrics depict seascape composition. Factor 2 described the seascape configuration as it was correlated to the mean patch size (MPS), mean patch edge (MPE) and the DIVISION metric, which is a measure of fragmentation. A third factor showed a strong correlation to the mean shape index (MSI) and a moderate correlation to the mean patch edge (MPE). The fourth factor was most strongly related to the mean perimeter to area ratio (MPAR).

The factor loadings (Table 3.5) showed that habitat diversity and evenness were moderately correlated to factor 1 (-0.41 and -0-39, respectively). Habitat richness was negatively correlated to factor 1, suggesting that a more fragmented seascape with smaller patches may have higher habitat richness. Interestingly, species richness showed the opposite trend. Species richness was negatively correlated to both factor 1 and 2, indicating that more species would be found in seascapes that consist of larger patches with a lower fractal dimension (MFRACT). The statistical significance of the negative relationship between species richness and habitat richness was confirmed by the Spearman's rank correlation analysis (Table 3.6) which showed a perfect negative correlation between the two variables. The results also showed a statistically significant positive correlation between species richness and habitat diversity and evenness. Depth had a relatively weak positive correlation to factor 1 and negative correlation to factor 2, suggesting that seascape composition increases with increased depth, while the configuration becomes less fragmented.

Table 3.5: The factor loadings are given for the factors. Significant correlations between the spatial metrics and the factors are highlighted.

	Factor 1	Factor 2	Factor 3	Factor 4
NP	**-0,910857**	0,276695	-0,074575	0,118167
CA	**-0,981325**	0,100489	-0,062750	0,072318
MPS	-0,238228	**-0,759823**	0,487454	0,153056
PSSD	**-0,835894**	-0,310506	-0,010603	-0,115796
MSI	-0,139479	0,174685	**0,875344**	-0,070432
MPAR	-0,147748	0,297572	0,277869	**-0,840430**
MFRACT	-0,225025	**0,719169**	0,280150	0,183302
TE	**-0,944540**	0,213708	0,008865	0,109852
MPE	-0,091677	**-0,676835**	**0,651602**	0,196688
PROP	**-0,981296**	0,100568	-0,062898	0,072332
DIVISION	0,247669	**0,759358**	0,284196	0,169348
SPLIT	0,245385	**0,619432**	0,267003	0,244761
MESH	**-0,912629**	-0,053280	-0,150348	-0,071645
MNN	0,217606	-0,045771	-0,288525	0,233926
*Habitat richness	0,316770	0,343723	-0,188922	0,058699
*Habitat diversity	-0,409637	-0,034933	0,173218	0,059753
*Habitat evenness	-0,392947	-0,205625	0,196036	0,000307
*Spp. richness	-0,306142	-0,355082	0,186557	-0,064313
*Depth	0,254059	-0,231693	0,147620	-0,188772

Table 3.6: The Spearman's rank correlation coefficients that are significant at p<0.01 are highlighted.

	Habitat richness	Habitat diversity	Habitat evenness	Species richness	Depth
NP	0,0232	0,1240	0,1240	-0,0232	**-0,5399**
CA	**-0,4097**	**0,5129**	**0,5129**	**0,4097**	-0,3577
MPS	**-0,6213**	**0,6502**	**0,6502**	**0,6213**	-0,0521
PSSD	**-0,5630**	**0,5999**	**0,5999**	**0,5630**	-0,2445
MSI	-0,2179	0,3404	0,3404	0,2179	0,0389
MPAR	0,0177	0,0487	0,0487	-0,0177	-0,1065
MFRACT	**0,4098**	-0,0443	-0,0443	**-0,4098**	-0,3707
TE	-0,3281	**0,4007**	**0,4007**	0,3281	-0,3410
MPE	**-0,5785**	**0,5070**	**0,5070**	**0,5785**	0,2276
Proportion	**-0,4083**	**0,5108**	**0,5108**	**0,4083**	-0,3545
DIVISION	0,2820	-0,2219	-0,2219	-0,2820	-0,0714
SPLIT	0,2820	-0,2219	-0,2219	-0,2820	-0,0714
MESH	**-0,5098**	**0,5524**	**0,5524**	**0,5098**	-0,2791
MNN	0,0040	0,0777	0,0777	-0,0040	-0,1582
Habitat richness		**-0,7672**	**-0,7672**	**-1,0000**	-0,1886
Habitat diversity			**1,0000**	**0,7672**	-0,0412
Habitat evenness				**0,7672**	-0,0412
Species richness					0,1886

The Spearman's rank coefficients (Table 3.6) showed a number of significant relationships between specific aspects of seascape structure (i.e. spatial metrics) and the heterogeneity variables at the 99% confidence level. Habitat richness was negatively correlated to class area (CA), mean patch size (MPS), the standard deviation of patch size (PSSD), mean patch edge (MPE), the proportion of each class and the mesh metric. Habitat diversity, evenness and species richness showed the opposite trends to habitat richness. Where habitat richness is higher for seascapes that consist of smaller patches that are less abundant in area, habitat diversity and species richness are lower. While this was expected for habitat diversity, which is calculated based on the proportion of different habitats in the seascape, the strong relationship between species richness and habitat diversity was not anticipated. Another unexpected result was that depth did not play a significant role and was only significantly correlated to the number of patches.

4.2.3. Multiple regression analysis

The stepwise multiple regression analysis showed that the individual habitat heterogeneity variables were the best predictors of species richness. Habitat diversity explained 46% of the variation (F=33,25, P<0.000, n=39), evenness explained 83% (F= 184,73, P<0.000, n=39) and habitat richness encompassed all variation (R^2=1, F=25,78, P<0.000, n=39). The beta coefficients showed that habitat richness was negatively correlated to species richness. A 1 unit positive standard deviation change in X, will results in a negative standardized beta coefficient change of 24 units. The standardized beta coefficients of habitat diversity and evenness were both positive (B=71 and B=527 respectively). Out of the three habitat heterogeneity variables, habitat evenness had the highest standard error of 38.8.

With the exception of the habitat heterogeneity variables, the patch complexity model had the highest correlation with species richness. This model explained 35% of the variation in species richness (F= 6,64, P<0.001, n=37) and showed that mean shape index (MSI) and fractal dimension (MFRACT) were significant predictors. The standardized beta coefficient of the MSI predictor showed a positive correlation of 44 (p<0,001). On the other hand, the fractal dimension had a strong negative slope with a standardized beta coefficient value of -201 (p<0.000, n=37). The model intercept had a positive standardized beta coefficient of 716 with a standard error of 80.5.

The depth and connectivity variables showed no correlation to species richness. The seascape composition model accounted for a low 18% of the variance, which was not significant (F= 1,49, P=0,21664, n=35). The configuration of the seascape explained 23% of the variance (F=2,11, P= 0,08714, n=35), but again this relationship was not statistically significant.

Table 3.7: The stepwise multiple regression analysis of species richness showed that habitat heterogeneity and shape complexity were significant predictors at $p<0.05$.

Model	Multiple R	Multiple R^2	F	P	Predictor	Beta	Std.Err.	B	Std.Err.	t(n)	p-level	n
F1: Composition	0,41946	0,17595	1,49461	0,21664	Intercept			**449,7352**	**18,96212**	**23,71757**	**0,000000**	35
					NP	-0,556263	0,713705	-0,1366	0,17525	-0,77940	0,440982	35
					CA	-0,849539	1,348183	0,0000	0,00001	-0,63014	0,532698	35
					PSSD	0,490003	0,357633	0,0001	0,00010	1,37013	0,179372	35
					TE	1,245159	1,347743	0,0005	0,00049	0,92388	0,361872	35
					MESH	0,065872	0,472143	0,0000	0,00001	0,13952	0,889841	35
F2: Configuration	0,48141	0,23175	2,11164	0,08714	Intercept			**595,2111**	**99,10380**	**6,00594**	**0,000001**	35
					MPS	0,137143	0,307178	0,0002	0,00040	0,44646	0,658014	35
					MFRACT	-0,260495	0,172018	-88,9664	58,74898	-1,51435	0,138917	35
					MPE	0,199712	0,294309	0,0147	0,02173	0,67858	0,501867	35
					DIVISION	0,098243	0,211958	0,3033	0,65446	0,46350	0,645873	35
					SPLIT	-0,120590	0,187624	-0,7258	1,12926	-0,64272	0,524594	35
Connectivity	0,15256	0,02327	0,92932	0,34098	Intercept			**464,1302**	**13,59897**	**34,12979**	**0,000000**	39
					MNN	0,152559	0,158254	0,1013	0,10507	0,96401	0,340980	39
Complexity	0,59165	0,35005	6,64255	**0,00106**	Intercept			**715,550**	**80,50616**	**8,88814**	**0,000000**	37
					MSI	**0,522624**	**0,155741**	**43,897**	**13,08117**	**3,35572**	**0,001841**	37
					MPAR	-0,000196	0,141025	-0,002	1,78994	-0,00139	0,998898	37
					MFRACT	**-0,589875**	**0,147610**	**-201,459**	**50,41280**	**-3,99618**	**0,000295**	37
Depth	0,21523	0,04632	1,89433	0,17657	Intercept			**501,4254**	**25,62787**	**19,56562**	**0,000000**	39
					Depth	0,215227	0,156375	1,3266	0,96388	1,37635	0,176566	39
Habitat richness	1,00	1,00	24912,78	**0,00000**	Intercept			**745,8689**	**1,811928**	**411,644**	**0,000000**	39
					Richness	**-0,999218**	**0,006331**	**-23,5471**	**0,149186**	**-157,838**	**0,000000**	39
Habitat diversity	0,67837	0,46018	33,24665	**0,00000**	Intercept			**254,0493**	**38,54292**	**6,591336**	**0,000000**	39
					Diversity	**0,678368**	**0,117650**	**71,0599**	**12,32398**	**5,765991**	**0,000001**	39
Habitat evenness	0,9087	0,8257	184,7336	**0,00000**	Intercept			**315,6273**	**12,46067**	**25,32988**	**0,000000**	39
					Evenness	**0,908672**	**0,066855**	**526,9057**	**38,76679**	**13,59168**	**0,000000**	39

3.4. Discussion

Effective management of heavily threatened coastal ecosystems requires information on the ecological consequences of spatial patterns (Pittman et al. 2011). In this chapter, the relevance of landscape ecology techniques to provide meaningful information on the relationship between spatial patterns and biodiversity in the marine coastal environment was tested.

The results indicated that species richness was determined by habitat diversity, which is in turn driven by the class area, mean patch size and the edge metrics. This suggests that seascape managers should focus on protecting the abundance of different habitats rather than the number of habitat types. The value of diversity over richness is not new (Magurran 1988), however conclusive evidence of the relationship between habitat diversity and faunal diversity had previously been lacking in seascape ecology studies (Wedding et al. 2011). The relationship between habitat diversity and species richness is based on the assumption that a higher abundance and variety of habitats can support more species. In a study on the use of surrogates of biodiversity for marine reserve selection, Ward et al. (1999) found that habitat-level surrogates included around 93% of the available taxa, suggesting that the findings of this chapter are feasible.

The negative correlation between habitat richness and species richness found in this chapter was not anticipated. When the species were divided into phyla (Figure 3.3), the study sites showed similar patterns in the types of species present, however the Cabrera Archipelago had considerably fewer mollusc species, bryozoans and arthropod species. This is also the study site with the highest habitat richness. This shows that the negative relationship between species richness and habitat richness is specific to one study site and is not a general phenomenon. The fact that the lowest species richness is found in the Cabrera Archipelago which has the highest level of protection (IUCN Category II) and that has been protected the longest, excludes the possibility that disturbance is the cause. The most likely explanations are that there is not enough of each habitat type or that an environmental factor such as salinity, nutrients or oxygenation is limiting species richness. It is possible that the steep depth gradient at this site is also a limiting factor.

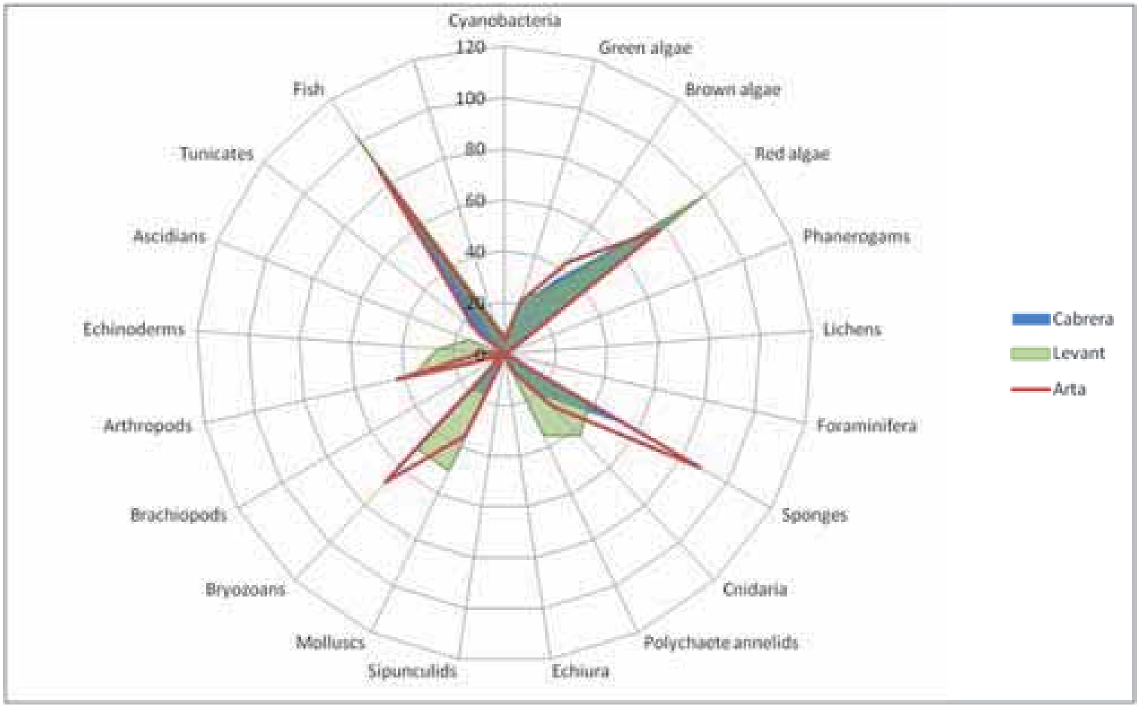

Figure 3.3: The radar chart shows the number of species in each phylum for the three study sites.

The findings of this study also show that species richness is influenced by patch shape complexity. The multiple regression analysis showed that the mean shape index was positively correlated to species richness, but that an increase in the fractal dimension of a patch results in decreased species richness. The study by Kostylev et al. (2005) used fractals to explore the species-area relationship in intertidal zones and found that complex habitats support more species. Their measurements using fractals was applied to surface complexity rather than shape complexity which is measured in this study. Further research on the relationship between shape complexity and surface complexity is needed to determine the reason for the contrasting fractal dimension results. The assumed relationship between the complexity of patches and species richness is based on the concept that complex habitats provide a greater variety of physical and biotic features that can be colonized thereby supporting a greater number of species (Kohn & Leviten 1976; Kostylev et al. 2005). Studies have generally found positive correlations between rugosity (i.e. the habitat available for colonization, foraging and shelter for organisms) and species richness (Gratwicke & Speight 2005). Inconclusive results linking marine species richness to habitat complexity are largely a result of different approaches and definitions of complexity (Gratwicke & Speight 2005). In this study, the mean shape index is calculated based on a relationship between perimeter and area. Patch complexity, therefore refers to the shape of the perimeter of the patch. The results of this study suggest that the more complex the shape, the higher the species richness.

The influence of habitat fragmentation on coastal habitats differs to that of the terrestrial environment due to the higher functional connectivity of the marine realm (Montefalcone et al. 2010; Worm et al. 2006). While marine organisms have a greater capacity for dispersal, isolation of patches can reduce the connectivity of marine ecosystems significantly (Meynecke et al. 2008). A disruption of connectivity negatively impacts population dynamics, community structure and has been found to increase the susceptibility to invasion by alien species (Montefalcone et al. 2010).

In conclusion, the cuantification of seascape structure can provide useful information on the relationships between spatial patterns, species richness and habitat diversity. The findings of this study show that species richness is dependent on the abundance of habitats and shape patch complexity. The two factors should be priorities for the conservation of biodiversity in the seven Sites of Community Importance. This chapter confirms the relevance and application of spatial pattern metrics as the class level for coastal Mediterranean seascapes.

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CHAPTER 4

Determining the effects of anthropogenic disturbance on seascape structure

Abstract

The greatest loss of marine biodiversity occurs in the coastal zone, where pressures from both the land and sea combine. Effective management of this unique interface requires quantitative information on the response of coastal ecosystems to changes in environmental conditions and sea use, but such information is often lacking. Spatial pattern metrics, originating from the interdisciplinary field of landscape ecology, provide quantitative information on the structure of the seascape. In this study, I use such metrics to identify the aspects of Mediterranean seascape structure that are most sensitive to disturbance. Spatial metrics were calculated for seven typical Mediterranean seascapes in Spain's Balearic Islands. A Principal Components Analysis (PCA) was used to explore the relationship between anthropogenic disturbance and spatial metrics. The results indicate that land-based disturbance is resulting in degradation and fragmentation of coastal habitats. Multivariate analysis confirmed that the number of patches in each class, the class area and the proportion of each class showed the strongest correlations to pollution. These results indicate that reducing pesticide and fertilizer inputs should be a priority in the Balearic Islands. I conclude that the application of spatial pattern metrics is a useful tool for providing information on which aspects of seascape structure are at the greatest risk from disturbance. This technique can provide a cost and time effective decision-support tool for marine spatial planning.

4.1. Introduction

Coastal marine ecosystems are among the most diverse and productive in the world (Suchanek 1994). They are also some of the most heavily threatened ecosystems with the greatest loss of marine diversity occurring in the coastal zone (Barbier et al. 2011; Gray 1997). Most of the threats to these ecosystems are a result of human pressures (Gray 1997). With more than three billion people residing within 200 km of the coast and the number expected to double by 2025, the concern over the growing human population has lead to increased urgency to understand and reduce the impact of anthropogenic pressures on these coastal ecosystems (Creel 2003).

Marine spatial planning (MSP) is a highly advocated approach to balancing human uses of the marine environment and the protection of the ecosystems and the services they deliver (Collie et al. 2012; Halpern et al. 2008). This interdisciplinary framework provides an effective and rational approach to the management of human activities by using spatial data to identify conflicts with marine ecosystems (Collie et al. 2012). Understanding how these anthropogenic pressures affect marine ecosystems is needed to improve and fine-tune management responses. This is often challenging as the ecological functioning of marine ecosystems is poorly understood. The emerging field of seascape ecology provides an interesting option for bridging this knowledge gap by using seascape structure as an indicator of the functioning of an ecosystem. This spatially explicit approach can potentially be applied to identify how anthropogenic pressures influence specific aspects of seascape structure, thereby providing information for targeted management decisions.

The seascape ecology approach has been derived from the analytical and theoretical framework of landscape ecology. Landscape ecology is based on the concept that ecosystem functioning is determined by the spatial configuration of habitat patches within a landscape and their location relative to each other (Pittman 2013). The application of landscape ecology concepts and techniques to the seascape have been explored for coastal environments, with particular success in shallow-water benthic ecosystems (Boström et al. 2011). This approach allows for a better understanding of the multi-scale relationships between spatial patterns and ecological processes (Boström et al. 2011; Wedding et al. 2011).

In this study I use spatial pattern metrics to identify which aspects of Mediterranean seascape structure are most sensitive to disturbance. Little information is available about the response of seascape structure to human pressures and I hope to demonstrate that spatial pattern metrics can be a valuable and consistent tool for measuring and monitoring the structure of disturbed coastal habitats for marine spatial planning. Understanding the ecological consequences of spatial patterns caused by human activities is of the utmost importance for effective management and planning of the coastal environment (Boström et al. 2011; Pittman et al. 2011; Wedding et al. 2011).

4.2. Method

4.2.1. Study site description

Study sites in Spain's Balearic Islands were chosen due to the presence of a mosaic of habitats representative of typical Mediterranean seascapes, the high conservation interest in the area and the availability of accurate and fine scale benthic habitat maps. The location of the study sites on Mallorca, Ibiza and Formentera islands also made for an interesting comparative study between different cisturbance levels (Figure 4.1).

Historically, the Balearic Islands relied on agriculture and cattle as the primary economy, however a massive shift towards the tourism sector has occurred in recent decades (Morales-Nin et al. 2005). Tourism related pressures such as coastal development and the seasonal increase and exploitation of natural resources is a significant source of disturbance (Morales-Nin et al. 2005). Agricultural runoff, shipping (container ships, ferries, and recreational vessels), trawling and commercial fishing constitute other important sources of disturbance (Box et al. 2007; Diedrich et al. 2010).

4.2.2. Data collection

Benthic habitat maps were cbtained from the Posidonia LIFE project for the seven study sites in the Balearic Islands at a scale of 1:1,000 (Posidonia Life, 2003). The cartography was performed using a side-scan sonar technique for areas between 5 and 40 m deep and orthophotos for areas at depths between 0 and 5 m.

Spatial maps of the cisturbance indicators were downloaded from the National Center for Ecological Analysis and Synthesis (NCEAS) project on the human impacts on Mediterranean marine ecosystems (Halpern et al. 2008; Micheli et al. 2013). The rasters were projected in Lambert Azimuthal Equal Area projection centred on the Mediterranean sea, however further georeferencing using control points was necessary. A mask was used to crop the rasters to the study area and data was extracted at patch level in ArcGIS 9.3 (ESRI).

Eight disturbance variables were selected based on the resolution of the spatial data and the relevance of the variable as a common form of disturbance in the Balearic Islands (Figure 4.1). The disturbance variables chosen were: (a) commercial shipping, (b) fertilizer inputs, (c) imperviousness (non-point pollution inputs), (d) pesticide inputs, (e) risk of hypoxia, (f) distance to area at risk of hypoxia, (g) the Global Human Influence Index (HII), and (h) the distance to the area with an HII score. A description of the disturbance data is given in Table 4.1. The data were extracted at patch level and the average score per habitat class was used in this study.

Table 4.1: Eight disturbance variables and two additional variables (depth and distance to shore) were used in this study.

Variable	Description	Unit	Resolution
Shipping[1]	The intensity of commercial shipping was calculated based on the number of ships per year.	Ships yr^{-1}	Lines
Fertilizers[1]	The nutrient inputs were based on the annual use of fertilizers modelled by Halpern et al. (2008).	N+P205 tonnes / 1000 Ha	$1km^2$
Impervious surface pollution[1]	The non-point source inorganic pollution inputs from impervious surfaces (urban runoff) were modelled by Halpern et al. (2008).	Tonnes / 1000 Ha	$1km^2$
Pesticides[1]	Inorganic pollution was modelled by Halpern et al. (2008).	Tonnes / 1000 Ha	$1km^2$
Hypoxia[1]	The risk of hypoxic events in low-oxygen conditions was modelled by Djavidnia et al. (2005).	Risk value between 0 (low) and 1 (high)	$1.5km^2$
Distance to hypoxic area[1]	The distance to an area with a risk of hypoxia was calculated using the 'Near' tool in ArcGIS 9.3 (ESRI 2009).	Metres	
HII[2]	The Global Human Influence Index is based on nine global data layers that cover human land use and infrastructure, human population pressure and human access.	Index values ranging from 0 to 64	$1km^2$
Distance to area with HII	The distance to an area with a HII score was calculated using the 'Near' tool in ArcGIS 9.3 (ESRI 2009).	Metres	
Depth	Bathymetry data was obtained at a coarse resolution for the entire study area and where possible, at a fine resolution for individual study sites.	Lines	5-10m
Distance to shore	The distance to the shore was calculated using the 'Near' tool in ArcGIS 9.3 (ESRI 2009).	Metres	

[1]http://www.nceas.ucsb.edu/globalmarine/mediterranean

[2]http://sedac.ciesin.columbia.edu/data/set/wildareas-v2-human-influence-index-geographic

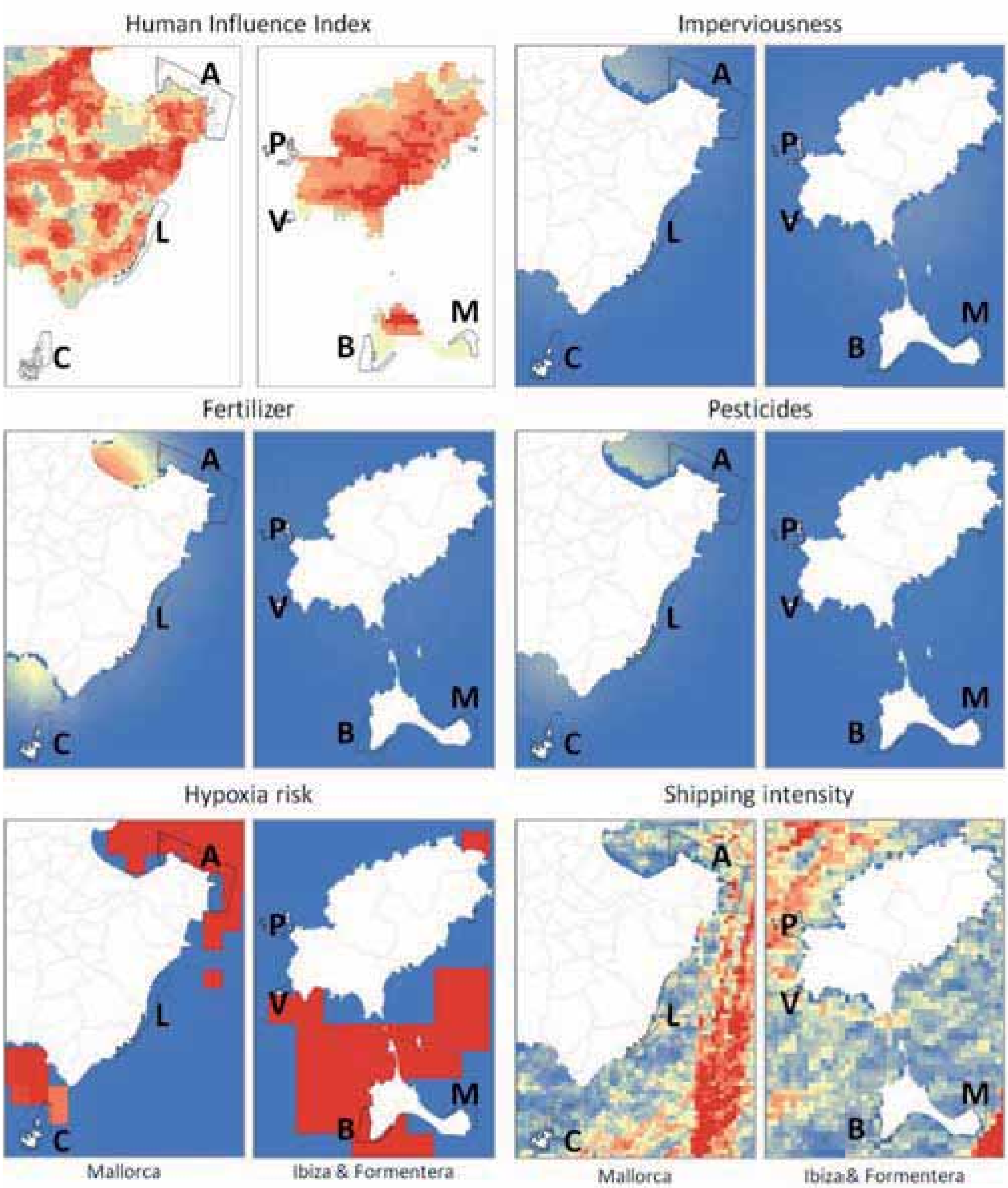

Figure 4.1: Six types of disturbance were included in the analysis, including the Human Influence Index, pollution from impervious surfaces, fertilizers, pesticides, risk of hypoxia and commercial shipping intensity. A: Muntanyes d'Artà, B: Cape de Barbaria, C: Cabrera Archipelago, L: Costa de Llevant, M: La Mola, P: Llots de Ponent, V: Es Vedra.

4.2.3. Seascape structure quantification

Spatial pattern metrics were calculated at class level for the 26 habitat types present in the seven study sites. The ^{V}LATE 1.1 (Tiede 2005) vector-based extension was used to calculate the spatial metrics in ArcGIS 9.3 (ESRI). A total of 13 spatial metrics were extracted that quantify area, form, edge, diversity, subdivision and proximity (Table 4.2). The detailed descriptions and equations used to calculate the spatial metrics are given in Appendix A.

Table 4.2: A total of 13 spatial pattern metrics were calculated using vector-based software. The metrics provide information at patch (P), class (C) and seascape (S) level.

	Abbreviation	Spatial pattern metric	Level
Area	NP	Number of patches	C, S
	CA	Class area	P, C, S
	MPS	Mean patch size	C, S
	PSSD	Stand deviation of patch size	C, S
Form	MSI	Mean shape index	P, C, S
	MPAR	Mean perimeter to area ratio	P, C, S
	MFRACT	Mean fractal cimension	P, C, S
	TE	Total edge	C, S
	MPE	Mean patch edge	P, C, S
Diversity	PROP	Proportion	S
Subdivision	DIV	Division	C
	SPLIT	Splitting index	C
	MESH	Mesh	C

4.2.4. Multivariate analysis

A Principal Components Analysis (PCA) was performed using STATISTICA 8.0 (StatSoft 2007) to extract any underlying, uncorrelated factors and to determine if, and which disturbance indicators were influencing the spatial patterning of the seascape. Factors with an eigenvalue >1 were extracted and the variables and cases plotted on the factor plane to explore the correlations with the PCA axes. The spatial metrics were projected as active variables with the disturbance intensities as supplementary variables. The Spearman's rank correlation coefficient was calculated to test the statistical relationship between the spatial metrics and the disturbance variables.

4.3. Results

The influence of disturbance on seascape structure was determined using Principal Components Analysis. The PCA showed three factors that explain 75% of the variance (Table 4.3). Factor 1 describes seascape composition, namely the number of patches (NP), class area (CA), the proportion of each class and the total edge (TE). Factor 2 is described by the mean patch size (MPS), mean patch edge (MPE) and the DIVISION metric, which is a measure of fragmentation. Factor 2 depicts the seascape configuration. The third factor is composed of the shape complexity metrics; mean shape index (MSI) and mean perimeter to area ratio (MPAR).

Table 4.3: The eigenvalues of the correlation matrix show that three factors account for 75% of the variance.

	Eigenvalue	% Total	Cumulative	Cumulative
1	**5,156842**	**39,66802**	**5,15684**	**39,6680**
2	**2,908194**	**22,37072**	**8,06504**	**62,0387**
3	**1,690521**	**13,00401**	**9,75556**	**75,0427**
4	1,216470	9,35746	10,97203	84,4002

5	0,773748	5,95191	11,74578	90,3521
6	0,516729	3,97484	12,26250	94,3270
7	0,343281	2,64062	12,60578	96,9676
8	0,196342	1,51032	12,80213	98,4779
9	0,126781	0,97524	12,92891	99,4531
10	0,042637	0,32797	12,97154	99,7811
11	0,020336	0,15643	12,99188	99,9375
12	0,008119	0,06245	13,00000	100,0000
13	0,000001	0,00001	13,00000	100,0000

The effect of the depth of a patch and its distance from the shore were also explored in the PCA. Depth was found to have a moderately weak positive influence (0.22) on seascape patch complexity (factor 3), suggesting that patches in shallower areas are less complex in shape. A weak positive correlation was also found with factor 2 (configuration). The distance from the shore (shoredist) was found to have a strong positive influence (0.78) on factor 1 and a moderate negative correlation with factor 2. This finding indicates that the proportion of habitats increases with the distance from shore, while the average patch size decreases. This suggests that seascapes are more heterogeneous further from the coastline.

Table 4.4: The loadings of the spatial metrics and the disturbance variables (*) are given for the three factors.

	Factor 1	Factor 2	Factor 3
NP	**0,790648**	-0,523562	0,092539
CA	**0,961389**	-0,208840	-0,096037
MPS	0,385623	**0,776664**	-0,059091
PSSD	**0,808464**	0,397574	-0,093888
Proportion	**0,961438**	-0,208791	-0,096254
TE	**0,907706**	-0,320793	0,065262
MPE	0,268798	**0,757030**	0,097570
MSI	0,215220	0,540981	**0,712765**
MPAR	0,313956	0,211853	**0,683278**
MFRACT	-0,024160	0,312521	0,332884
DIVISION	-0,088182	**-0,617572**	0,520301
SPLIT	-0,090862	-0,541273	0,506046
MESH	**0,907237**	0,003540	-0,157714
*Depth	-0,029582	0,137172	0,201702
*Shoredist	**0,770493**	-0,325593	-0,103630
*Ship	-0,072828	0,065741	-0,084420
*Fert	0,475962	-0,283597	-0,038295
*Impv	0,315477	-0,059518	-0,113964
*Pest	0,436132	-0,178966	-0,067107
*Hypo	-0,033784	0,116365	-0,084774
*Hypodist	-0,132458	0,050566	0,076136
*HII	0,133381	0,237600	-0,142824
*HIIdist	0,024292	-0,104151	-0,046474

The disturbance variables were found to influence different aspects of the seascape structure. Fertilizers, imperviousness and pesticides had a moderate positive effect on factor 1 with values of 0.49, 0.29, and 0.45, respectively. These results may suggest that the proportion of different habitats increases with more land-based inputs. Fertilizers were also negatively correlated to the configuration of the seascape (factor 2), albeit moderately weakly (-0.27). This may indicate that fragmentation of habitats is occurring with increased fertilizer inputs. A moderately weak correlation (0.24) was found between the Human Influence Index (HII) and factor 2. Shipping and hypoxia risk did not appear to influence seascape structure.

The Spearman's rank correlation coefficients were calculated to determine statistically significant correlations between the disturbance variables and individual spatial pattern metrics. The results (Table 4.5) show that the average patch depth and distance to the shore did not influence the structure of patches at the 99% significance level.

Fertilizer was found to have a significant relationship with the number of patches, total edge, division and the splitting index. This is in accordance with the PCA results which suggested that fertilizers may be increasing fragmentation in seascapes. The Spearman's correlation coefficient also confirmed the PCA results for pesticides. Statistically significant correlations were found for the number of patches, class area, proportion, total edge, division and the splitting index. In other words, pesticides are affecting seascape composition. The HII score was found to influence mean patch size and mean patch edge significantly. The results indicate that patches nearer areas that are heavily influenced by humans, tend to be larger in size and have more edge.

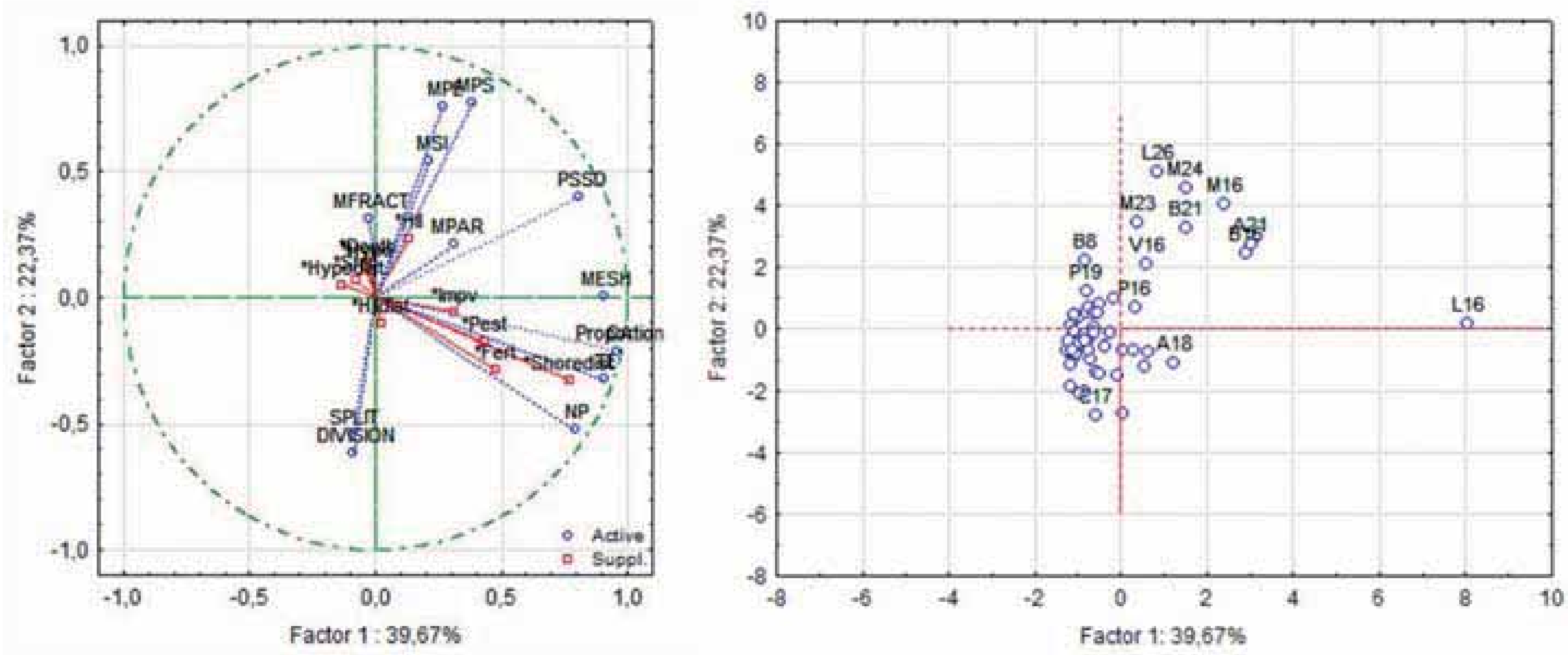

Figure 4.2: The factor loadings of the variables (left) and cases (right) are plotted on the factor 1x2 plane.

Table 4.5: The Spearman's rank correlation coefficients that are statistically significant at p<0.01 are highlighted.

	Depth	Shoredist	Ship	Fert	Impv	Pest	Hypo	Hypodist	HII	HIIdist
NP	-0,1480	0,1672	-0,1186	**0,5923**	0,2704	**0,6427**	-0,1122	-0,0542	-0,2324	0,1265
CA	-0,0533	0,1859	-0,0345	0,2931	0,1755	**0,3476**	0,0601	-0,1617	0,1081	0,0092
MPS	0,0550	0,1273	0,0263	-0,0897	-0,0312	-0,1142	0,1620	-0,1356	**0,4012**	-0,0969
PSSD	0,0414	0,1704	0,0083	0,0800	0,0274	0,1310	0,1682	-0,0765	0,2334	-0,0451
Proportion	-0,0460	0,1866	-0,0284	0,3019	0,1842	**0,3560**	0,0534	-0,1503	0,1018	0,0107
TE	-0,0114	0,0946	-0,0701	**0,4416**	0,2555	**0,5134**	-0,0505	-0,1460	0,0366	0,0160
MPE	0,2321	-0,1104	0,0062	-0,1317	-0,0446	-0,1151	0,0786	-0,1360	**0,4232**	-0,1885
MSI	0,2626	-0,2727	-0,0236	-0,1575	0,0155	0,0499	0,0625	0,0287	0,0861	-0,0604
MPAR	0,1716	-0,1567	-0,0449	0,0310	0,0896	0,1939	0,0524	0,1755	-0,1203	0,0164
MFRACT	0,0659	-0,0737	-0,0571	-0,0790	-0,0255	0,1367	0,1089	0,1376	-0,2313	-0,0435
DIVISION	0,0685	-0,1509	-0,2535	**0,3875**	0,1174	**0,4720**	-0,1381	-0,1771	-0,2577	-0,0548
SPLIT	0,0691	-0,1508	-0,2519	**0,3933**	0,1235	**0,4735**	-0,1424	-0,1716	-0,2595	-0,0510
MESH	-0,0534	0,2384	0,0847	0,1145	0,0780	0,0953	0,1149	-0,0385	0,2619	0,0200

Spatial pattern metrics (class level)
NP: number of patches
CA: class area
MPS: mean patch size
PSSD: patch size standard deviation
Proportion: proportion of each class
TE: total edge
MPE: mean patch edge
MSI: mean shape index
MPAR: mean perimeter to area ratio
MFRACT: fractal dimension
DIVISION: division
SPLIT: splitting index
MESH: mesh

Disturbance variables
Depth: average depth of class
Shoredist: distance to shore
Ship: shipping intensity
Fert: fertilizer inputs
Impv: imperviousness
Pest: pesticide inputs
Hypo: risk of hypoxia
Hypodist: distance to area at risk of hypoxia
HII: Human Influence Index
HIIdist: distance to area with HII score

4.4. Discussion

Due to the rapid loss and degradation of marine biodiversity and coastal ecosystem services globally, it has become imperative to understand how disturbance affects seascape structure so that appropriate management responses can be made. In this chapter, landscape ecology techniques were used to explore how anthropogenic disturbance influenced coastal Mediterranean seascape structure.

The results indicated that the area of habitats and their relative proportion to the seascape increased with distance from the shore and with increased land-based inputs. Specifically, the proportion of dispersed and degraded habitats was found to increase, while continuous habitats appear to become fragmented.

Land-based inputs, such as fertilizers and pesticides, increase the nutrient concentration of the water column, which can degrade pollution-sensitive habitats like seagrass and coralligenous communities (Carpenter et al. 1998; Invers et al. 2004). Indeed, the case loadings in the PCA and the 2-way joining cluster analysis (Figure 4.3) showed that fertilizers and pesticides were correlated to *Posidonia oceanica* meadows that are isolated, degraded and have sink-holes. This trend of fragmentation as a result of pollution was confirmed by the results of the Spearman's correlation coefficients.

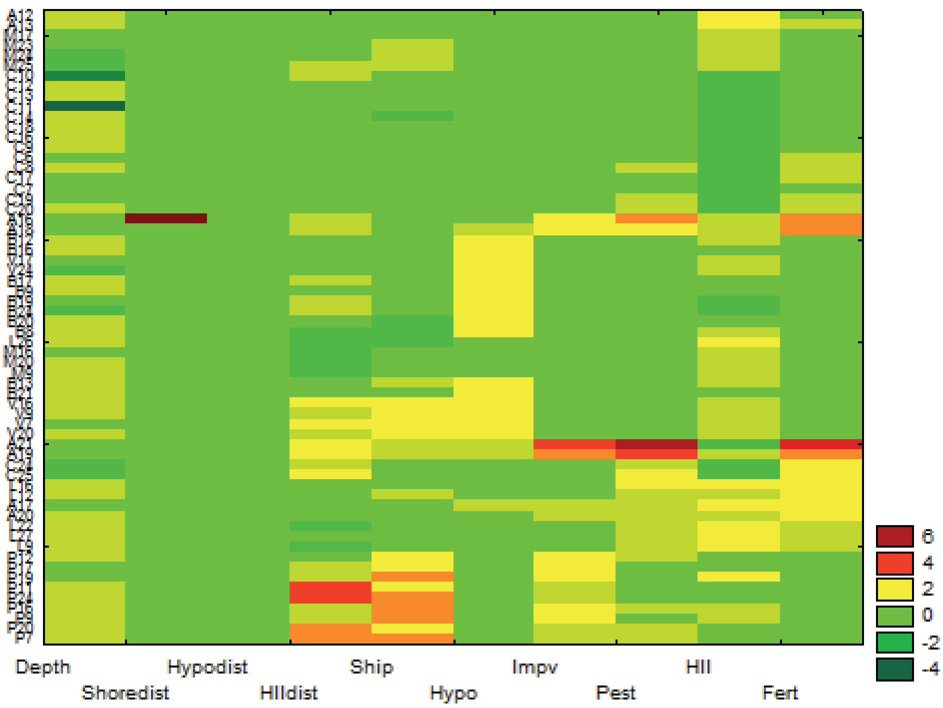

Figure 4.3: The 2-way joining cluster analysis shows the relationships between the disturbance variables and the cases. Red squares show strong positive relationships, while dark green squares represent strong negative relationships.

An additional correlation that was highlighted by the Spearman's correlation coefficients was that between the Human Influence Index (HII) and mean patch size and mean patch edge. The average patch size near areas of high human influence tends to be larger. Interestingly, a similar trend was found in Chapter 5, where the largest patches of *Posidonia oceanica* were found in the most populated study sites. While further research is required to investigate the exact cause, I hypothesize that the most likely explanation for this pattern is that while the largest patches (which increase MPS) lie alongside heavily populated terrestrial areas, steep cliffs along the coastline separate them from disturbance. This is certainly the case in the northern part of the Costa de Llevant site and Muntanyes d'Artà where human access to the sea is limited by cliffs. These two sites also contain the largest patches of *Posidonia oceanica*.

While land-based inputs do certainly contribute to habitat degradation and fragmentation, it should be kept in mind that the pollution data used in this study is modelled based on river plumes, urban runoff and catchments. The model does not take currents into account, meaning that the quantities of modelled pollution do not necessarily correspond to actual values. With this is mind, physiological variables like habitat depth and the distance from shore were included to eliminate to the possibility that habitat quality was simply responding to its location rather than disturbance. A difference in the spatial resolution of the disturbance data and the seascape maps is a limitation of this study that meant that only class and seascape level analysis could be performed. Accurate predictions could not be made regarding the impact of disturbance on the seascape at patch level.

Few studies have applied landscape ecology techniques to coastal habitats in Mediterranean seascapes (Garrabou et al. 2002; Garrabou et al. 1998) and none, to my knowledge, have explored the effects of disturbance on seascape structure at the scales used in this study. Using spatial pattern metrics to interpret large scale spatial data could be a powerful conservation tool. Not only can priority conservation areas be identified using this technique, but the consequences of anthropogenic disturbance on specific seascape components can be determined. This could be of particular use for marine spatial planning by providing an ecologically-relevant decision-support tool.

Future studies should take care to match the scale of the data with the level of focus for the study. In this case, simply exploring the relationship between seascape structure and select disturbance variables was sufficient, however further research should combine field-based measurements with this technique to determine the accuracy of the results. It is hoped that 'ground-truthing' will confirm the applicability and relevance of landscape ecology spatial metrics for coastal Mediterranean seascapes.

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CHAPTER 5

Protecting ecosystem service delivery in Mediterranean seagrass-dominated seascapes

Abstract

Seagrass meadows are highly productive ecosystems and important carbon sinks. Their high conservation value coupled with the availability of high resolution spatial data make seagrass meadows an ideal focal habitat with which to assess the relationship between spatial patterns and ecosystem service delivery. Understanding this link is crucial for identifying conservation priorities and designing target-specific management responses. In this chapter, the effect of the drivers of spatial patterns on seascape structure and the associated ecosystem services in Spain's Balearic Islands is explored using regression tree analysis. The standing stock of seagrass carbon was used as an indicator for ecosystem service delivery. Carbon stocks can be valued and are therefore a useful link to the socio-economic aspects of ecosystem-based management. The results indicate that land-based pollution is having a significant effect on carbon stocks. Interestingly, more seagrass and therefore greater carbon stocks were present in areas with additional nutrient inputs. This study is a first step towards understanding the consequences of anthropogenic pressures on ecosystem service delivery in coastal Mediterranean seascapes.

5.1. Introduction

The ecological functioning and processes of coastal and marine ecosystems are heavily threatened by anthropogenic pressures (Barbier et al. 2011; Daily et al. 1997). These pressures alter the composition and configuration of the mosaic of habitats, thereby changing the fluxes of energy, matter and species and disrupting key ecological processes (Ernoult et al. 2003; Pickett & Cadenasso 1995). Understanding the consequences of spatial patterns on ecological processes is one of the key challenges in both landscape and seascape ecology and is of the utmost importance for the effective management of coastal and marine ecosystems (Pittman et al. 2011).

The link between seascape structure and ecosystem service delivery is a notable knowledge gap in the field of seascape ecology and the complexity of coastal and marine ecosystems makes filling this gap challenging. Spatial pattern metrics have been recommended to link the spatial structure of landscapes to ecosystem service delivery in terrestrial environments (Frank et al. 2012). As the central concepts and analytical approaches of landscape ecology have been suggested to be equally applicable to benthic environments as to their terrestrial counterparts (Boström et al. 2011; Pittman et al. 2011; Wedding et al. 2011), spatial metrics were used to quantify seascape structure at habitat level. In this way, standing seagrass carbon stocks could be linked to the spatial patterns of the habitat mosaic. Carbon stocks were used as an indicator of an ecosystem service provided by seagrass. Based on biomass and primary productivity, carbon stocks were modelled using the species, the quality of the seagrass meadows and the area covered by each class. The wealth of studies on carbon captured by Mediterranean seagrass species contribute to the accuracy at which carbon stocks can be quantified (Alcoverro et al. 2001b; Barrón et al. 2004; Duarte 1990; Duarte & Chiscano 1999; Duarte et al. 2013; Fourqurean et al. 2012; Gacia et al. 1999; Kennedy et al. 2010; Pérez & Romero 1994).

Due to the hierarchical nature and complexity of ecological systems, it is particularly challenging to isolate the product of an ecological process and attribute it to the spatial patterns of the seascape. Non-linear interactions between a large number of ecosystem components often result in unexpected dynamics or properties of self-organization (Wu & David 2002). Hierarchy theory provides a framework for incorporating the complexity of systems in ecological studies (Allen & Starr 1982). A detailed review of the concepts of hierarchy theory proposed by Allen & Starr (1982) are given by Wu & David (2002) and Turner et al. (2001). In the simplest form, hierarchy theory states that an ecosystem is structured vertically into levels and horizontally into holons or subsystems (Allen & Starr 1982). A holon is a term coined to describe something that is simultaneously a whole and a part. Interactions occur between holons, but also between levels. The lower levels in the hierarchy are constrained by higher levels (Allen & Starr 1982). Based on this concept, ecological studies should include a minimum of three levels, namely the level of interest, the level above (which controls the focal level), and the level below (which provides the mechanisms for the focal level) (Turner et al. 2001). In this way, logical and functional links between levels can be identified. The principal advantage of the hierarchy theory approach is that complex systems can be decoupled into levels and holons without losing important information (Wu & David 2002). Decomposing a system vertically allows for the effects of upper and lower levels on a subsystem to be studied in isolation without between-subsystem interactions (Wu & David 2002).

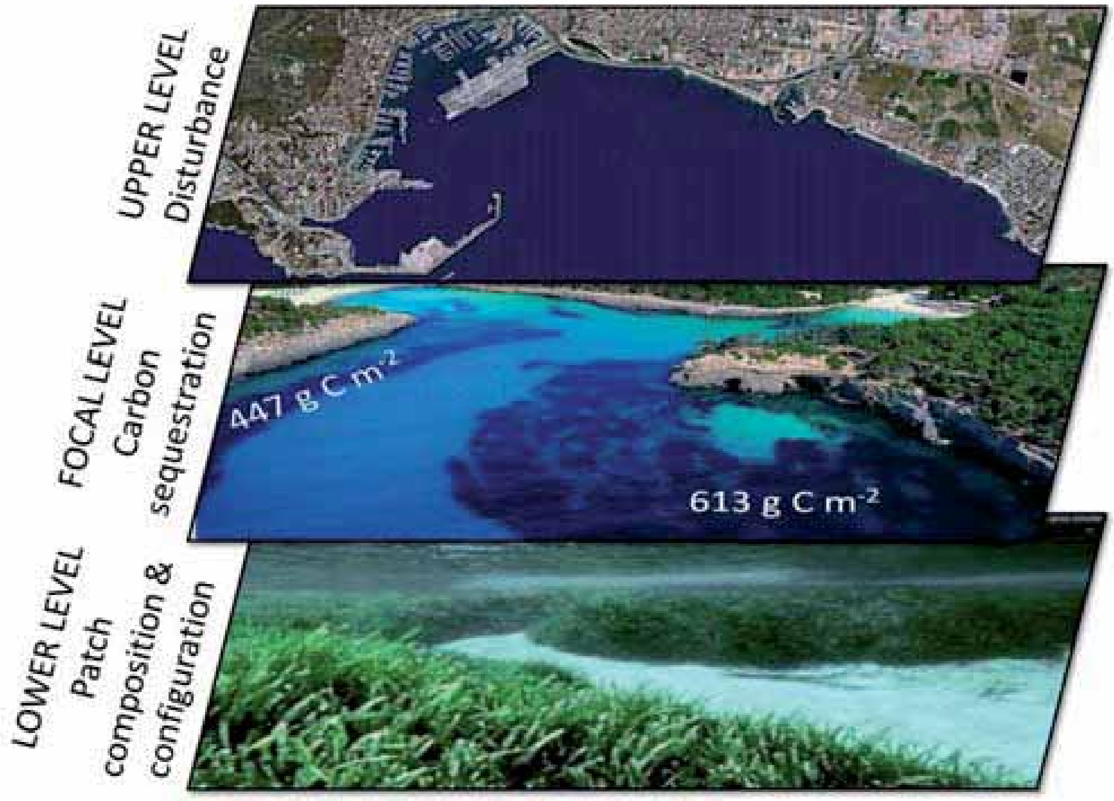

Figure 5.1: The hierarchy theory approach incorporates the influence of the upper and lower hierarchical levels on the focal level in order to simplify the complexities of ecosystems.

Regression tree analysis was used to address the hierarchical nature and interactions of ecological systems in this study. The ability of regression trees to deal with non-linear relationships, high-order interactions and missing values make it an ideal analytical method for describing and modelling complex ecological interactions (De'ath & Fabricius 2000). A significant advantage of this approach was its ability to assess hierarchical interactions (Michaelsen et al. 1994). Using the hierarchy theory to justify data selection, and regression trees to analyse it, the top-down influence of anthropogenic disturbance and the bottom-up effects of seascape structure on seagrass carbon stocks was explored in this study.

5.2. Method

5.2.1. Study site selection

Standing carbon stocks of seagrass were modelled for seagrass meadows in seven study sites in the Balearic Islands (Figure 5.2). The study sites were chosen due to the presence of a mosaic of habitats representative of typical Mediterranean seascapes, the high conservation interest in the area and the availability of accurate and fine scale benthic habitat maps. The location of the study sites on Mallorca, Ibiza and Formentera islands also made for an interesting comparative study between different disturbance levels. Historically, the Balearic Islands relied on agriculture and cattle as the primary economy, however a massive shift towards the tourism sector has occurred in recent decades (Morales-Nin et al. 2005). Tourism related pressures such as coastal development and the seasonal increase and exploitation of natural resources is a significant source of disturbance (Morales-Nin et al. 2005). Agricultural runoff, shipping (container ships, ferries, and recreational vessels), trawling and commercial fishing constitute other important sources of disturbance (Box et al. 2007; Diedrich et al. 2010).

Two species of seagrass are present in the study sites, namely *Posidonia oceanica* (L.) Delile and *Cymodocea nodosa* (Ucria) Aschers. *Posidonia oceanica* is considered a key species because of its extensive distribution in the littoral zone and the essential role this species plays in biological, biogeochemical and physical processes in the Mediterranean coastal areas (Fornes et al. 2006). Seagrass meadows are highly productive ecosystems that support a variety of fauna and flora as well as acting as a nursery and breeding ground for marine organisms (Boström et al. 2006). They also play an important role in coastal processes such as sediment deposition, attenuating currents and wave energy, and stabilizing unconsolidated sediments (Gacia et al. 1999). *Posidonia oceanica* is highly sensitive to deterioration in water quality and pollution is recognized as one of the greatest threats to this species. As a result of the preference of soft sediments by *Cymodocea nodosa*, its biggest threat is coastal erosion. Seagrass meadows are being lost at an alarming rate, particularly in the north-western part of the Mediterranean (Montefalcone 2009).

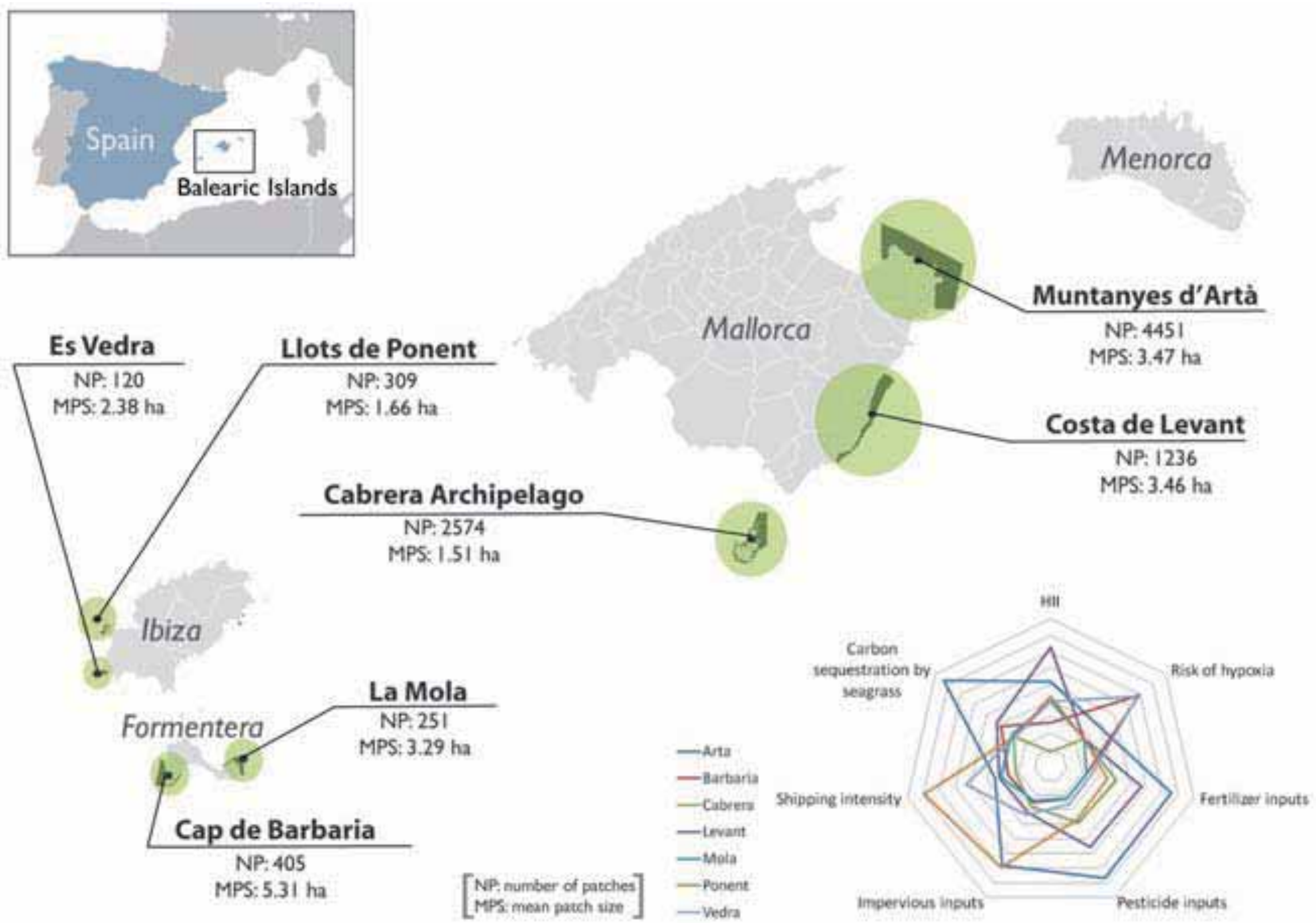

Figure 5.2: Carbon sequestration by seagrass was modelled in the seven study sites in the Balearic Islands. A radar chart shows the relative intensity of the disturbance variables for each study site.

5.2.2. Carbon captured by seagrass

Through the process of photosynthesis, seagrass plants combine CO_2, water and light to produce glucose for energy, releasing O_2 as a bi-product. As the glucose is metabolised, carbon molecules are absorbed by the plant and are stored in the leaves, roots and rhizomes. Carbon is also accumulated in seagrass meadows through particle trapping and sediment deposition. Accumulated carbon is stored in the soil under seagrass meadows and in dense rhizome mattes that can remain intact for hundreds, if not thousands of years. As accumulated carbon is a relatively small quantity for the spatial and temporal scale of this study, only carbon captured through metabolism was quantified.

The criteria for selecting a carbon capture model were: i) the model must include leaf, root and rhizome storage; ii) the model must be representative of the study area; and iii) the species of seagrass present in the study area should be accounted for.

Carbon stocks were calculated at patch level based on seagrass species, the quality of the meadow and the patch size. These carbon stock estimates were then summed to provide a total amount of carbon captured by seagrass classes in each study site. As two species of seagrass were present in the study sites, two species-specific models were chosen. The model selected for *Posidonia* oceanica was based on the studies by Alcoverro et al. (2001b) and Fourqurean et al. (2007). The biomass values were based on the number of shoots per square metre. The amount of carbon captured by the leaves, rhizomes and roots is given as a percentage of the dry weight. The average shoot density of *P. oceanica* meadows was estimated to be 625 shoots m^{-2} (Terrados & Medina-Pons 2011). Degraded meadows were assumed to be 75% of the density of healthy meadows, which is 469 shoots m^{-2}. Equation 5.1 was used to calculate the megagrams of carbon captured by *P. oceanica* meadows (C_{Po}). Table 5.1 shows the values used for the leaf, root and rhizome biomass and the corresponding carbon capture for each.

Equation 5.1: Carbon capture (Mg) by Posidonia oceanica meadows was calculated as the sum of the percentages of carbon contained in the leaf, rhizome and root biomass (g dry weight) for the area (m^2).

$$C_{Po} = \frac{(biomass_x \times shoots\ m^{-2} \times area) \times ({}^{\%C_x}/_{100})}{10^6}$$

Table 5.1: The carbon captured by Posidonia oceanica *meadows was based on the percent carbon of the dry biomass calculated by Alcoverro et al. 2001 and *Fourqurean et al. 2007.*

	Biomass (g DW shoot^{-1})	% C	Average g C m^{-2} for healthy meadows	Average g C m^{-2} for degraded meadows
Leaf*	0.57	37.8	135	101
Rhizome	0.79	38.4	190	142
Root	0.47	41.5	122	91

The study by Pérez & Romero (1994) was used to estimate the carbon captured by *Cymodocea nodosa*. The biomass and carbon content were calculated for the dry weight of the leaves, rhizomes and roots under different nutrient concentrations in their study. The biomass values for medium nutrient concentration were chosen to represent the conditions in the Balearic Island study sites. Nutrient concentrations are strongly seasonal, therefore a value corresponding to the average annual concentration was deemed appropriate. As in the *Posidonia oceanica* model, this carbon capture model was based on the biomass of leaves, roots and rhizomes, the difference being that the biomass was assessed per square meter rather than based on the number of shoots. Degraded meadows were included in the analysis under the assumption that the density of the seagrass meadows was 75% of the healthy meadows. The mass (Mg) of carbon captured by the leaves, rhizomes and roots of C. nodosa was estimated using equation 5.2.

Equation 5.2: The quantity of carbon captured by Cymodocea nodosa *(in Mg) was calculated as the sum of the percentages of carbon stored in the biomass of the leaves, rhizomes and roots (g dry weight m^{-2}) for the area of the seagrass meadows (m^2).*

$$C_{Cn} = \frac{(biomass_x \times area) \times (^{\%C_x}/_{100})}{10^6}$$

Table 5.2: The carbon captured by Cymodocea nodosa *meadows was calculated using the biomass and percent carbon values derived by Pérez & Romero 1994.*

	Biomass (g DW m^{-2})	% C	Average g C m^{-2} for healthy meadows	Average g C m^{-2} for degraded meadows
Leaf	319	34	108	81
Rhizome	263	42	110	83
Root	125	32	40	30

5.2.3. Spatial pattern metric selection

Benthic habitat maps were obtained from the Posidonia LIFE project for the seven study sites in the Balearic Islands at a scale of 1:1,000 (Posidonia Life, 2003). The cartography was performed using a side-scan sonar technique for areas between 5 and 40 m deep and orthophotos for areas at depths between 0 and 5 m.

Spatial pattern metrics were calculated at class level for the all of the benthic habitats present in the seven study sites. These included *Posidonia oceanica*, *Cymodocea nodosa*, precoralligenous communities and coralligenous communities, detritus, fine and coarse sediments, rocky bottoms and photophilic and sciaphilic algae. The ^{V}LATE 1.1 (Tiede 2005) vector-based extension was used to calculate the spatial metrics in ArcGIS 9.3 (ESRI 2009). A total of 14 spatial metrics were extracted that quantify area, form, edge, diversity, subdivision and proximity (Table 5.3). The detailed descriptions and equations used to calculate the spatial metrics are given in Appendix A.

Table 5.3: A total of 14 spatial pattern metrics were calculated using vector-based software. The metrics provide information at patch (P), class (C) and seascape (S) level.

	Abbreviation	Spatial pattern metric	Level
Area	NP	Number of patches	C, S
	CA	Class area	P, C, S
	MPS	Mean patch size	C, S
	PSSD	Stand deviation of patch size	C, S
Form	MSI	Mean shape index	P, C, S
	MPAR	Mean perimeter to area ratio	P, C, S
	MFRACT	Mean fractal dimension	P, C, S
	TE	Total edge	C, S
	MPE	Mean patch edge	P, C, S
Diversity	PROP	Proportion	S
Subdivision	DIV	Division	C
	SPLIT	Splitting index	C
	MESH	Mesh	C
Proximity	MNN	Mean Nearest Neighbour	C

5.2.4. Disturbance data

Spatial maps of the disturbance variables were downloaded from the National Center for Ecological Analysis and Synthesis (NCEAS) project on the human impacts on Mediterranean marine ecosystems (Halpern et al. 2008; Micheli et al. 2013). The rasters were projected in Lambert Azimuthal Equal Area projection centred on the Mediterranean sea, however further georeferencing using control points was necessary. A mask was used to crop the rasters to the study area and data was extracted at patch level in ArcGIS 9.3 (ESRI 2009).

Eight disturbance variables were selected based on the resolution of the spatial data and the relevance of the variable as a common form of disturbance in the Balearic Islands (Figure 5.1). The disturbance variables chosen were: (a) commercial shipping, (b) fertilizer inputs, (c) imperviousness (pollution from urban runoff), (d) pesticide inputs, (e) risk of hypoxia, (f) distance to area at risk of hypoxia, (g) the Global Human Influence Index (HII), and (h) the distance to the area with an HII score. A description of the disturbance data is given in Table 5.4.

Table 5.4: Eight disturbance variables and two additional variables (depth and distance to shore) were used in this study.

Variable	Description	Unit	Resolution
Shipping[1]	The intensity of commercial shipping was calculated based on the number of ships per year.	Ships yr^{-1}	Lines
Fertilizers[1]	The nutrient inputs were based on the annual use of fertilizers modelled by Halpern et al. (2008).	N+P205 tonnes / 1000 Ha	$1km^2$
Impervious surface pollution[1]	The non-point source inorganic pollution inputs from impervious surfaces (urban runoff) were modelled by Halpern et al. (2008).	Tonnes / 1000 Ha	$1km^2$
Pesticides[1]	Inorganic pollution was modelled by Halpern et al. (2008).	Tonnes / 1000 Ha	$1km^2$
Hypoxia[1]	The risk of hypoxic events in low-oxygen conditions was modelled by Djavidnia et al. (2005).	Risk value between 0 (low) and 1 (high)	$1.5km^2$
Distance to hypoxic area[1]	The distance to an area with a risk of hypoxia was calculated using the 'Near' tool in ArcGIS 9.3 (ESRI 2009).	Metres	
HII[2]	The Global Human Influence Index is based on nine global data layers that cover human land use and infrastructure, human population pressure and human access.	Index values ranging from 0 to 64	$1km^2$
Distance to area with HII	The distance to an area with an HII score was calculated using the 'Near' tool in ArcGIS 9.3 (ESRI 2009).	Metres	
Depth	Bathymetry data was obtained at a coarse resolution for the entire study area and where possible, at a fine resolution for individual study sites.	Metres	5-10m
Distance to shore	The distance to the shore was calculated using the 'Near' tool in ArcGIS 9.3 (ESRI 2009).	Metres	

[1]http://www.nceas.ucsb.edu/globalmarine/mediterranean

[2]http://sedac.ciesin.columbia.edu/data/set/wildareas-v2-human-influence-index-geographic

5.2.5. Data analysis

A regression tree analysis was used to explain the variation in the carbon stocks (as the response variable) using explanatory variables that described seascape structure (i.e. spatial pattern metrics) and anthropogenic disturbance. This approach incorporated data at three hierarchical levels. Spatial pattern metrics represent the lower level that determine carbon stocks (focal level), while anthropogenic disturbance limits carbon capture at the upper hierarchical level. Regression tree analysis uses combinations of explanatory variables to split data into homogenous groups that describe the variation in the response variable (De'ath & Fabricius 2000). An exhaustive search procedure was used to split the data in such a way that the homogeneity of the two resulting groups was maximized with respect to carbon capture. In this way, the impurity of the data is reduced (Therneau & Atkinson 1997). A V-fold cross-validation using ten random sub-samples and a standard error rule of 1 was used to determine the optimal tree. The tree with a relatively low cross-validation cost (CV cost) and node complexity was chosen because it incorporated minimal impurity while explaining the complexity of the ecological system.

5.3. Results

The regression tree (Figure 5.3) illustrated the complexity of the interactions that determined seagrass carbon stocks in the study sites. The most important splitting variable was fertilizer. In areas where the fertilizer concentration exceeded 9.77 kg ha^{-1}, seagrass carbon stocks were highest. Below this concentration, class area was an important determinant of carbon capture. This was largely due to class area being an input in the carbon capture models. The regression tree split class area into three groups, namely classes smaller than 137 ha, classes between 137 and 332 ha, and classes larger than 332 ha. In the third category (i.e. >332 ha), the human influence index (HII) determined the next split. This variable describes the intensity of human influence on the coastline and the results showed that areas with lower HII accounted for lower carbon stocks. The mean perimeter:area ratio (MPAR), which is an indicator of the complexity of patches, determined the next node which showed that the larger the MPAR, the higher the carbon captured. This metric was reflecting the quality of seagrass meadows. Continuous patches have more carbon stocks than degraded meadows. Fertilizer and class area emerged again as important determinants.

For the smaller class area category (classes < 136 ha), carbon stocks with less mean patch edge (MPE) were more important. The mean rating for low concentrations of fertilizer (less than 5.6 kg ha^{-1}) showed reduced carbon capture. The final split was determined by shipping intensity. Seagrass carbon stocks were higher in areas with low shipping intensity.

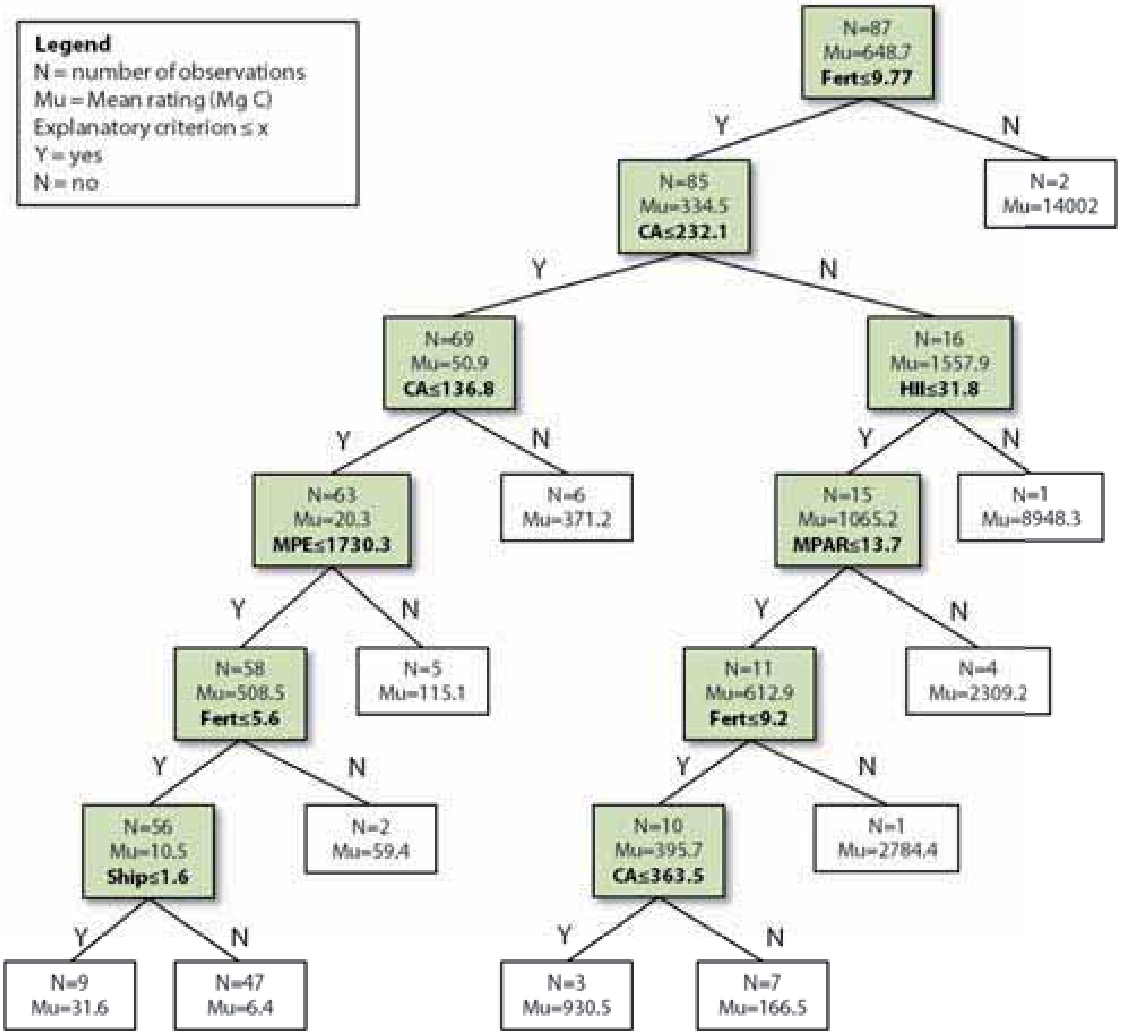

Figure 5.3: The regression tree for seagrass carbon stocks shows the number of observations (N) and the mean rating (Mu) of the nodes and leaves. The leaves (white boxes) are homogenous groups. Each node (in green) also includes the criteria (in bold) used to split the data. The left branches (denoted 'Y') correspond to the data for which the criteria are true. The higher the Mu, the more important the explanatory variable.

The relative importance of the explanatory variables were plotted in Figure 5.4. Fertilizer was the main contributor to seagrass carbon stocks, followed by class area, the proportion of the class and the total edge. Interestingly, the two protection variables, namely the IUCN protection category (IUCN) and whether the area was a Specially Protected Area for Birds (SPA) had the lowest levels of importance. This suggests that protected areas are not necessarily targeting the protection of seagrass patches, despite *Posidonia oceanica* being classified as a priority habitat type for conservation in the Habitats Directive (Dir 92/43/CEE).

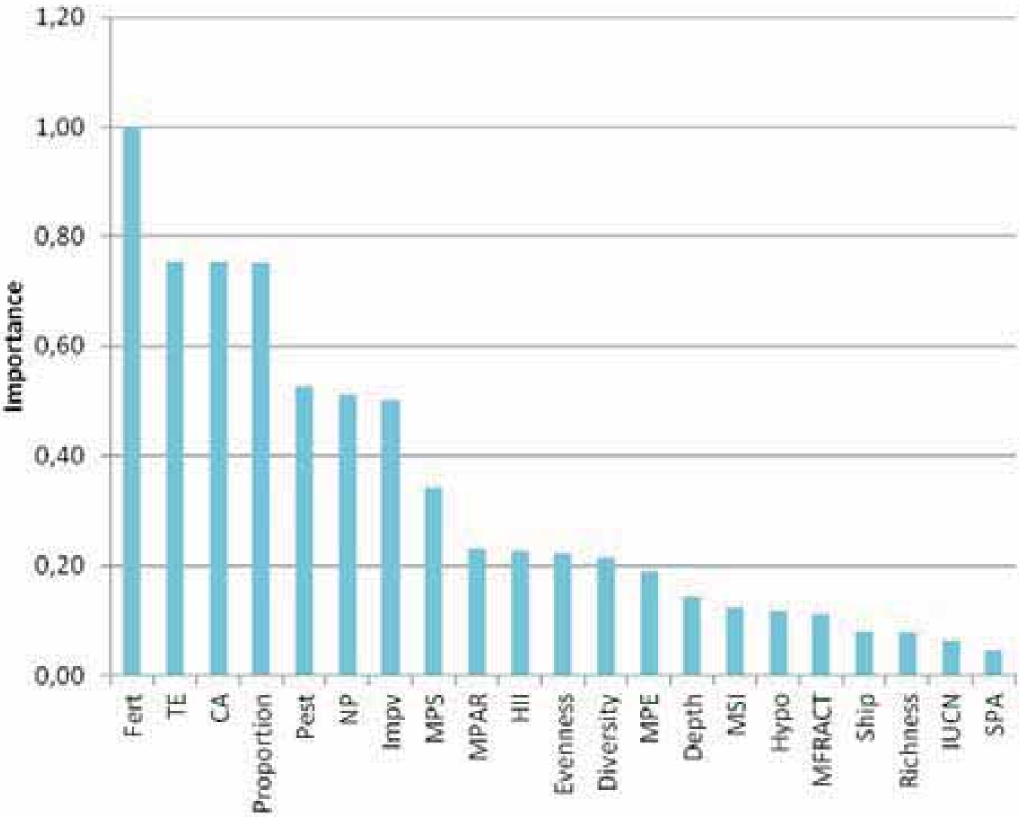

Figure 5.4: The relative importance of the explanatory variables shows that fertilizer plays an important role in carbon capture, while the protection level contributes little to seagrass presence.

5.4. Discussion

Understanding how the spatial arrangement of seascapes relates to the provision of ecosystem services is essential for the effective management of the heavily threatened coastal zone. This knowledge would assist managers and decision-makers who seldom have ecological information upon which to base their decisions. The need for information on the ecological consequences of spatial patterns is further exacerbated by the pressing need to address anthropogenic pressure in the coastal and marine environment (Pittman et al. 2011). In this chapter, I determined how the spatial patterns of the seascape and anthropogenic pressures influence seagrass carbon stocks using a seascape ecology approach.

The results of the regression tree analysis showed that fertilizer inputs in the marine environment were having a significant effect on the area of seagrass meadows. Interestingly, the effect was positive. Increased nitrogen and potassium appeared to enhance the area of seagrass meadows. Nitrogen and phosphorus are the two most important nutrients limiting the growth and production of both *Posidonia oceanica* and *Cymodocea nodosa* (Touchette & Burkholder 2000; Zhang et al. 2011). Studies in the north western Mediterranean Sea have shown that artificially adding moderate amounts of phosphorus and nitrogen can increase seagrass biomass (Alcoverro et al. 1997; Pérez et al. 1991; Udy & Dennison 1997). Large inputs of nitrogen, on the other hand, have detrimental effects on seagrass and can reduce carbon storage (Invers et al. 2004). The results in chapter 4 of this dissertation found that pesticides and fertilizers were associated with degraded habitats and higher levels of fragmentation in the same study sites.

Two other disturbance variables were found to play a role in determining carbon stocks, namely the intensity of human presence (HII) and the number of ships passing through the study areas. Counter-intuitively, the human influence index also showed a positive relationship with carbon stocks. This suggested that carbon captured was higher in areas where human activity was highest. Coastal development has been shown to have severe negative consequences on seagrass meadows (Montefalcone et al. 2010; Ralph et al. 2006) contradicting the spatial patterns observed in this chapter. The most likely explanation for this is the presence of a very large patch (2006 ha in size) of *Posidonia oceanica* in one of the most populated study sites, Costa de Llevant. This patch occurs in the north of the study sites where steep cliffs along the coastline protect it from human influence. The Spearman's correlation coefficient was calculated to determine the effects of the disturbance variables on seagrass only (i.e. the mosaic of habitats was excluded) and the results revealed that the relationship between HII and carbon stocks was not statistically significant (Table 5.5).

Table 5.5: Spearman's rank correlation coefficients identified statistically significant relationships between the seagrass carbon stocks and the disturbance variables at class level. Significant correlations at p>0.05 are highlighted.

	Carbon stock
Commercial shipping intensity	-0,069158
Fertilizer inputs	**0,337911**
Impervious inputs	0,262187
Pesticide inputs	**0,414518**
Risk of hypoxia	0,219669
Distance to area at risk of hypoxia	-0,189408
Human Influence Index	0,111014
Distance to area with HII	0,158332

The regression tree showed that shipping had a negative effect on carbon stocks. Again, this relationship was not statistically significant when other habitat types were excluded from the analysis (Table 5.5). The Spearman's correlation coefficient analysis revealed that pesticides also played a significant role in determining carbon stocks. Like fertilizers, pesticides appeared to enhance seagrass area and therefore carbon capture. This may be because glyphosates like Roundup, contain growth hormones that can improve seagrass growth and primary productivity when concentrations are diluted, however little is known about the effects of pesticides on seagrass meadows (Alcoverro et al. 2001a; Bester 2000; Udy & Dennison 1997). Ralph (2000) found that glyphosates had no significant effects on the photosynthetic capacity of *Halophila ovalis*, a widely-distributed seagrass. Studies have been inconclusive, but generally showed high levels of pesticides in areas where seagrass meadows had been degraded or destroyed (Bester 2000).

The positive relationship between fertilizers and seagrass area exposes an interesting quandary for the protection of seagrass in the study sites. Evidently, organic and inorganic nutrient inputs can be favourable for seagrass meadows and therefore also for the ecosystem services they provide. Excessive nutrients can, however, be detrimental. The results of chapter 4 suggest that while the total area covered by seagrass appears to be favoured at the current concentrations of fertilizers and pesticides, these seagrass meadows tend to be degraded and fragmented. Continued monitoring of the situation is recommended.

In conclusion, the seascape ecology approach to linking ecosystem service delivery to the anthropogenic pressures driving seascape patterning holds much potential for target-specific responses to enhance the protection of ecosystem services. This study is a first step towards exploring the influence of disturbance on ecosystem services. The modelled data used in the analysis may differ from actual in-field disturbance levels or carbon stocks, however it serves to demonstrate a technique that can be applied to more accurate data, should it become available. Carbon stocks have a potential market which means that information on the effects of disturbance on carbon capture can be used to justify conservation efforts in socio-economic terms. Quantifying the economic value of the carbon stocks would be an interesting next step for similar studies.

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CHAPTER 6

Conclusions and recommendations

The high productivity and diversity of coastal ecosystems, coupled with some of the most heavily impacted coastlines in the world amplify the urgency of conserving the biodiversity of the Mediterranean Sea. A lack of data and the high levels of complexity of these ecosystems are a significant challenge for the effective management and conservation of coastal and marine ecosystems. Understanding the causes and consequences of spatial patterns is essential for target-specific responses to anthropogenic disturbance. The emerging field of seascape ecology has the potential to bridge this knowledge gap using techniques developed for landscape ecology to interpret spatial data and provide ecologically meaningful information on spatial patterns.

In this dissertation, I explored the anthropogenic causes and ecological consequences of spatial patterns in Mediterranean coastal seascapes. The application of seascape ecology techniques to quantify the structure of benthic habitat mosaics and the ecological relevance of these techniques for conservation planning and informing ecosystem-based management was determined through three studies.

In the first study, the relationship between seascape structure, habitat diversity and species richness was explored. Spatial pattern metrics had been used to link species to spatial patterns in other parts of the world (Meynecke et al. 2008; Pittman et al. 2004; Pittman et al. 2007a; Pittman et al. 2007b). Two studies by Garrabou et al. (2002; 1998) had successfully applied seascape ecology techniques in the Mediterranean, however these studies were done at fine-scales and not at the landscape scale required for management purposes. The application of spatial pattern metrics to coastal Mediterranean seascapes at the landscape scale was explored in chapter 3. This study was necessary to ensure that the quantification of seascape structure could provide ecologically meaningful data as chapter 4 and 5 used the same method.

The structure of the seascape has been shown to influence the movement of energy, materials and organisms (Pickett & Cadenasso 1995). Anthropogenic pressures alter the seascape structure, thereby changing the fluxes of these movements and disrupting key ecological processes (Ernoult et al. 2003; Pickett & Cadenasso 1995). Information on the ecological consequences of anthropogenic disturbance is necessary for the effective management of coastal seascapes. I used the effects of anthropogenic disturbance on seascape structure, which I quantified in chapter 4, to explore the impact on an ecosystem service in chapter 5. Spatial pattern metrics and multivariate analysis were used to assess the influence of different disturbance variables on specific components of the seascape and then link them to an indicator ecosystem service, namely seagrass carbon stocks. This study was a first step towards understanding the consequences of anthropogenic pressures on ecosystem service delivery in coastal Mediterranean seascapes.

Main findings

In summary, the main findings of the studies showed that biodiversity was determined by the area of each habitat, which is negatively affected by shipping and positively related to fertilizers and pesticides.

Species richness was determined by habitat diversity, which is in turn driven by the class area, mean patch size and the edge metrics. A higher abundance and variety of habitats has the potential to support more species. Patch shape complexity also contributed to increased species richness. Studies have shown positive correlations between rugosity (i.e. the habitat available for colonization, foraging and shelter for organisms) and species richness, supporting the findings (Gratwicke & Speight 2005).

Land-based pollutants have a significant influence on the abundance of habitats. While fertilizer and pesticides appeared to increase the class area of habitats like the seagrass, *Posidonia oceanica*, these meadows tended to be degraded and dispersed. The total area covered by seagrass was the major determinant of seagrass carbon stocks, however a decrease in the quality of seagrass meadows as a result of pollutants would correspond to less carbon capture. The positive effect of fertilizers and pesticides on carbon stocks identifies an interesting predicament for managing these meadows.

In conclusion, landscape ecology quantification techniques were found to be equally applicable and relevant to coastal Mediterranean seascapes. Spatial pattern metrics have the potential to be a valuable conservation tool for assessing the causes and consequences of spatial patterns.

Management recommendations

The focus of this dissertation has been on the application of seascape quantification techniques as a conservation tool. The effectiveness of this tool in providing information on biodiversity and ecosystem services based on digital benthic habitat maps has been demonstrated. Using GIS-based spatial pattern metrics to interpret spatial data is a cost and time-effective approach that can be used to identify priority conservation areas and provide a sound ecological foundation for management decisions. Furthermore, this technique can be used to assess the effects of disturbance on coastal seascapes making it a valuable decision-support tool for marine spatial planning, which is one way to implement ecosystem-based management (Collie et al. 2012).

In terms of managing biodiversity, the abundance of different habitats should be a management priority rather than simply the number of habitat types. Patch shape complexity is also an important factor for species richness. To meet the objective of Habitats Directive and the Marine Strategy Framework Directive to conserve biodiversity, the maintaining the complexity of patches should be a conservation priority.

The findings of the two chapters on anthropogenic disturbance showed that land-based pollution has the largest influence on seascape structure and ecological processes. While fertilizers and pesticides appeared to have a positive effect on the area of habitats, further exploration showed that these pollutants were linked to degraded habitats. This may be a result of inaccuracies in the modelled data, however monitoring of the effects of land-based pollution is strongly recommended, especially in Muntanyes d'Artà and the Cabrera Archipelago where land-based inputs are estimated to be highest. An assessment of whether land-based pollutants are causing habitat degradation or promoting the colonization of un-vegetated areas would discern whether fertilizer and pesticides are positive inputs into coastal ecosystems or degrading habitat quality.

Future research

Results of the studies conducted in this dissertation suggest that future research should explore the cross-scale effects of disturbance of seascape structure and the associated ecological consequences. Spatial pattern metrics are applicable at all scales, however the availability of fine-scale data was the limiting factor in these studies. An assessment of the importance of patch/habitat context with respect to disturbance would also be interesting, should fine-scale disturbance data become available. A possible approach to this problem would be to do in-field samples of fertilizers, pesticides and other types of pollution. Using techniques like regression tree analysis that can assess data of different scales, in-field data and coarser resolution spatial data could be combined to better model coastal ecosystem patterns. Adding a temporal scale is also suggested for future research. The literature indicated strong seasonality in the study area, which could also be included in models to make better management recommendations. Including environmental variables like salinity, pH, water temperature and water clarity could also benefit similar studies.

Aside from the technical improvements that future studies should consider, a pressing knowledge gap that this study touches on is the need to quantify marine and coastal ecosystem services and to understand how anthropogenic pressures affect their delivery. I present a first step towards developing a method for doing this in chapter 5, however making realistic predictions based on a modelled ecosystem service is challenging. The inclusion of real carbon stocks that reflect the natural variability of their environment would be a necessary step towards a more realistic assessment of the effect of disturbance on ecosystem service provision. Furthermore, the potential market for blue carbon means that the delivery of ecosystem services could be valued. Valuation of ecosystem services is of the utmost importance for justifying conservation efforts, especially in the coastal and marine environment.

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