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Advancing systematic conservation planning in the Mediterranean Sea

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Abstract

Marine biodiversity is globally declining due to a plethora of anthropogenic threats. Some of these threats are local, such as light pollution and coastal development, while other threats such as climate change operate at larger spatial scales. A range of conservation actions need to occur to effectively mitigate these mounting threats. Due to constraints on time and money for conservation actions we must set spatial priorities. These priorities need to achieve conservation goals and have a high chance of success.

The aim of this thesis is to develop approaches that improve the spatial prioritisation of marine biodiversity conservation. Implementation of conservation plans is largely dependent on the cohesion of conservation objectives within the larger economic and social context that it lies within. In this thesis, I use the complex setting of the Mediterranean Sea to explore and propose innovative systematic prioritisation approaches. The Mediterranean Sea is a global biodiversity hotspot surrounded by over twenty countries. In this region threats to biodiversity and ecosystems are high and regional conservation plans based on systematic planning are still limited, providing an ideal system for investigating novel conservation planning approaches. This thesis is composed of seven chapters, which address three key themes for improving spatial conservation prioritisation.

Chapter 1 is a broad introduction to the thesis. This chapter examines previous applications of conservation planning and prioritisation, highlighting gaps and limitations of current approaches. I introduce systematic conservation planning, applying it to the marine realm and specifically the Mediterranean Sea. Chapters 2 to 6 each present an approach to spatial conservation prioritisation relevant to protected area design in the Mediterranean Sea.

The first theme (Chapters 2 and 3) explores ways to improve conservation efficiency in the marine realm. In **Chapter 2** I present the first study that quantifies the increase in cost-efficiency of collaborative conservation in the marine realm. This approach can help deliver efficient conservation outcomes when planning spatially explicit actions within marine environments shared by many countries. **Chapter 3** examines the importance of cost in marine conservation planning. I develop an approach for addressing cost when planning large-scale marine protected areas networks that span across multiple countries. I reveal that area is a poor cost surrogate for conservation cost in marine systems and that the most effective surrogates are those that account for multiple sectors or stakeholders.

The second key theme of the thesis (Chapters 4 and 5) addresses the issue of adequately protecting species in conservation planning. In **Chapter 4**, I address the importance of incorporating species migration information into conservation planning. To ensure that species are adequately protected it is crucial to underpin conservation planning by the biological life-stages of species. In **Chapter 5**, I explore how to determine species threats using remote sensing and satellite imagery. I investigate a case study of nesting sea turtles in the eastern Mediterranean, and show that artificial night lights can affect their spatial nesting patterns. This study reveals the importance of satellite night-time imagery for conservation purposes. It also defines the first step of any conservation plan that strives to adequately protect species from threatening processes.

The third theme (covered in Chapter 6) aims to improve implementation success. **Chapter 6** applies systematic zoning tools to a country's entire territorial waters, aiming to protect biodiversity when faced with multiple marine activities. Specifically, I quantify the trade-offs between conservation and economic objectives. The case study in this chapter is relatively complex, allowing for multiple zones and costs; it shows that prospective offshore hydrocarbon resources can have a very large influence over conservation plans.

Finally, **Chapter 7** is a synthesis of the thesis. I address the contributions of this research towards advancing marine conservation prioritisation both in the Mediterranean region and globally. Unifying my findings from this thesis, I propose additional steps to improve the framework of systematic conservation planning when applied to the marine realm. This thesis advances the theory of marine prioritisation, but also delivers practical outcomes, providing the first large-scale prioritisation of conservation actions for the Mediterranean Sea.

Overall, this research advances our knowledge of conservation prioritisation in the marine realm. It provides strategies and methods to improve systematic conservation planning efficiency, adequacy and implementation success. This thesis focuses on how to make good decisions regarding the selection of marine protected areas (MPAs) and priority areas for marine conservation in the Mediterranean Sea that can have implications for many other parts of the world.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

I have clearly stated the contribution of others to my thesis as a whole, including statistical assistance, survey design, data analysis, significant technical procedures, professional editorial advice, and any other original research work used or reported in my thesis. The content of my thesis is the result of work I have carried out since the commencement of my research higher degree candidature and does not include a substantial part of work that has been submitted to qualify for the award of any other degree or diploma in any university or other tertiary institution. I have clearly stated which parts of my thesis, if any, have been submitted to qualify for another award.

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Publications during candidature

Mazor, T., Possingham, H.P., Edelist, D., Brokovich, E. & Kark, S. (2014). The crowded sea: Incorporating multiple marine activities in conservation plans can significantly alter spatial priorities. *PLoS ONE*, **9**, e104489. doi:10.1371/journal.pone.0104489.

Mazor, T., Giakoumi, S., Kark, S. & Possingham, H.P. (2014). Large-scale conservation planning in a multinational marine environment: cost matters. *Ecological Applications*, **24**, 1115-1130.

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- Incorporated as Chapter 2

Contributor	Statement of contribution
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Contributions by others to the thesis

Chapters 2, 3, 5 and 6 of this thesis consist of published papers. Chapter 4 is in preparation for submission to a journal. I have retained the text consistent with their published form. I use the plural first-person pronoun “we” and I refer to my own work within these chapters as per their published format (e.g., Mazor et al. 2013). In Chapters 1 and 7, where I introduce, summarise and interpret Chapters 2–6, I use the singular first-person pronoun “I” and refer to my own work by chapter number. This format is consistent with the fact that Chapters 1 and 7 are my own work, whereas Chapters 2 – 6 are collaborative papers which I initiated and led as the primary author.

Chapter 1

This chapter was written by the Candidate, with editorial input from Salit Kark and Hugh Possingham.

Chapter 2

Mazor, T., Possingham, H.P. & Kark, S. (2013). Collaboration among countries in marine conservation can achieve substantial efficiencies. *Diversity and Distributions*, **19**, 1380-1393.

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Chapter 3

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This chapter is taken from a manuscript by the Candidate, MB, JM, HPP and SK, which is soon to be submitted. The idea for the paper was conceived by the Candidate, SK and HPP, and refined with the assistance of MB and SK. The Candidate collected the data and carried out the analysis with advice from MB and JM. The Candidate wrote the paper with editorial input from all other authors.

Chapter 5

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- Contributions stated above.

Chapter 6

Mazor, T., Possingham, H.P., Edelist, D., Brokovich, E. & Kark, S. (2014). The Crowded Sea: Incorporating multiple marine activities in conservation plans can significantly alter spatial priorities. *PLoS ONE*, **9**, e104489. doi:10.1371/journal.pone.0104489.

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Appendix 1

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This appendix was co-authored by the Candidate and is published in BioScience. The idea for the paper was conceived by all authors during a workshop. NL, AT and the Candidate collected data. NL, AT, AG and the Candidate conducted the analysis. NL led the writing with contributions and editorial input from all authors.

Appendix 2

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This appendix was co-authored by the Candidate and is published as a review in Reviews in Fish Biology and Fisheries. The idea for the paper was conceived by all authors during a workshop organised by SK, GS and the Candidate; and chaired by SK and HPP. GS and TM led the writing with contributions and editorial input from all authors.

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כמה נאים ומשובחין הן וכל מה שבראתי בשבילך בראתי, תן דעתך שלא תקלקל ותחריב את
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Keywords

adequacy, cost-effective, implementation success, marine conservation planning, marine reserve network, Mediterranean Sea, marine protected areas (MPAs), spatial conservation prioritisation, systematic conservation planning, threatened marine species.

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List of Figures and Tables

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List of Abbreviations used in the thesis

CBD	Convention on Biological Diversity
CSM	Connectivity Strength Modifier
EEZ	Exclusive Economic Zone
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GDP	Gross Domestic Product
GFCM	General Fisheries Commission for the Mediterranean
GLM	Generalised linear model
GPS	Global Positioning System
GSA	Geographical Sub Areas
IMF	International Monetary Fund
IUCN	World Conservation Union or International Union for the Conservation of Nature and Natural Resources
MCP	Marine Conservation Planning
MPA	Marine Protected Area
MSP	Marine Spatial Planning
NIS	New Israel Shekel
NPA	Israel Nature and Parks Authority
SAUP	Sea Around Us Project
UNEP	United Nations Environment Programme

Chapter 1

Introduction

Rosh Hanikra, Israel. Photo credit: T.Mazor

1.1 The evolution of biodiversity conservation

The world is facing a rapid loss of biodiversity. The rate of species extinction has increased to ~100-1000 times faster than background extinction rates (Pimm et al. 1995; Balmford 1996; Pimm et al. 2014). Several studies have predicted that up to fifty-percent of all species will become extinct within the next fifty years (Pimm & Raven 2000; Thomas et al. 2004). While the rate of species extinction is debatable (Adams 2009), there is widespread scientific consensus that such losses are almost entirely caused by human activities (Prance 1991; Smith et al. 1993; Vitousek et al. 1997; Chapin III et al. 2000; Ceballos & Ehrlich 2002; Hooper et al. 2005; Sodhi et al. 2008; Gonzalez-Suarez & Revilla 2014). Causes of species decline have been well documented, with those of primary significance being species overexploitation (e.g., fishing and hunting), habitat loss and fragmentation, invasive species, and climate change (Diamond 1989; Pimm & Raven 2000; Brooks et al. 2004; Thomas et al. 2004; Butchart et al. 2010; Dawson et al. 2011). Unless there is immediate action to address species extinction, losses will continue to escalate (Balmford 1996; Ricketts et al. 2005; Martin et al. 2012). Out of the need to protect, conserve and halt species from extinction, the scientific discipline of conservation biology emerged (Soulé 1985; Sarkar et al. 2006).

Known as a "crisis" discipline, or a discipline with a "deadline", conservation scientists must act quickly to determine the most effective and efficient methods for addressing the challenge of preserving the natural world and its species (Soulé 1985; Myers 1993; Pullin 2002). There are a range of optional strategies for conserving biodiversity, such as ecosystem restoration, community engagement and invasive species control. However, the designation of protected areas is recognised as one of the most effective and successful strategies for protecting biodiversity on land and in the sea (Bruner et al. 2001; Chape et al. 2005; Possingham et al. 2006; Lester et al. 2009).

Protected areas are by no means a new conservation strategy. The reservation of areas to maintain their inherent quality has been a long-standing practise (evidence from the second and third centuries BC; Grove 1995) across the globe for religious practises (e.g., sacred groves), food resources, hunting and animal management (MacKenzie 1988; Nelson 1991; Chape et al. 2005). National parks and protected areas, as known today, emerged in the mid-nineteenth century in Europe and North America (Mcneely 1994b; Pullin 2002; Primack 2010). However, the establishment of Yellowstone National Park in the USA in 1872 is often referred to as the beginning of modern protected areas (Mcneely 1994a; 1994b; Primack 2010). Since then, and up until the 1960s, protected areas were established with minimal scientific input. The *ad hoc* approach to protected area establishment resulted in a biased sample of biodiversity, favouring the protection of habitats that are not threatened and reserves that favour ease of establishment – places in remote areas that minimise conflict with other natural resource users (Pressey et al. 2002; Watson et al. 2011; Devillers et al. 2014).

Given the limited resources available for conservation, conservationists must be selective (Pullin 2002; Bottrill et al. 2008). Hence, considerable attention has focused on the need to improve the efficiency of protected area selection. The first scientific concepts of reserve design emerged from MacArthur and Wilson's (1967) pioneering studies on island biogeography. Their research introduced some fundamental ecological principles such as the importance of reserve size for species persistence. Another historical milestone was the establishment of protected area categories by the International Union for Conservation of Nature (IUCN 1978) in the 1970s. This international classification helped consolidate worldwide protected area policy and management (Ravenel & Redford 2005). Further advancements and attempts were made during the 1980s to use selection algorithms for protected area network design (Kirkpatrick 1983). However, much progress was made possible by the emergence of technological advancements such as computational speed, high-resolution Geographic Information System (GIS) data and remote sensing tools which expanded the field of conservation biology with spatial concepts and theory (Wilson et al. 2009). Such developments have led to spatial prioritisation and systematic conservation planning; a framework developed to improve the efficiency of protected areas and networks (Margules & Pressey 2000; Watson et al. 2011).

Conservation planning requires a systematic approach to reap effective outcomes (Margules & Pressey 2000; Smith et al. 2006; Pressey et al. 2007). Margules and Pressey (2000) clarified the steps in systematic conservation planning, a flexible and defensive framework for selecting protected areas that is driven by explicit goals and objectives. This process began with six stages and has since been expanded by Pressey and Bottrill (2009) to eleven stages (Fig. 1.1). The underlying aim of this framework is to identify priority conservation areas that are *comprehensive* in sampling every kind of biodiversity, *representative* across the full range of variation of each biodiversity feature, e*fficient* at achieving objectives for minimal cost and *adequate* at ensuring the persistence of biodiversity features (Margules & Pressey 2000; Possingham et al. 2006; Margules & Sarkar 2007). Most systematic conservation planning exercises have focused on comprehensiveness and representativeness (Margules & Pressey 2000; Olson & Dinerstein 1998), and only recently have they begun to address the principles of efficiency and adequacy (Sarkar & Illoldi-Rangel 2010). This thesis focuses on spatial conservation prioritisation applied to protected area design (incorporating stage 1 through to stage 9; Fig. 1.1), and aims to address the principle of *efficiency* in

Chapter 2 and 3, *adequacy* in Chapter 4 and 5, and presents a newly evolving principle; identifying priority conservation areas that address *implementation success* in Chapter 6.

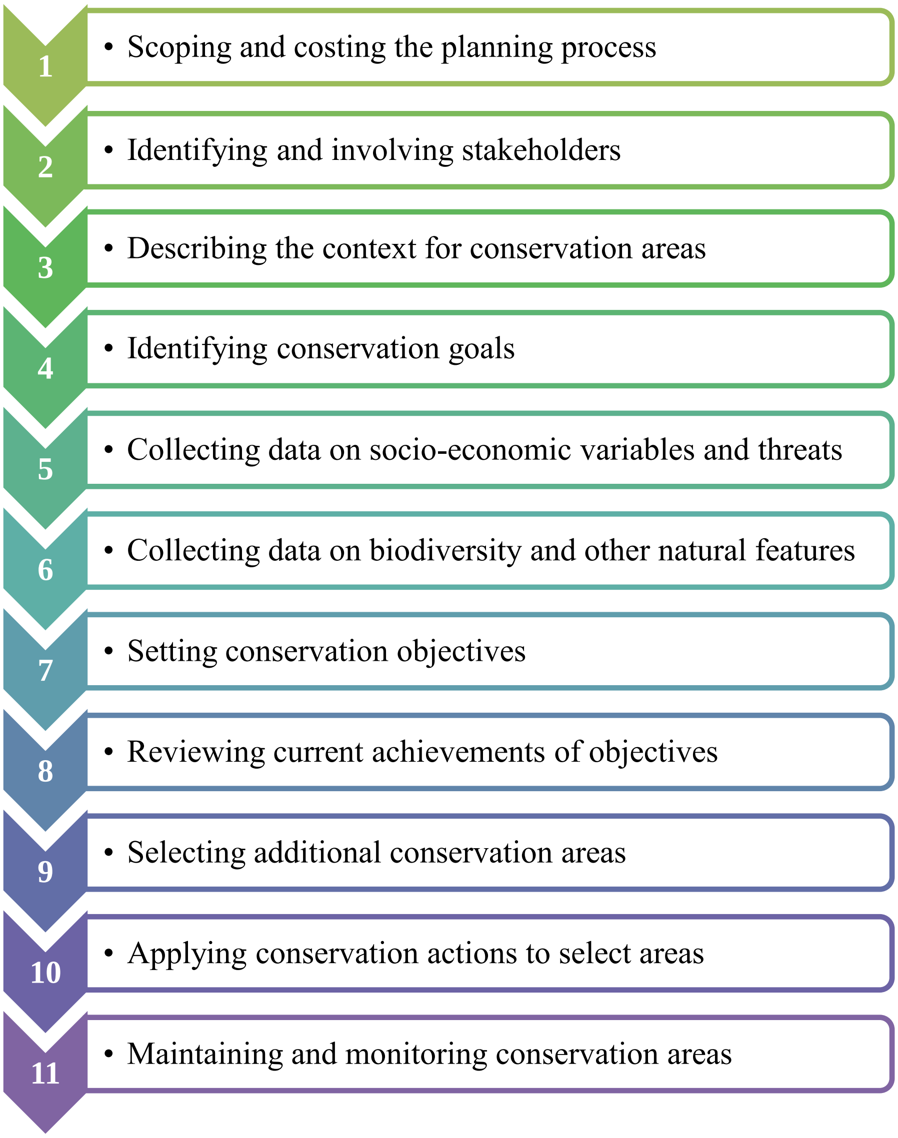

Figure 1.1. Framework for systematic conservation planning (Margules & Pressey 2000; Pressey & Bottrill 2009).

1.2 Spatial conservation prioritisation

Spatial conservation prioritisation aims to strategically identify locations for allocating scarce conservation resources for conservation action (Macdonald & Willis 2013). This approach uses spatial analysis of quantitative data to determine conservation investment locations (Wilson et al. 2009). Prioritisation approaches have greatly improved the selection of protected areas, and have also been applied to a variety of other conservation actions such as invasive species management and restoration (Wilson et al. 2007).

The first approaches developed for setting spatial priorities for conservation use a simple scoring method (Game et al. 2013). Scoring based approaches evolved in the 1970s, scores are assigned to sites based on attributes such as: habitat condition, species richness or rarity, size, diversity, naturalness and anthropogenic threats (Margules & Usher 1981; Mittermeier et al. 1998, Myers et al. 2000). The highest priority sites are generally those that have the highest sum of scores (Murdoch et al. 2010). Although such strategies are an improvement from the *ad hoc* selection of conservation sites, scoring approaches have been criticised for their limitations (Pressey & Nicholls 1989; Game et al. 2013). Scoring-based approaches are not able to recognise the complementarity between sites. Thus, there is no guarantee that all species or even most species will be included in the selected conservation sites or reserves (Kirkpatrick 1983; Orme et al. 2005). Scoring approaches are also arbitrary as they are underpinned by difficult societal value judgements that lack transparency (Murdoch et al. 2010; Game et al. 2013). Further limitations exist, and compared with other prioritisation methods scoring approaches are less efficient - more expensive and often fail to meet conservation targets (Pressey & Nicholls 1989; Pressey 1994; Underwood et al. 2008).

Despite the more effective methods that are available today, many conservation priorities are still determined by less effective scoring approaches (e.g., hot spots, multi-criteria analysis; Margules & Usher 1981; Usher 1986). These approaches disregard the key principles of spatial conservation prioritisation (Margules & Pressey 2000; Possingham et al. 2006). Moreover, scoring approaches are not solving a well-defined problem and ignore basic principles of decision science; where to identify priorities there must be an objective function that explicitly states the goal (clearly states what is being maximized or minimized), a set of specific actions to prioritise between, a model that relates these actions towards meeting the objectives and any defined resource constraints (Possingham et al. 2001; Game et al. 2013).

There are two well-accepted approaches for selecting priority areas that do overcome the limitations of scoring approaches. The first approach aims to deliver the greatest possible return on investment "maximisation approach" for a fixed budget. The second approach aims to achieve fixed conservation goals for the least cost, "minimum-set" approach (Brooks et al. 2006; Possingham et al. 2006; McBride et al. 2007; Murdoch et al. 2007; Vazquez et al. 2008; Funk & Fa 2010). These two approaches enable conservation priority areas to be chosen efficiently; using limited resources to achieve defined conservation goals (Margules & Pressey 2000; Ball et al. 2009; Funk & Fa 2010).

There are a variety of software packages that have been developed to support spatial conservation planning decisions using these two approaches (e.g., Marxan, C-Plan, Zonation, ConsNet; Ball et al. 2009; Carwardine et al. 2006; Sarkar et al. 2006; Moilanen et al. 2009). This thesis applies Marxan and its advanced versions to aid the selection of priority conservation areas in the marine realm. Marxan is one such decision support tool for systemic conservation planning that is based on the minimum set approach (Ball et al. 2009; Watts et al. 2009). It uses a simulated annealing algorithm to solve a well-defined mathematical problem (McDonnell et al. 2002). Marxan identifies sites that fulfil quantitative targets for biodiversity features in a compact system of protected areas for the least possible cost (Ball et al. 2009). An extension to Marxan is Marxan with Zones (Watts et al. 2009). The purpose of zoning for conservation is to determine trade-offs between conflicting interests. This tool enables the user to reach multiple conservation objectives by defining different conservation targets for specific areas (zones) (Ball et al. 2009; Smith et al. 2009; Watts et al. 2009; Klein et al. 2010).

Marxan and Marxan with Zones have been used for conservation planning in regions all around the world (Ball et al. 2009; Ardron et al. 2010). A prime example is the Great Barrier Reef Marine Park, where Marxan was originally created to help assist in its rezoning in 2004 (Lewis et al. 2003; Moilanen et al. 2009). Other examples where Marxan has been applied include marine protected areas for the Irish Sea (Lieberknecht et al. 2004), avian conservation in Northern America (Pearce et al. 2008), community-based protected areas in the Choiseul Province of the Solomon Islands (Game et al. 2011), a national network of marine protected areas in Palau (Hinchley et al. 2007), forest regeneration in Switzerland (Bolliger et al. 2010), protection of mammals in Central America (Jenkins & Giri 2008), coral reefs of the Coral Triangle (Klein et al. 2010), and freshwater fish of the Guadiana river basin (Hermoso et al. 2010). These examples highlight the flexible and dynamic method of systematic conservation planning for aiding conservation prioritisation of biodiversity within both terrestrial and marine ecosystems.

1.3 Conservation prioritisation in the marine realm

Awareness of biodiversity loss in the marine realm was recognised much later than the terrestrial realm (Allison et al. 1998; Salm et al. 2000; Roberts et al. 2003; Norse & Crowder 2005). Therefore, marine conservation efforts today generally lag behind terrestrial efforts, with ~2.8% (IUCN & UNEP-WCMC 2013) of the world's oceans protected compared with ~10-15% (Soutullo 2010) of the globe's terrestrial spaces. While marine conservation hurries to catch up and learn from the well-established theoretical foundation of terrestrial conservation strategies (Meffe & Carroll

1994; Beck 2003), such methods often cannot be easily transferred. The inherent differences of the marine realm such as the connectivity of marine waters, dispersal by ocean currents, the absence of physical borders and coastal pressures means that marine conservation and reserve design needs tailored tools and approaches (Carr et al. 2003; Roberts 2005).

Many countries are embracing marine protected areas (MPAs) as a tool for ocean management. MPAs were first developed for fishery management (Cushing 1988; Dugan & Davis 1993; Rowley 1994). There are different types of MPAs, such as no-take reserves that restrict all human activities, and multiple use reserves where fishing and other activities are permitted. The type of reserve and its necessary restrictions is dependent on the MPA objectives, which can range from the protection of single threatened species to marine biodiversity as a whole (Allison et al. 1998). A plethora of studies have shown the effectiveness of MPAs as a global conservation tool (Castilla 1999; Roberts 1995; Gell & Roberts 2003; Hastings & Botsford 2003; Halpern 2003; Edgar et al. 2014; Guidetti et al. 2014). The benefits of MPAs are evident by increased habitat heterogeneity, species richness, biomass and size (Roberts 1995; Wantiez et al. 1997; Edgar & Barrett 1999; Halpern 2003; Lester et al. 2009). There are also benefits for fisheries that can help maintain sustainable fishing stocks, such as larval transport and spill over of juveniles and adults to areas outside the MPA (Gerber et al. 2003; Goñi et al. 2010). However, establishing an MPA also involves costs. The establishment of a no-take MPA displaces fishing effort (potential revenue), may limit recreational activities e.g., fishing, diving and boating and can have conflicting interests with other marine stakeholders e.g., aquaculture and hydrocarbon industries (Douvere 2008). Overcoming these conflicting issues is needed for enabling implementable marine conservation actions.

In recent years, marine conservation prioritisation and planning have begun to incorporate more socioeconomic considerations (Ban & Klein 2009; Ban et al. 2013). This is due to the acknowledgment that minimising societal costs also has an important bearing on the establishment of an MPA (Naidoo et al. 2006; Klein et al. 2009; Smith et al. 2009). Pursuing unfeasible options is timely and costly and conservation objectives may be able to be met in other places (Ban & Klein 2009). Several studies have explored the inclusion of commercial fisheries objectives and evaluated the trade-offs with meeting conservation goals (Stewart & Possingham 2005; Klein et al. 2008a; Klein et al. 2008b; Grantham et al. 2013). Both Stewart and Possingham (2005) and Klein et al. (2008a) found that adverse socioeconomic impacts for fisheries could be minimised (by one-third based on Stewart & Possingham 2005; and by ~21% based on Klein et al. 2008a) when including economic design constraints, and actually increased reserve size (~3%). Other studies have aimed to minimise socioeconomic impacts by including population pressure (Ban et al. 2009), the density of small boats (Sala et al. 2002) and local tenure (Weeks et al. 2010a). Most recent is the use of a multi-zone planning (Marxan with Zones), which enables multiple objectives to be met with different zones. Klein et al. (2009) first used this tool to meet equitable solutions for eight commercial fisheries in California. They found that a loss of less than 9% for each fishery was required to meet conservation targets (Klein et al. 2009). Such tools substantially contribute to the advancement of conservation prioritisation and implementation success. However, the focus has predominantly been on minimising fishery impacts and only very recently have other marine stakeholders, activities and social concerns been examined (e.g., Weeks et al. 2010a; Ban et al. 2013; Halpern et al. 2013; Levin et al. 2013).

While marine conservation prioritisation has advanced rapidly there are still many gaps and advancements to achieve. One of the biggest problems for conservation in the marine realm is the lack of spatial data in comparison to terrestrial landscapes (Levin et al. 2014). There is a lack of biodiversity data as well as spatial cost information. Surrogates are often heavily relied upon (Ferrier 2002; Lombard et al. 2003; Rodrigues & Brooks 2007). However, they are sometimes wrongly used, for example spatial costs in marine environments have different patterns than terrestrial costs - this is further explained in this thesis within Chapter 3. The connectivity of marine systems also plays a role in determining conservation priorities. Yet, species movement and migration is often not explicitly included in conservation prioritisation. There are also marine environments that are surrounded by land such as the Caribbean Sea, the Coral Triangle, the Black Sea, Baltic Sea and the Mediterranean Sea. These regions are well connected by neighbouring countries and conservation actions taken in one place can largely affect that of another (Carr et al. 2003; Klein et al. 2012; Makino et al. 2013a). These geographic areas often require collaborative strategies to adequately protect biodiversity of the region. Given these gaps and limitations this thesis aims to develop tools and approaches which contribute to better conservation prioritisation in marine environments.

1.4 The case of the Mediterranean Sea

Until recently there has been very limited application of systematic marine conservation prioritisation approaches in the Mediterranean Sea (Giakoumi et al. 2012b). Given the infancy of conservation prioritisation in this region and its complex geographic setting, the Mediterranean Sea is an ideal case study to further develop methods for marine conservation. All chapters in this thesis aim to advance conservation planning in Mediterranean Sea. Here, I describe the case of the Mediterranean Sea and its current progress in conservation.

The Mediterranean Sea is the largest and deepest semi-enclosed sea in the world (Boudouresque 2004; Coll et al. 2010). It features unique ecosystems and habitats such as: the endemic seagrass *Posidonia oceanica* that forms large underwater meadows, reefs built by sea snails *Dendropoma petraeum* and Mediterranean coralligenous assemblages (Bianchi & Morri 2000; Boudouresque 2004; Santangelo et al. 2007). Other important systems that provide habitat for many endemic species include deep-sea areas of submarine canyons and seamounts, pelagic waters and coastal zones (Arvanitidis et al. 2002; Manconi et al. 2013; Kvile et al. 2014). These distinctive habitats within the Mediterranean Sea support a rich marine biodiversity that is compacted into a small area surrounded by twenty-five countries (Abdulla et al. 2009).

The Mediterranean Sea constitutes less than 1% of global ocean surface space, but it contains immense biodiversity relative to its size (Bianchi & Morri 2000; Cuttelod et al. 2008). The sea provides habitat for twenty-three cetacean species of which nine are year-round residents, along with seventy-one cartilaginous fishes (*Chondrichthyes*) and 7% of the total global marine fish species (Abdul Malak et al. 2011). This area supports ~18% of the world's macroscopic marine species, of which 25-30% of these species are endemic (Bianchi & Morri 2000; Cuttelod et al. 2008; Abdulla et al. 2009). A prime example is the endemic Mediterranean monk seal (*Monachus monachus*), believed to be the world's most Endangered pinniped (Panou et al. 1993). Other emblematic species of conservation concern are sea turtles (Casale & Margaritoulis 2010) and the eastern-Atlantic bluefin tuna population, for which the Mediterranean Sea serves as a major breeding ground (Fromentin 2003).

Currently the biodiversity of the Mediterranean Sea is threatened by a wide range of sea and land-based anthropogenic activities (Coll et al. 2010; Micheli et al. 2013a). The coastal areas of the sea are highly populated, with ~600 cities and a population of ~250 million inhabitants, along with ~250 million tourists that visit annually (Cuttelod et al. 2008). This region is also divided into various geopolitical units with an array of socio-economic factors. Thus, this heavily populated area with many stakeholders that make coordination challenging means that the Mediterranean Sea is particularly susceptible to threats such as habitat degradation, pollution, invasive species, climate change and exploitation of marine species. In addition, social (e.g., cultural, religious, political) and economic divisions such as the large contrast between Europe and North Africa, greatly challenges our ability to minimise and control threats to the surrounding environment (Fraschetti et al. 2009).

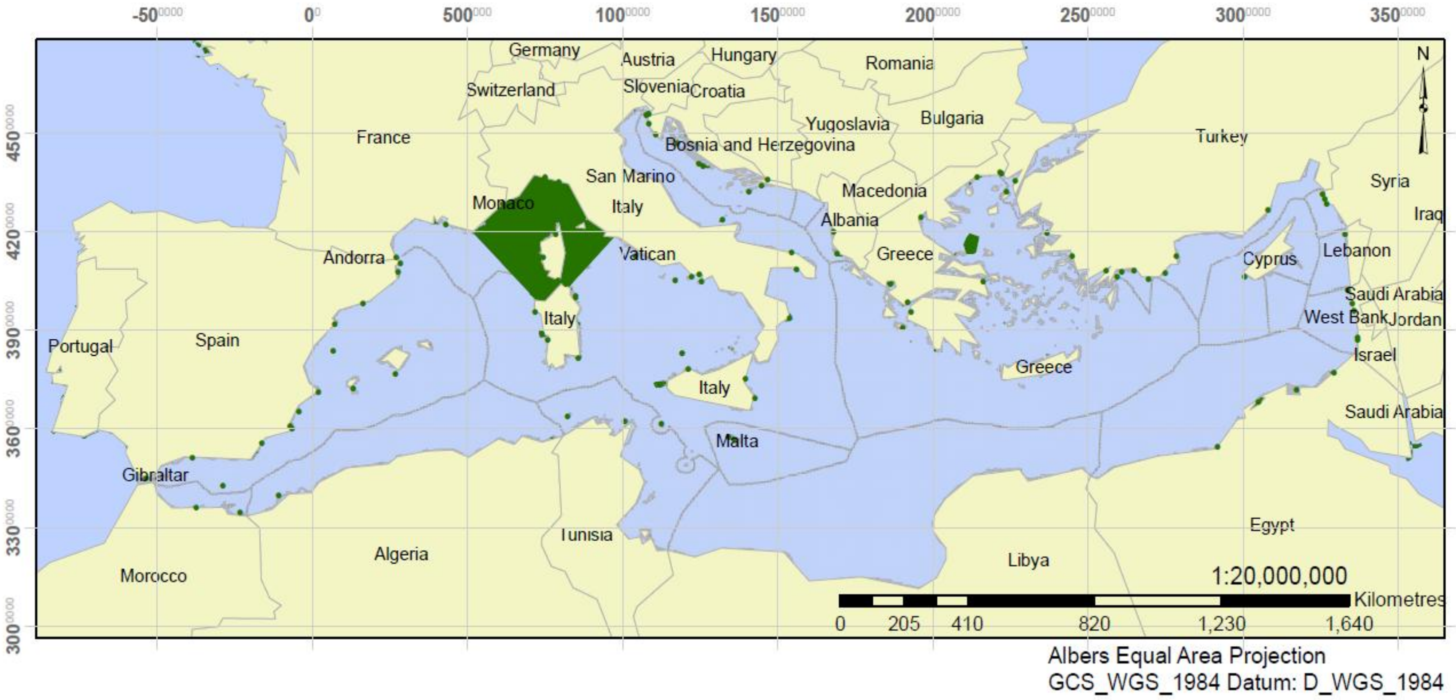

Figure 1.2. Marine protected areas of the Mediterranean Sea (in green) with Exclusive Economic Zone (EEZ[1]) for each country. The large area between France, Monaco and Italy is the Pelagos sanctuary, which is the only transboundary protected area in the Mediterranean Sea (Data provided by IUCN & UNEP-WCMC 2010 & Protected Planet).

The Mediterranean Sea has a small and unrepresentative reserve system. There are currently 117 Marine Protected Areas (MPAs) within the region (Portman et al. 2012), which covers approximately 0.01% of its area (excluding the Pelagos sanctuary; Fig. 1.2). These protected areas are small, often unenforced (Guidetti et al. 2008), uncoordinated units (Andrello et al. 2013), that follow no combined legislation or criteria for establishment; each country has their own guidelines (Fraschetti et al. 2005). They are all concentrated along the coast, mainly in the north side of the basin, and offer no protection for deeper open-water regions (Portman et al. 2012; Fig. 1.2). The only exception is the Pelagos sanctuary (IUCN rank IV) for marine mammals, which is the only protected area recognised internationally (Notarbartolo-di-Sciara et al. 2008). However, the Pelagos sanctuary, often regarded as a “paper park” (Fenberg et al. 2012), is based on ongoing monitoring rather than explicit management actions (Moulins et al. 2008; Notarbartolo-di-Sciara et al. 2008; Panigada et al. 2008). These factors indicate that the current protected areas within the Mediterranean poorly represent the diversity of the basins ecosystems (Abdulla et al. 2008) and are even considered ineffective in their protection of marine life (Dimitrakopoulos et al. 2004; Giakoumi et al. 2010). While the implementation of marine protected areas benefits conservation

[1] Most Mediterranean countries have not yet formally claimed or agreed on the spatial delimitation of their exact EEZ boundaries (Suárez de Vivero et al. 2009). For this thesis I will refer to tentative EEZ boundaries as provided by VLIZ 2012, although not formally defined under the Law of the Sea treaty.

awareness, the lack of structural integrity and cross-country collaboration challenges the ability of such areas to protect and sustain the biodiversity of the Mediterranean Sea.
Current marine protected areas of the Mediterranean Sea are promulgated by individual countries and thus lack the context of a network system. However, there are organisations, policies and committees in the Mediterranean Sea that aim to conserve and protect the Mediterranean Sea via collaboration (Micheli et al. 2013b). For example, MedPAN is a network that aims to improve the management of marine protected areas in the Mediterranean basin. This organisation operates through the objective of the Convention on Biological Diversity and the Barcelona Convention and aims to facilitate the implementation of national and regional protected area networks with common goals and methods. The management of sustainable fisheries in the European Union is collaborative through the Common Fisheries Policy (CFP) and the Priority Actions Programme/Regional Activity Centre (PAP/RAC) aims at integrating coastal area management of the Mediterranean Sea. However, benefits of coordinating multinational conservation efforts in the Mediterranean Sea have not been examined or quantified in earlier research.

Systematic conservation planning has recently been used to monitor and locate protected areas in some parts of the Mediterranean Sea at regional scales. Publications that have used this method have concentrated on small areas of the Mediterranean such as the central Aegean Sea near Greece (Giakoumi et al. 2010), coast of southern Italy (Fraschetti et al. 2009) and the Ligurian and Tyrrhenian coasts along western Italy (Maiorano et al. 2009). Although recommendations from these studies have not been put into action and adopted by decision makers in the Mediterranean, they present working examples that can be expanded. The practical application of systematic conservation planning is still in its infancy in the Mediterranean Sea (Maiorano et al. 2009).

To protect the biodiversity of this global biodiversity hotspot (Myers et al. 2000; Brooks et al. 2006; Cuttelod et al. 2008; Coll et al. 2010), conservation plans that meet specific conservation goals and fit within realistic socio-economic constraints, must be developed. When MPAs and reserve networks are planned systematically, implemented and managed correctly, they can effectively preserve marine species and ecosystems (Allison et al. 2003; Lubchenco et al. 2003; Gaines et al. 2010a; 2010b; Edgar et al. 2014). Thus, this thesis will use systematic conservation planning and spatial prioritisation strategies to improve the effectiveness and efficiency of marine life protection in the Mediterranean Sea.

1.5 Structural overview of the thesis

This thesis aims to contribute novel methods to conservation prioritisation and planning that are integral components of achieving successful protection of the marine realm. While the uptake of such plans may not be physically undertaken, these chapters and their case studies highlight conceptual approaches that can directly benefit decision makers in producing cost effective, adequate and realistic conservation plans that can help sustain marine biodiversity. I use the Mediterranean Sea, a global biodiversity hotspot (Myers et al. 2000; Brooks et al. 2006; Cuttelod et al. 2008), as a model system to address three key themes (Fig. 1.3).

Following the Introduction, the first key theme addressed in **Chapter 2** and **Chapter 3**, aims to improve the efficiency of marine conservation prioritisation. Two questions are posed in this part: (i) Can collaboration save costs in marine conservation? (**Chapter 2**) and, (ii) How does cost affect large-scale multinational marine conservation planning? (**Chapter 3**). The second theme of the thesis aims to improve the adequacy of species protection. **Chapter 4** includes connectivity to better protect the life stages of moving and migrating marine species, and **Chapter 5** examines the use of satellite technology to better predict species habitat and determine major threats. The third theme (**Chapter 6**) aims to improve implementation success by understanding the flexibility and limitations of meeting conservation targets, while also minimising economic losses. This is an opportunistic case study that explores how conservation priorities can be altered by the inclusion of other marine stakeholders and activities such as the hydrocarbon industry and commercial fishers. In the last chapter of the thesis (**Chapter 7**), I outline the contributions of this research to the understanding of marine conservation prioritisation and planning, and its novelty in the Mediterranean region. In addition, I synthesise cross-cutting themes highlighted across Chapters 2 to 6. Finally, I propose future research directions that emerge from the outcomes of this thesis.

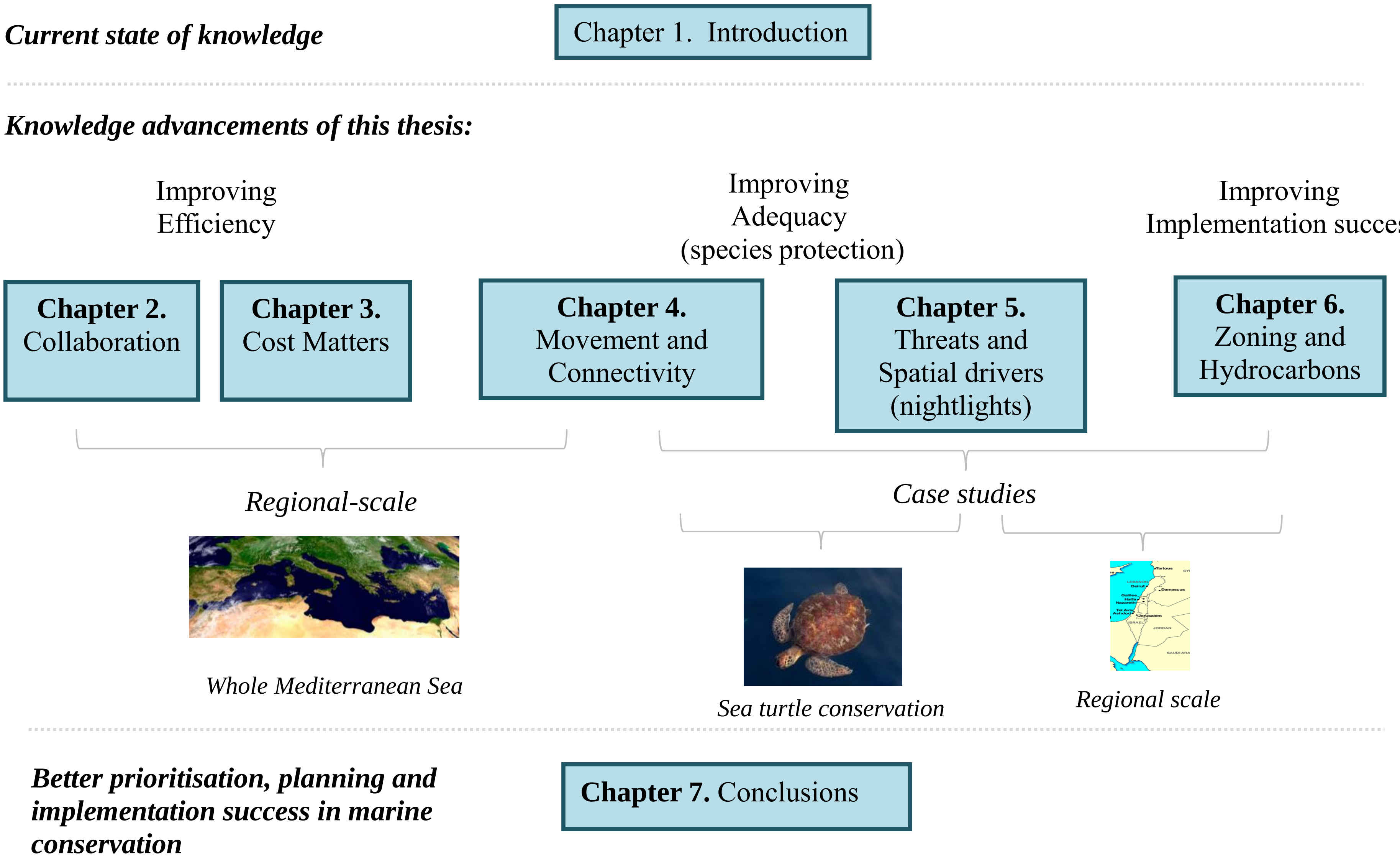

Figure 1.3. Structural overview and relationships between chapters of the thesis

Chapter 2

Collaboration among countries in marine conservation can achieve substantial efficiencies

Photo credit: Getty images

Diversity and Distributions (2013) **19**, 1380 – 1393

2.1 Abstract

Aim: Multinational collaboration is important for successfully protecting marine environments. However, few studies have assessed the costs and benefits incurred by taking collaborative action. One of the most complex marine regions in the world is the Mediterranean Sea biodiversity hotspot. The sea is shared by over 20 countries across three continents with a vast array of socio-economic and political backgrounds. We aimed to examine how collaboration between countries of the Mediterranean Sea affects conservation plans when costs and threats are considered.

Location: The Mediterranean Sea.

Methods: We compared three collaboration scenarios to test the efficiencies of coordinated marine conservation efforts: full coordination between Mediterranean countries, partial coordination within continents and no coordination where countries act in isolation. To do so, we developed four basin-wide surrogates for commercial and recreational fishing effort in the Mediterranean Sea. Using a systematic decision support tool (Marxan), we minimized the opportunity costs while meeting a suite of biodiversity targets.

Results: We discovered that to reach the same conservation targets, a plan where all the countries of the Mediterranean Sea collaborate can save over two- thirds of the cost of a plan where each country acts independently. The benefits of multinational collaboration are surprisingly unequal between countries.

Main conclusions: This approach, which incorporates biodiversity, costs and collaboration into a systematic conservation plan, can help deliver efficient conservation outcomes when planning spatially explicit actions within marine environments shared by many countries.

2.2 Introduction

While most marine conservation actions are currently conducted within single countries, multinational initiatives involving cross-country collaborations are increasing within the marine realm (Mackelworth 2012; Punt et al. 2012). These collaborative programmes between countries are perceived to incur large costs and resources (Stolton et al. 1999; Sandwith et al. 2001). Several studies have assessed the ability of collaborative initiatives to protect terrestrial biodiversity and reduce the costs incurred by taking collaborative conservation action (Strange et al. 2006; Bladt et al. 2009; Kark et al. 2009; Moilanen et al. 2013). However, no studies have investigated this issue within the marine environment.

Despite conservation efforts notoriously lagging behind in the marine realm (Chape et al. 2005), the intrinsic ecological connectivity of marine systems suggests that cross-country collaboration makes sense (Mackelworth 2012). The marine system is temporally dynamic, highly connected and unrestricted by national borders (Hyrenbach et al. 2000; Agardy et al. 2011). Marine borders between countries exist, but the absence of physical boundaries makes them less easily defined compared with terrestrial borders (Carr et al. 2003; Mackelworth 2012). The connectivity of waters in the marine realm means that countries invariably share many marine species, as well as conservation threats and challenges (Wilkinson et al. 2004). Moreover, actions conducted in one marine space often affect that of another, for example, pollution dispersion and invasive species (Boudouresque & Verlaque 2000). Such interdependence is especially evident in places where many countries or states share a common sea or ocean, such as the Caribbean, the Coral Triangle, the Baltic Sea and the Mediterranean Sea.

Establishing coordination between countries is challenging within heterogeneous systems (Stolton et al. 1999), and one of the most politically and ecologically complex regions in the world is the Mediterranean Sea (Cognetti 1993). The Mediterranean Sea supports a rich marine biodiversity (Abdulla et al. 2009) that is concentrated in a small area surrounded by over twenty countries across three continents: Europe, Asia and Africa. Thus, many conservation issues in the Mediterranean Sea involve two or more countries. The Mediterranean Sea is visited by *c.* 200 million tourists a year and supports the livelihood of *c.* 150 million people via small-scale subsistence fishing, employment within commercial fisheries and as a food source (Madau et al. 2009; UNEP 2013). In addition, the multiple users of this common resource face very different circumstances. Countries surrounding the Mediterranean Sea show a vast array of cultural values, economic statuses, political systems, religions and languages (Badalamenti et al. 2000; Kark et al. 2009). All these additional factors can impede successful collaboration (Sandwith et al. 2001).

Most conservation efforts in the Mediterranean Sea are uncoordinated (Giakoumi et al. 2012b) and are insufficient at protecting the sea's highly threatened biodiversity (Micheli et al. 2013a). The goals of the Convention on Biological Diversity (CBD 2013), which is an agreement signed by most Mediterranean countries aiming to conserve 10% of the sea, are far from being achieved (Gabri'e et al. 2012; Giakoumi et al. 2012b). With limited conservation measures in place, the sea's native species and ecosystems continue to face threats from both land and sea-based anthropogenic activities (Coll et al. 2010; 2012). Existing marine protected areas (MPAs) are relatively small and are not based on coordinated legislation or criteria for establishment; each country has its own guidelines for administering MPAs (Fraschetti et al. 2005). While the implementation of protected areas has raised conservation awareness, limited structural integrity and cross-country collaboration challenge the ability of these MPAs to protect and sustain the biodiversity of the Mediterranean Sea (Abdulla et al. 2009; Micheli et al. 2013b).

Planning collaborative conservation in complex environments requires advanced spatial prioritization tools (Kark et al. 2009). In the Mediterranean Sea, where the survival of biodiversity relies on the ability for countries to collaborate and collaboration is obstructed by the socio-economic and political complexity of the region, conservation plans and actions should include costs (Ando et al. 1998; Stewart et al. 2003) and other anthropogenic factors (Kark et al. 2009). Systematic conservation planning for the Mediterranean Sea has only recently been considered within local studies (Fraschetti et al. 2009; Maiorano et al. 2009; Giakoumi et al. 2011). Until today, no plans have explicitly included the cost of conservation actions or considered socio-economic factors when choosing priority conservation areas at the whole Mediterranean scale. Systematic methods driven by explicit objectives that incorporate costs of conservation actions (Moilanen et al. 2009) can help better direct and inform decision-makers.

To our knowledge, no conservation plans have quantified the effectiveness of between-country collaboration within the Mediterranean Sea. Nevertheless, spatial priority areas identified in the Mediterranean Sea via species richness estimates, expert advice and/or threat mapping (Notarbartolo-di-Sciara & Agardy 2009; Coll et al. 2012) often cover several countries, and their establishment requires coordinated conservation action. Similarly, global conservation priority areas and hotspots often involve several countries (Myers et al. 2000; Brooks et al. 2006). While the need for collaborative action is evident, the benefits and cost efficiency of collaboration are often not as clear (Sandwith et al. 2001; Mackelworth 2012). Successful cross-country collaborations in conservation are often attributed to a transparent planning process with defined costs and savings (Sandwith et al. 2001; Agardy et al. 2011). Therefore, quantifying the benefits of potential collaborative initiatives in the marine realm may provide incentives for countries or stakeholders to collaborate (Agardy et al. 2011).

Here, we present the first study to quantify the effectiveness of collaborative conservation between countries in the marine realm. Prior knowledge of the costs and benefits of turning marine areas into collaborative MPAs can better direct us to forge collaborative ties which will reap benefits, despite the challenges they pose. We explore the role of cross-country collaboration in the Mediterranean Sea. We assess three collaboration scenarios by examining spatial priorities for the protection of threatened Mediterranean vertebrate species using a systematic conservation planning tool that incorporates the cost of conservation actions. We aim to address the following question: Can collaboration between countries of the Mediterranean Sea improve conservation efficiency, achieving the same conservation outcomes for less cost?

2.3 Methods

2.3.1 Quantitative systematic planning

To evaluate the conservation efficiency of protecting threatened species within the Mediterranean Sea, we used Marxan. Marxan is a decision support tool for systemic conservation planning that implements a minimum set approach (Ball et al. 2009; Watts et al. 2009). It uses a simulated annealing algorithm to solve a well-defined mathematical problem, identifying sites that fulfil quantitative targets for biodiversity features in a compact system of protected areas for the least possible cost (McDonnell et al. 2002; Ball et al. 2009). Thus, Marxan works to reach a set target for the least cost, which in our case is the opportunity cost to commercial fisheries and non-commercial (subsistence and recreational) fishers.

Distribution range data of all known 77 threatened or near threatened vertebrate species (six of which are endemic to the Mediterranean Sea) were compiled from the recent IUCN database (IUCN 2012; Fig. 2.1; see Table S2.1). Five taxa groupings comprising seven marine mammal species (Reeves & Notarbartolo-di-Sciara 2006), five seabird species (Birdlife International 2012), 24 native fish, 39 shark and ray (cartilaginous fish) species (Abdul Malak et al. 2011) and nesting sites of two sea turtle species were used (Casale & Margaritoulis 2010; Table S2.2). These data provide a baseline at a whole Mediterranean Sea scale, at a reasonable resolution for conservation plans at large scales. All data were overlayed and projected into Albers Equal Area Projection with a resolution of 10 x 10 km planning units (creating 26,946 planning units), using ARCGIS software (ESRI 2008). We set a conservation target to protect 10% of each species' distribution, following targets of the Convention on Biological Diversity (CBD 2013). This 10% target was set for each scenario based on the distribution ranges of the species present within the defined spatial extents (in collaboration scenarios, targets were met jointly between collaborating countries). While this approach does not consider whether the target is adequate at conserving the species or maintaining population viability, it does deliver a baseline of equitable representation (Tear et al. 2005; Carvalho et al. 2011).

Input parameters were held constant in Marxan to ensure comparisons were valid. To enable comparisons between scenarios, we did not preferentially cluster planning units in space, but set algorithm parameters so that all targets (10% of each species' distribution) were met. Ten Marxan runs were performed with 1000 repetitions each, producing ten 'best solution' outputs for each collaborating area of a scenario. The 'best solution' output is the reserve system that performs best at reaching its conservation target with minimal cost. High-priority conservation areas were identified by the percentage of times (e.g., ≥ 90%) each planning unit was selected in the ten 'best solutions'.

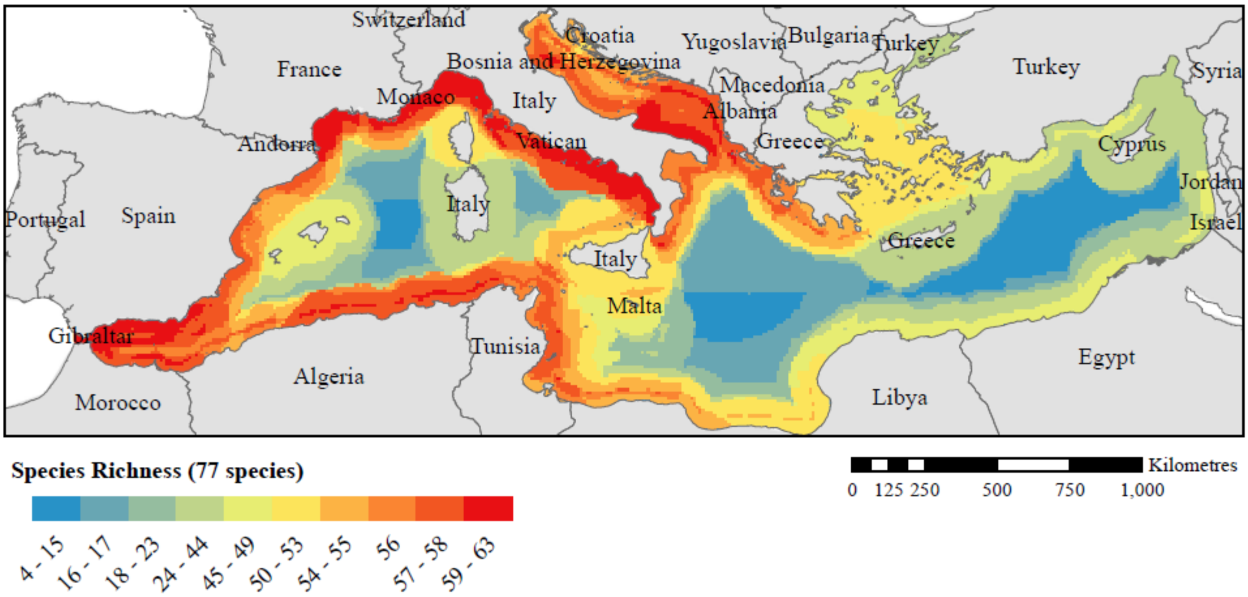

Figure 2.1. Species richness of 77 threatened vertebrates in the Mediterranean Sea (IUCN 2012; see details in Table S2.1).

2.3.2 Cross-country collaboration

To test the role of cross-country collaboration, we used the Mediterranean Sea's tentative division into Exclusive Economic Zones (EEZs; VLIZ 2012) and applied Marxan to find good reserve systems following varying collaboration scenarios. To enable an economic comparison between different collaboration scenarios, we used a fishing opportunity cost layer as a reservation cost for each planning unit (Fig. 2.2). We compared three collaboration scenarios following Kark et al. (2009): (1) fully coordinated: all countries collaborating, (2) partly coordinated: countries within each continent collaborating, including Africa (8116 planning units: Morocco, Algeria, Tunisia, Libya, Egypt), Asia (1672 planning units: Turkey, Syria, Lebanon, Israel, Palestinian territories) and Europe (17,158 planning units: EU countries, Monaco, Croatia, Bosnia–Herzegovina, Montenegro, Albania), and (3) uncoordinated: each country acts in isolation. To clarify, for the fully coordinated scenario, there were a total of 10 Marxan runs (producing 10 best solutions); for the partly coordinated scenario, there were 30 Marxan runs (10 best solutions for each of the three continents); and for the uncoordinated scenario, there were 180 Marxan runs (10 best solutions for each of the 18 countries). The selection frequency of the ten 'best solutions' was compared for each country with respect to the three levels of collaboration.

For the uncoordinated scenario, we only considered countries with an EEZ covering an area of five or more planning units (eliminating: Bosnia–Herzegovina, Gibraltar, Monaco and Slovenia, where the EEZ area was too small for selecting spatial priorities). We compared the cost, area and spatial arrangement of high-priority conservation areas with respect to the three levels of collaboration.

2.3.3 Incorporating opportunity cost

There are a range of costs involved with implementing and planning for an MPA network (Ban & Klein 2009). An important cost to consider is opportunity cost (Klein et al. 2010), which is the forgone cost (lost benefit) when an activity takes place where another occurred or can occur (e.g., fishing net benefit or profit that will be forgone when an area is declared an MPA; Cameron et al. 2008). Here we constructed four cost layers that represent the opportunity cost to fishers of the Mediterranean Sea (Fig. 2.2). We constructed these layers by summing two spatial layers (both layers in US$) that represent revenue (an approximation for opportunity cost) for commercial fishing and non-commercial fishing as described below. The four resulting cost layers were used to test the sensitivity of our results and were used in Marxan for all three collaboration scenarios.

2.3.3.1 Commercial fishing cost

We developed an equation to represent the opportunity cost of commercial fishing at a spatial scale of 100 km^2. As a surrogate for commercial fishing revenue at the whole Mediterranean Sea scale, we used data on tonnes of fish caught in 28 geographical sub-areas (GSAs) for the year 2008 provided by the General Fisheries Commission for the Mediterranean (FAO 2011). To date, this is the most current and spatially refined data available for the Mediterranean Sea on fish catch.

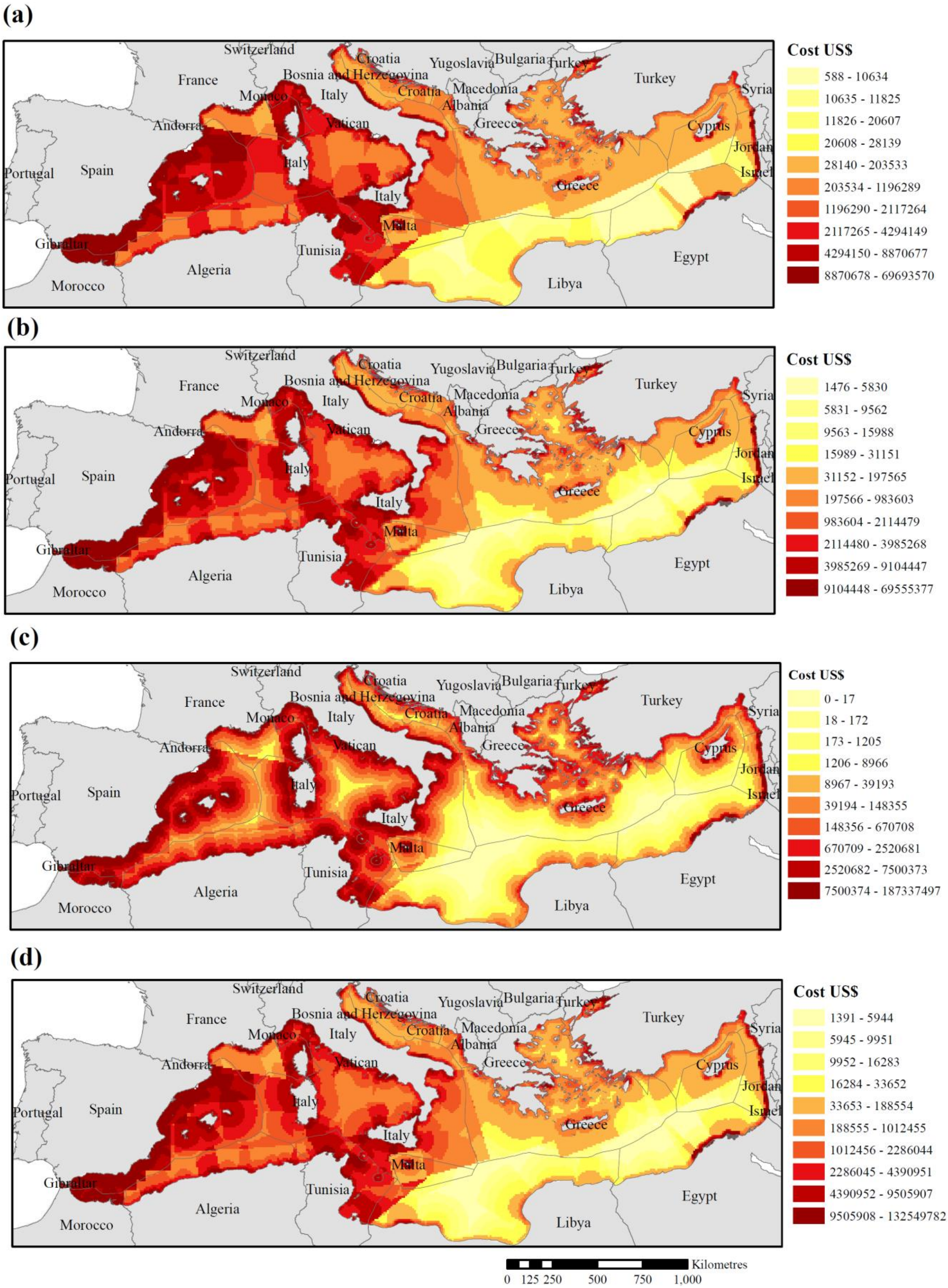

Figure 2.2. These four cost layers represent annual revenue (opportunity cost in US$ displayed by a quantile range) for commercial and non-commercial fishing in the Mediterranean Sea. Commercial fishing is based on Food and Agriculture Organization (FAO) data (FAO 2011) and the distance to ports (National Geospatial-Intelligence Agency 2005), which is exponentially weighted by a constant α. Non-commercial fishing opportunity cost is based on local population size (CIESIN

2005) and the relative Gross domestic product (GDP) of the country or an equal value of $50 per day for all countries; a) commercial fishing (decay rate from port $\alpha = 0.001$) and non-commercial fishing use GDP values, b) commercial fishing ($\alpha = 0.01$) and non-commercial fishing use GDP values, c) commercial fishing ($\alpha = 0.05$) and non-commercial fishing use GDP values, d) commercial fishing ($\alpha = 0.01$) and non-commercial fishing use an equal value ($50) for all countries.

To estimate the annual catch of commercial fishing C_i in each planning unit i (100 km^2), we assumed that it is proportional to the size of the nearest port P_{size}[2] and the distance d to that port (km) weighted exponentially by a constant α (we used three values to test the sensitivity of our results: 0.001, 0.01, 0.05; Fig. S2.1) and then multiplied by the area A of planning unit i. To ensure that the total value of catch in the region sums to its real value (value of each GSA region stated in FAO (2011)), we normalized the catch of commercial fishing in each planning unit by a measure of total regional effort R_{effort}, which is equal to

$$R_{effort} = \sum_{i=1}^{m} P_{size} e^{-\alpha d} A_i \ ,$$

where m is the number of planning units in a given region (28 GSA regions). We multiplied the final value by the total biomass of fish in the region (ton) $R_{biomass}$, multiplied by the price of fish (US$ per ton) C_{fish}, to obtain a monetary cost. As a surrogate for C_{fish}, we used the price of European anchovy (*Engraulis encrasicolus*), which is US$3990 per ton (FAO 2010). We chose the European anchovy parameter based on a ranking of fish species that contribute to most of the landings within the Mediterranean Sea (Lleonart & Maynou 2003; FAO 2010) and because comparably it is an average-priced fish species on the market (FAO 2010). The final expression for an estimate of the opportunity cost of commercial fishing C_i in each planning unit i is as follows:

$$C_i = \left(\frac{P_{size} e^{-\alpha d} A_i}{R_{effort}}\right) C_{fish} R_{biomass} \ ,$$

where R_{effort} is defined above. In an effort to smooth hard boundaries between the GSAs within our opportunity cost layer, we used the spatial low-pass filtering tool in ArcGIS (ESRI 2008). The three resulting opportunity cost layers provide baseline estimates of the spatial cost involved with closing commercial fisheries in any place that is part of the Mediterranean Sea (Fig. S2.1).

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[2] There are four port sizes P_{size}:*1 = very small; 2 = small; 3 = medium; 4 = large*. "*The classification of port size is based on several applicable factors, including area, facilities and wharf space*" - National Geospatial-Intelligence Agency (2005).

2.3.3.2 Non-commercial fishing cost

Very limited quantitative information exists for the revenue and effort of non-commercial fishing (subsistence and recreational fishers) in the Mediterranean Sea (Lloret et al. 2008). Therefore, we assumed that the opportunity cost of non-commercial fishing within the Mediterranean Sea is a function of human population size along the coastline. Human population data along the coast have been used as a surrogate for fishing (Ban et al. 2009) and linked with declining fish species (Stallings 2009). We developed an equation to represent the annual opportunity cost of non-commercial fishing, NC_i, in each planning unit i (100 km^2), where the cost is a multiplication of the number of days fishing per fisher per year N_{visits}, the cost (US$) of 1 day fishing per year Cf_{ishing} and the annual number of fishers frequenting the planning unit Nf_i:

$$NC_i = N_{visits} C_{fishing} Nf_i \,.$$

For the parameter N_{visits}, previous studies suggest that on average, recreational fishers in mainland Spain and France engage in *c*. 30–35 days of fishing per year (SFITUM 2004); however, higher frequencies have been found within other parts of the Mediterranean (Morales-Nin et al., 2005; Ünal et al., 2010). The frequency of subsistence fishing in the Mediterranean Sea is unknown. As such data were unavailable we used a conservative estimate of 30 visits per year per person (N_{visits}).

To calculate $C_{fishing}$, we followed two approaches resulting in two separate cost layers: (1) cost equal to 1-day salary per country using 2011 Gross domestic product (GDP) (International Monetary Fund 2012; Fig. S2.2a), and (2) a constant cost of US$50 per-day salary per person regardless of the country (Fig. S2.2b).

We calculated non-commercial fishing effort per planning unit F_i, from human population data at a resolution of 2.5″ for 2010 (CIESIN 2005). To measure fishing effort, we buffered each planning unit by a 22-km radius (*c*. 12 nautical miles – average width of territorial waters for each country within the Mediterranean (Cacaud 2005)), but no more than 10 km inland from the coast (approximate distance a fisher would travel to the coast; Clark et al. 2002; Sidman & Fik 2007; Ellender et al. 2009). This buffer was chosen because the majority of non-commercial fishing of the Mediterranean Sea occurs within this distance (IEEP 2002; Morales-Nin et al. 2005). Within this buffer, we calculated the population by summing *n* units (all 1-km^2 population units in the buffer; CIESIN 2005), thus obtaining *k*.

For each country, a total non-commercial fishing effort, C_{NF}, was calculated. Studies indicate that *c.* 10% of each country's population engages in recreational fishing in Mediterranean Sea (GFCM 2011); however, for developed countries such as Spain, it is around 30% (SFITUM 2004; Ditton 2008). Currently, no estimate of subsistence fishing exists within the Mediterranean, although studies indicate that in developing countries a large portion of the population is reliant on fishing as a source of food, income and livelihood (Feidi 1998; Jacquet et al. 2010). As these values were unavailable for each country within the Mediterranean Sea, we used a surrogate for C_{NF} and calculated 30% of each countries coastal population density at a buffered distance of 10 km inland.

The annual number of non-commercial fishers frequenting a planning unit, Nf_i, is a function of the non-commercial fishing effort F per planning unit i and the area A of planning unit i, divided by, C_{effort}, the sum of non-commercial fishing effort for m planning units, where m is the number of planning units of a country's EEZ. We further scaled the cost of non-commercial fishing in each planning unit by the country's total non-commercial fishing effort, C_{NF}. For F_i, we assume that it is determined by the population k along the coastline, weighted exponentially by distance d (km) from the midpoint of k to the midpoint of the planning unit, with a constant α = 0.01. The annual number of fishers per planning unit is equal to:

$$Nf_i = \left(\frac{F_i\,A_i}{C_{effort}}\right) C_{Nf},$$

where,

$$C_{effort} = \sum_{i=1}^{m} F_i A_i.$$

and,

$$F_i = \sum_{k=1}^{n} k\, e^{-\alpha d}.$$

Using ArcGIS software (ESRI 2008), we constructed the opportunity cost layers along the Mediterranean coastline, giving a total of 8964 planning units. This provides a basic framework for calculating the cost of non-commercial fishing across the entire Mediterranean Sea using surrogate data; however, if such data become available in the future, it could feed into this equation to help create a more informative cost layer.

2.4 Results

We discovered that planning for marine conservation in the Mediterranean Sea when countries collaborate can significantly improve conservation efficiency compared with a scenario where countries act separately. Based on all four cost proxies and conservation targets, a fully coordinated scenario where the Mediterranean Sea is treated as a single integrated entity can save 70–77% of the total cost of an uncoordinated scenario where countries act in isolation (Table 2.1). A partly coordinated scenario, where countries from each continent coordinate, can save 55-71% of the total cost of an uncoordinated plan. Comparing a fully coordinated scenario with a partly coordinated scenario, we find that a fully coordinated scenario is still most cost-effective with savings 21–46% of the cost of the partly coordinated scenario. Thus, we found that to meet the same conservation targets, opportunity costs were substantially reduced when considering higher levels of coordination among countries (Table 2.1). The area required to implement the three conservation plans slightly decreased when collaboration between countries increased. The average area (of the 10 best solutions of each collaboration level) required to implement a fully coordinated conservation plan was reduced by 900-1200 km^2 when compared with an uncoordinated plan. The area required for implementing a partly coordinated scenario was also reduced by 600-1000 km^2 compared with the uncoordinated scenario (Table 2.1).

We found that when partaking in a collaborative plan, the savings in marine conservation costs differ among various Mediterranean countries. Due to the similarity in our collaboration findings for the four cost layers, we report here on our findings from one cost layer (see Fig. 2.2b for cost layer). We found that while 12 of 18 countries had their highest savings when conducting conservation in a coordinated plan (Table S2.3), six countries had greatest savings when conducting conservation with no coordination. Countries that had the greatest reductions in cost with a fully coordinated plan compared with an uncoordinated plan include Spain (saving *c.* US$1053 million), Tunisia (saving *c.* US$197 million), Italy (saving *c.* US$185 million) and Morocco (saving *c.* US$74 million; Figs 2.3 and 2.4). Countries that had the largest cost addition in a fully coordinated plan compared with an uncoordinated plan were France (cost of *c.* US$85 million), Libya (cost of *c.* US$23 million) and Malta (cost of *c.* US$14 million; Figs 2.3 and 2.4). Two countries saved the most from a partly coordinated scenario, Egypt (saving *c.* US$4 million) and Montenegro (saving *c.* US$0.5 million) (Table S2.3; Fig. 2.4).

Our results showed higher clustering of spatial priorities with high levels of collaboration (Fig. 2.5). For the fully coordinated plan, high-priority conservation areas for all threatened vertebrate species that minimize opportunity cost and meet conservation targets were identified. These areas included the coastal waters of France and Malta, the Adriatic Sea, the Aegean Sea, deep waters of Israel, coastal waters of Egypt and Libyan coastal waters near the Tunisian border (Fig. 2.5a). The spatial priorities found in the fully coordinated scenario were present in all four cost layers (Fig. S2.3). For the partly coordinated scenario, we found that the priority areas changed and become less clustered compared with the fully coordinated scenario. The Aegean Sea and the deep waters between the border of Greece and Libya became higher priority areas in the partly coordinated scenario (Fig. 2.5b). However, the Libyan and Egyptian coastal waters did not persist as high-priority areas as found in the fully coordinated scenario. Spatial priorities became even more dispersed in the uncoordinated scenario compared with the fully and partly coordinated scenarios (Fig. 2.5c).

Table 2.1. Cost and area associated with each of the three collaboration scenarios (see Table S2.4 for the per country costs of the uncoordinated scenario).The cost is the average cost over ten best solutions (1 best solution is from 1000 Marxan runs) of the MPA network. The 4 different cost layers represent opportunity cost for both commercial and non-commercial fishing for the entire Mediterranean Sea.

Cost Layers	**Average cost** (US$ million per year) **Area required to achieve conservation targets** (average number of planning units)		
	Fully Coordinated Scenario	**Partly Coordinated Scenario**	**Uncoordinated Scenario**
Cost Layer (a) Commercial fishing (decay rate α = 0.001) and non-commercial fishing uses GDP values	652 (2541)	910 Africa: 292 Asia: 20 Europe: 598 (2546)	2162 (2553)
Cost Layer (b) Commercial fishing (decay rate α = 0.01) and non-commercial fishing uses GDP values	614 (2536)	1140 Africa: 297 Asia: 256 Europe: 587 (2540)	2540 (2548)
Cost Layer (c) Commercial fishing (decay rate α = 0.05) and non-commercial fishing uses GDP values	275 (2543)	350 Africa: 133 Asia: 16 Europe: 201 (2,546)	1219 (2552)
Cost Layer (d) Commercial fishing (decay rate α = 0.01) and non-commercial uses an equal value ($50) for all countries.	600 (2537)	899 Africa: 315 Asia: 22 Europe: 562 (2537)	2104 (2547)

(MPA, marine protected area; GDP, Gross domestic product)

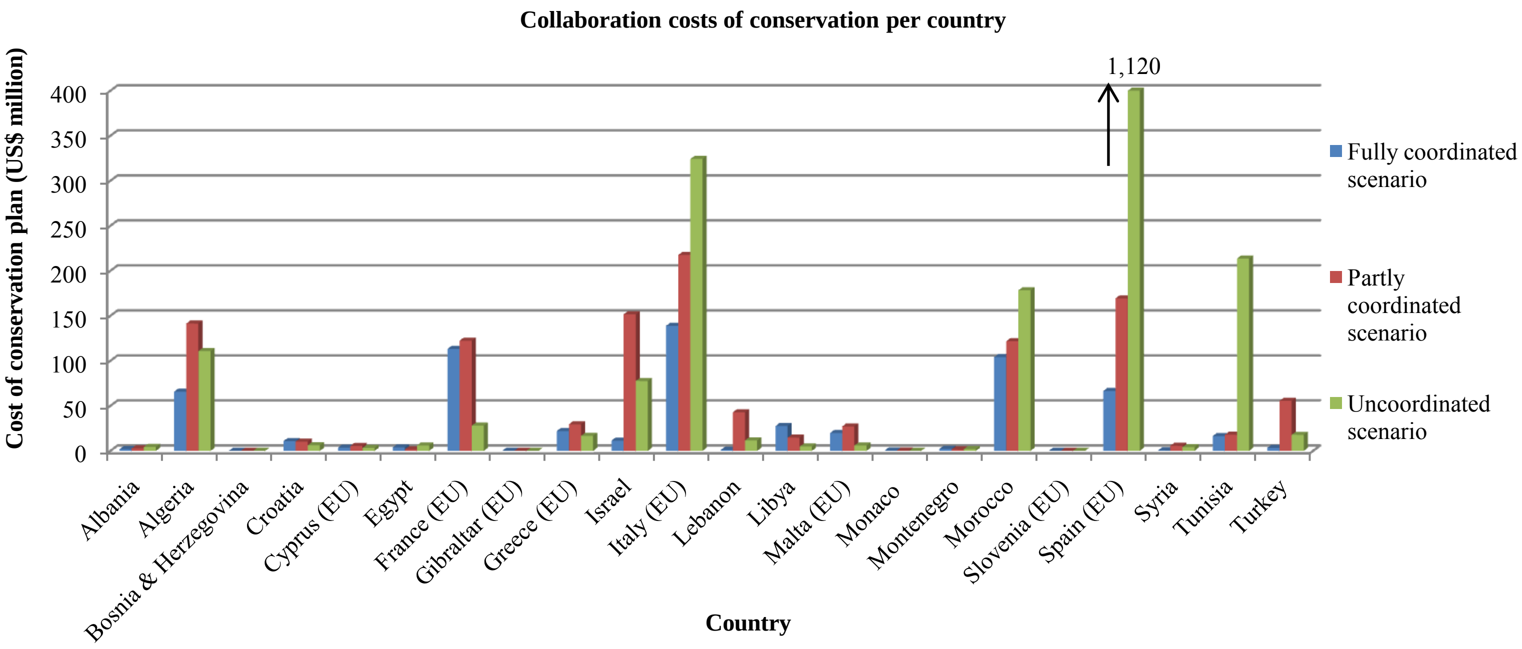


Figure 2.3. The cost of three conservation planning scenarios (fully coordinated, partly coordinated and uncoordinated) for each country's Exclusive Economic Zones (EEZ) in the Mediterranean Sea. Bosnia and Herzegovina, Gibraltar, Monaco and Slovenia were not included in the analysis due to their small EEZ area. The cost of the conservation plan (annual cost in US$ million) is the average cost of ten 'best solution' outputs (one best solution from 1000 Marxan runs) that were run for each geographical area of a scenario. The cost layer used in this analysis represents the opportunity cost of commercial fishing based on Food and Agriculture Organization (FAO) data (FAO 2011) and the distance to ports (National Geospatial-Intelligence Agency 2005), which is exponentially weighted by a constant $\alpha = 0.01$ and the opportunity cost of non-commercial fishing based on local population size (CIESIN 2005) and the relative Gross domestic product (GDP) of the country.

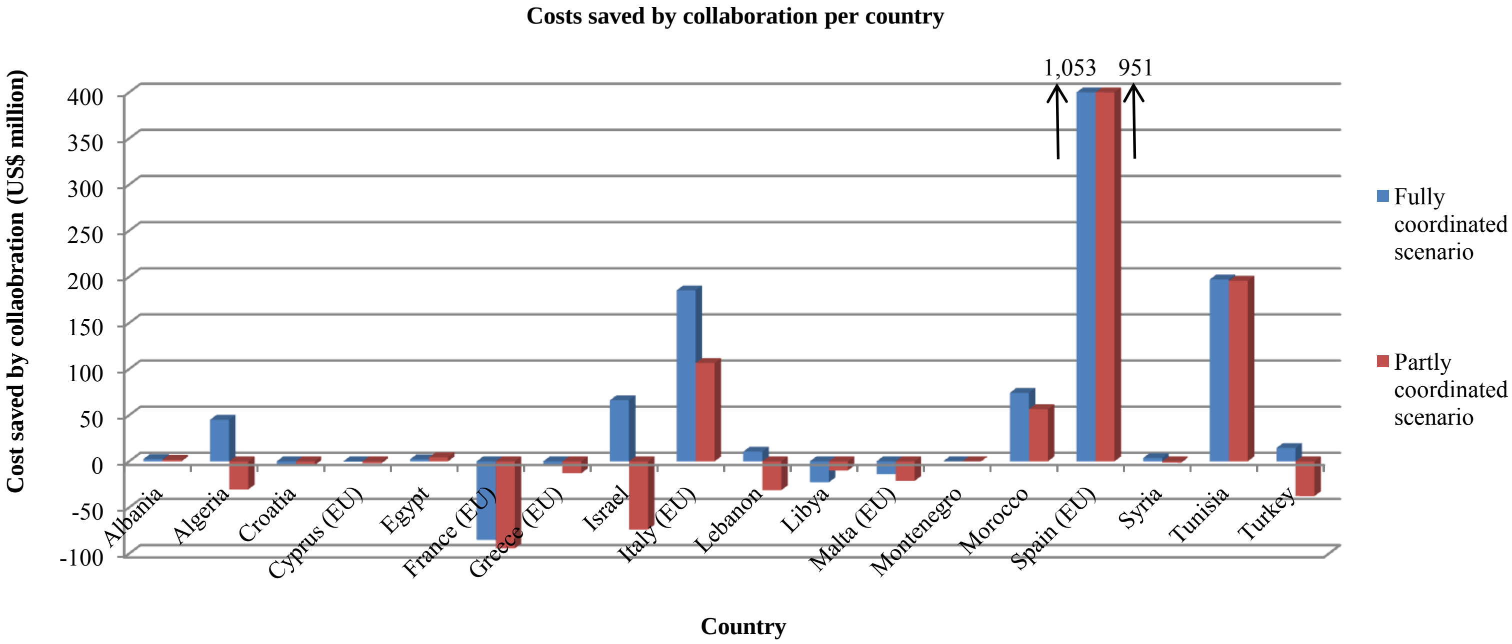


Figure 2.4. The cost saved by each country when taking part in a collaborative plan (fully or partly coordinated) compared with a plan where each country acts in isolation. Collaboration costs were subtracted from the costs of a non-collaborative plan. The cost of the conservation plan (annual cost in US$ million) is the average cost of ten 'best solution' outputs (one best solution from 1000 Marxan runs) that were run for each geographical area of a scenario. The cost layer used in this analysis represents the opportunity cost of commercial fishing based on Food and Agriculture Organization (FAO) data (FAO 2011) and the distance to ports (National Geospatial-Intelligence Agency 2005), which is exponentially weighted by a constant $\alpha = 0.01$ and the opportunity cost of non-commercial fishing based on local population size (CIESIN 2005) and the relative Gross domestic product (GDP) of the country.

2.5 Discussion

Conservation efficiency can be significantly increased when countries of the Mediterranean Sea coordinate their conservation actions to protect marine species. A fully coordinated plan across the Sea can reduce conservation costs by more than two-thirds (70%-77%; Table 2.1). Thus, an uncoordinated plan is almost four times more expensive than a coordinated plan to meet the same conservation targets. A partly coordinated plan where countries from each continent collaborate can also reduce conservation costs by more than a half (55–71%; Table 2.1) of the cost of an uncoordinated plan. In the light of our findings, collaboration between countries of the Mediterranean Sea should be encouraged as a means to improve conservation efficiency in the marine environment.

Costs were considerably reduced by increasing collaboration among countries, but area requirements were largely unaffected (Table 2.1). To reach the same conservation targets, all collaboration scenarios in our study required almost the same amount of area to be devoted to marine conservation. For the same spatial extent, we found that spatial priorities for marine conservation were clustered differently for the three levels of collaboration (Fig. 2.5). Thus, the huge cost efficiencies are realized via choosing to take conservation action in the cheapest places (Carwardine et al. 2008). If we assumed that the area and the cost are equal when planning for conservation, then we would be assuming a homogenous system (Ando et al. 1998; Naidoo et al. 2006), which is clearly not the case in the Mediterranean Sea or most other real systems.

Findings from terrestrial studies, while showing similar trends, do not translate directly to the marine realm when planning for conservation (Halpern & Warner 2003). Kark et al. (2009) explored collaboration of the Mediterranean terrestrial basin for terrestrial vertebrates and found that a fully coordinated conservation plan can save 45% of the total cost compared with an uncoordinated plan, whereas our coordinated marine plan delivered greater savings, up to three quarters. Area requirements were also largely reduced with increased collaboration on land (Kark et al. 2009), but were found to be almost constant in our marine study (Table 2.1). Kark et al. (2009) found that high-priority areas for terrestrial conservation efforts were concentrated in the European part of the Mediterranean. Here, we found that priority areas were spread throughout the Mediterranean Sea with no concentration in one geographical area (Fig. 2.5). We suggest that collaborative conservation efforts may be even more mutually beneficial and feasible in the marine realm, than the terrestrial realm, in this complex part of the globe.

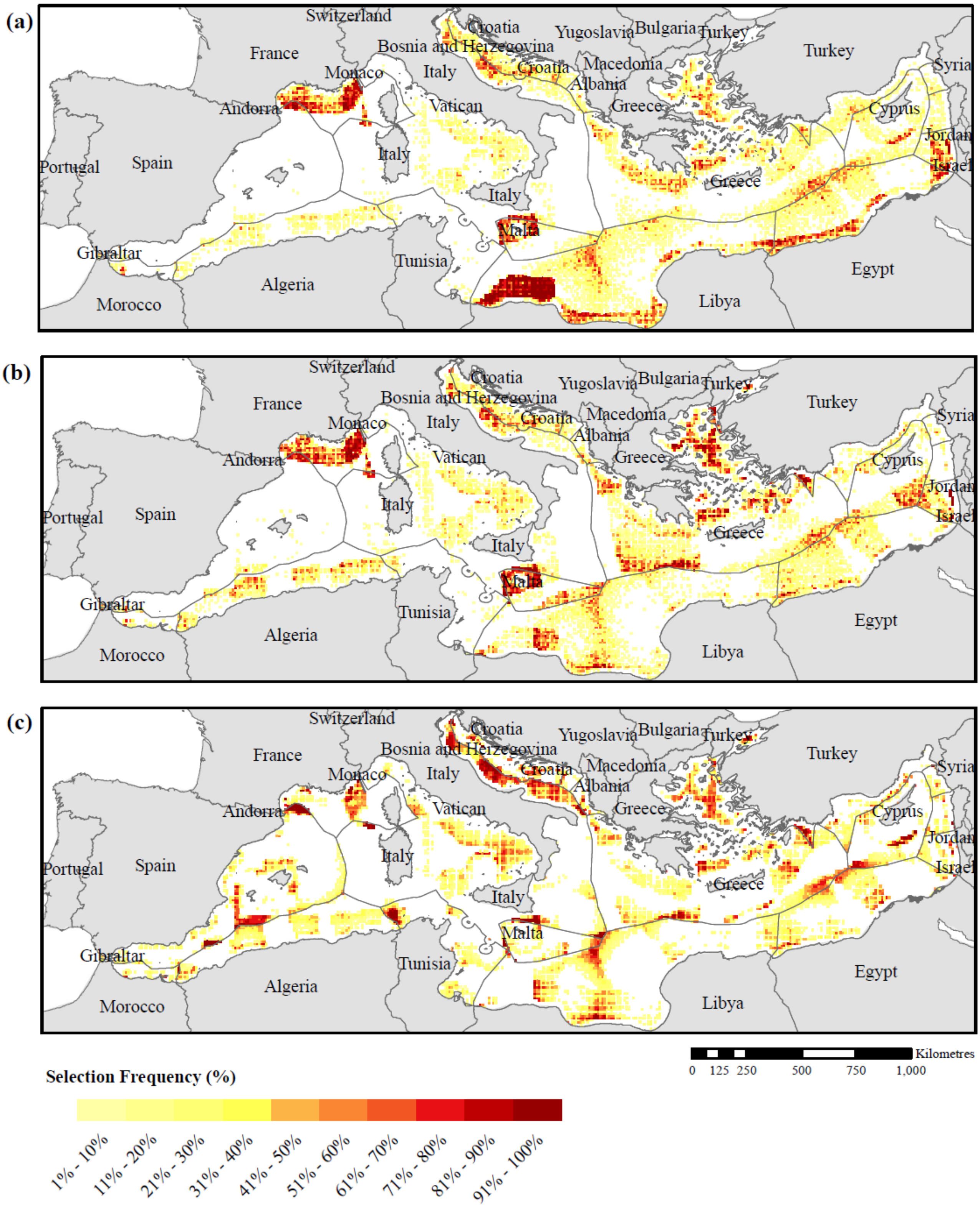


Figure 2.5. Three collaboration scenarios displaying the selection frequency for 10 'best solutions' (each best solution is from 1000 Marxan runs): (a) fully coordinated, (b) partly coordinated (coordination only between countries in each continent Europe, Asia and Africa), (c) uncoordinated (no coordination between Mediterranean countries). Each scenario protects 10% of the distribution of 77 threatened marine vertebrate species IUCN (2012). The cost layer used in this analysis

represents the opportunity cost of commercial fishing based on Food and Agriculture Organization (FAO) data (FAO 2011) and the distance to ports (National Geospatial-Intelligence Agency 2005), which is exponentially weighted by a constant $\alpha = 0.01$ and the opportunity cost of non-commercial fishing based on local population size (CIESIN 2005) and the relative Gross domestic product (GDP) of the country.

While we have shown that a fully coordinated plan is most cost-effective, establishing collaboration is hampered by economic, political or social barriers (Sandwith et al. 2001; Kark et al. 2009). Even a partly coordinated plan, where countries within each continent collaborate, is difficult to achieve in this socio-politically complex region. Large-scale spatial plans may be difficult to implement (Kark et al. 2009; Agardy et al. 2011), but our study indicates that even partial collaboration between countries can deliver huge benefits (Table 2.1). Therefore, one option is to plan collaboration across countries that already have established ties. In 2008, the European Commission announced the establishment of an MPA network to protect marine biodiversity in European waters following the Marine Strategy Framework Directive (European Commission 2008a). The European Union, where collaborative legislation and initiatives already exist, may provide a suitable platform for establishing collaboration (Kark et al. 2009). North African collaborations also exist for conservation (IUCN Centre for Mediterranean Cooperation 2012); most African countries in the region are linked to the Union of Arab Maghreb (UAM) Agreement and the Pan-African Parliament treaty (2006).

Interestingly, we found that multinational collaboration does not reduce conservation costs equally for every country when they collaborate. Such results are related to the high heterogeneity of costs (Fig. 2.2) and species (Fig. 2.1) in the Mediterranean Sea and are perhaps a likely outcome for complex regions where high diversity between countries exists. The majority of countries around the Mediterranean Sea saved money by engaging in a collaborative plan. However, for France, Libya and Malta, costs remained high despite collaboration (Figs 2.3 and 2.4). We find that these countries may have areas that remain high conservation priorities regardless of the level of collaboration (Fig. 2.5a,b), which means the benefits of collaboration for conservation are small or negative. While the benefits are inequitable, cross-country collaboration reduced costs for most Mediterranean countries and is far more efficient for the Mediterranean Sea as a whole. Because the costs and benefits between the collaborating countries are highly variable (Fig. 2.5), but the overall benefits are substantial, we believe that plans like this may require between-country compensations or subsidy measures. For example, Spain can gain from investing in a fully collaborative plan (*c.* US$1053 million), but to involve countries which do not gain, Spain may need to provide financial assistance or incentive to other countries. Despite the establishment costs, such countries may actually gain from investing in a collaborative plan in the long term when the benefits of marine reserves occur (Halpern & Warner 2003). Additional profits from establishing protected areas such as 'MPA spillover' where fish stocks within an MPA spill over to unprotected areas (increased fish biomass; Goñi et al. 2008; Stobart et al. 2009) and tourism profits (Agardy 1993) were not considered in this study and may in time cover the initial costs.

Large-scale conservation planning in a region with many countries that have different economics and data is challenging (Kark et al. 2009). Presently, there is poor availability of consistent socio-economic data within the Mediterranean basin at broad spatial scales. This is especially evident for recreational and subsistence fishing, where very little, if any, information is available at a country level (Lloret et al. 2008). In addition, there are unknown factors such as illegal fishing which may mean reported annual catches of some countries are actually higher (Coll et al. 2013). In making decisions for the entire Mediterranean Sea when there is little access or collaboration of data, using cost surrogates is a necessary alternative approach (Rodrigues & Brooks 2007). Our objective was to compare collaboration scenarios rather than provide a detailed conservation work plan. We had no restrictions on budgets; however, if this study was being explored as a conservation plan for the region, we would need to find more detailed economic data, including set budgets (e.g., budgets per country based on their capacity to contribute to marine conservation), transaction costs and the cost of protected area management. As more detailed and accurate data become available on species ranges and habitats for the entire Mediterranean, these can be incorporated into our methods. However, we expect that our finding of improved conservation efficiency when comparing a fully coordinated conservation plan with an uncoordinated plan will still hold true as tested by our four cost surrogates (Table 2.1) and supported by findings in Kark et al. (2009) and Moilanen et al. (2013) within terrestrial systems.

This is the first study that quantifies the benefits of between-country conservation in the marine realm. We found that conservation costs can largely be reduced if countries collaborate in the Mediterranean Sea. However, countries will not benefit from collaboration equally. This type of analysis could be valuable for decision-makers when considering the implementation of transboundary marine parks or multinational marine reserves and the allocation of international conservation funding for joint conservation agreements (e.g., the Convention on Biological Diversity (CBD 2013) signed by *c.* 190 countries with the aim to provide protection for at least 10% of each habitat type globally; Soutullo et al. 2008). Our approach is also helpful for assessing the potential benefits of collaboration as a way to engage and forge collaborative ties, particularly in areas where many countries or geopolitical divisions within a country (e.g., states) share marine waters and collaborative conservation is necessary. While collaboration among countries can be challenging, evaluating the costs and benefits of collaboration may provide incentives for partaking in collaborative action and its potential success. Incorporating collaboration into the systematic conservation planning framework is an important step for advancing such planning in the marine realm, delivering geographically applicable and efficient conservation outcomes.

2.6. Supplementary material

Table S2.1. Status of vertebrate taxa groups for the Mediterranean Sea. The "Total" indicates the number of species included in this study. Spatial data were from the IUCN (2012) and turtle nest data were from Casale and Margaritoulis (2012).

IUCN red list (Mediterranean Sea)	**Marine Mammals**	**Native Fishes** (jawless & bony)	**Seabirds**	**Sea turtles** (only nesting spp)	**Sharks and rays** (cartilaginous fishes)
Critically Endangered	1	1	1	-	14 (1)
Endangered	5	4 (2)	-	2	9
Vulnerable	1	7 (1)	1	-	7
Near Threatened	0	12 (1)	3 (1)	-	9 (1)
TOTAL	**7**	**24**	**5**	**2**	**39**
Least Concern	3	292 (39)	31	-	10
Data Deficient	2	126 (28)	0	-	25 (1)
*Distribution data unavailable – not included in study					• *Prionace glauca* (Vulnerable) • *Raja polystigma* (Near threatened)

Table S2.2. A list of all 77 species used in this study and their Mediterranean and global IUCN status. All spatial data were provided by the IUCN (2012) and turtle nesting site data (as distribution data for turtles is incomplete) was from Casale and Margaritoulis (2010). The inclusion of marine mammal species were based on Reeves and Notarbartolo-di-Sciara (2006), seabird species were based on Birdlife International (2012), native fishes and sharks and rays were based on Abdul Malak et al. (2011). Endemic species are marked with an asterisk (*).

Taxa group	Scientific name	Common name	IUCN Red List status	
			Mediterranean Red List	**Global Red List**
Marine mammals	*Balaenoptera borealis*	Sei whale	EN	EN
	Balaenoptera physalus	Fin whale	EN	EN
	Delphinus delphis (Mediterranean subpopulation)	Short-beaked common dolphin	EN	
	Eubalaena glacialis (vagrant in the Mediterranean Sea)	North Atlantic right whale	EN	EN
	Monachus monachus	Monk Seal	CR	CR
	Physeter macrocephalus	Sperm whale	VU	VU
	Tursiops truncatus ssp. ponticus	Common bottlenose dolphin	EN	
Native fishes (jawless and bony fishes)	*Dentex dentex*	Common dentex	VU	
	Dicentrarchus labrax	European seabass	NT	
	Epinephelus aeneus	White grouper	NT	NT
	Epinephelus marginatus	Dusky Grouper	EN	EN
	Hippocampus guttulatus	Long-snouted seahorse	NT	
	Hippocampus hippocampus	Short-snouted seahorse	NT	
	Labrus viridis	Green wrasse	VU	VU
	Merluccius merluccius	European hake	VU	
	Opeatogenys gracilis		VU*	VU*
	Platichthys flesus	European flounder	NT	
	Pleuronectes platessa	European plaice	NT	
	Pomatoschistus microps	Common goby	CR	
	Pomatoschistus minutus	Sand goby	VU	
	Pomatoschistus tortonesei		EN*	EN*
	Psetta maxima	Turbot	NT	
	Sciaena umbra	Brown meagre	VU	
	Scomber colias	Atlantic chub mackerel	NT	
	Syngnathus acus	Greater pipefish	NT	
	Syngnathus taenionotus	Dark-flank pipefish	EN*	EN*
	Syngnathus tenuirostris	Narrow-snouted pipefish	NT*	NT*
	Syngnathus typhle	Broad-nosed pipefish	NT	
	Thunnus thynnus	Atlantic bluefin tuna	EN	
	Umbrina cirrosa	Shi drum	VU	

	Xiphias gladius	Swordfish	NT	
Seabirds	*Larus audouinii*	Audouin's Gull	NT	NT
	Pelecanus crispus	Pelican	VU	VU
	Puffinus griseus	Sooty Shearwater	NT	NT
	Puffinus mauretanicus	Balearic Shearwater	CR	CR
	Puffinus yelkouan	Yelkouan Shearwater	NT*	NT*
Sea turtles	*Caretta caretta*	Loggerhead sea turtle	EN	EN
	Chelonia mydas	Green sea turtle	EN	EN
Sharks and rays (cartilaginous fishes)	*Alopias vulpinus*	Long-tailed, common thresher shark	VU	VU
	Carcharhinus plumbeus	Sandbar shark	EN	VU
	Carcharias taurus	Sand tiger shark	CR	VU
	Carcharodon carcharias	Great white shark	EN	VU
	Centrophorus granulosus	Gulper shark	VU	VU
	Cetorhinus maximus	Basking shark	VU	VU
	Dasyatis centroura	Roughtail stingray	NT	
	Dasyatis pastinaca	Common stingray	NT	
	Dipturus batis	Common skate	CR	CR
	Dipturus oxyrhynchus	Longnosed skate	NT	NT
	Gymnura altavela	Spiny butterfly ray	CR	VU
	Heptranchias perlo	Harpnose sevengill shark	VU	NT
	Hexanchus griseus	Bluntnose sixgill shark	VU	NT
	Isurus oxyrinchus	Ahortfin mako shark	CR	VU
	Lamna nasus	Porbeagle shark	CR	VU
	Leucoraja circularis	Sandy skate	CR	VU
	Leucoraja fullonica	Shagreen ray	NT	NT
	Leucoraja melitensis	Maltese skate	CR*	CR*
	Leucoraja naevus	Cuckoo ray	NT	
	Mobula mobular	Giant devil ray	EN	EN
	Mustelus asterias	Starry smooth-hound	EN	
	Mustelus mustelus	Common smooth-hound	EN	VU
	Myliobatis aquila	Common eagle ray	NT	
	Odontaspis ferox	Small tooth sand tiger, bumpy tail ragged-tooth	VU	VU
	Oxynotus centrina	Angular rough shark	CR	VU
	Pteroplatytrygon violacea	Pelagic stingray	NT	LC
	Pristis pectinata	Small-tooth sawfish	CR	CR
	Pristis pristis	Common sawfish	CR	CR
	Raja clavata	Thornback ray	NT	NT
	Raja undulata	Undulate Ray	EN	EN
	Rhinobatos cemiculus	Black-chin guitarfish	EN	EN
	Rhinobatos rhinobatos	Common guitarfish	EN	EN
	Rostroraja alba	Bottlenose Skate, Spearnose Skate, White Skate	CR	EN
	Scyliorhinus stellaris	Nursehound	NT	NT

	Sphyrna zygaena	Smooth hammerhead	VU	VU
	Squalus acanthias	Spiny dogfish, spurdog, mud shark, or piked dogfish	EN	VU
	Squatina aculeata	Sawback Angelsharks	CR	CR
	Squatina oculata	Smoothback angelshark	CR	CR
	Squatina squatina	Angelshark	CR	CR

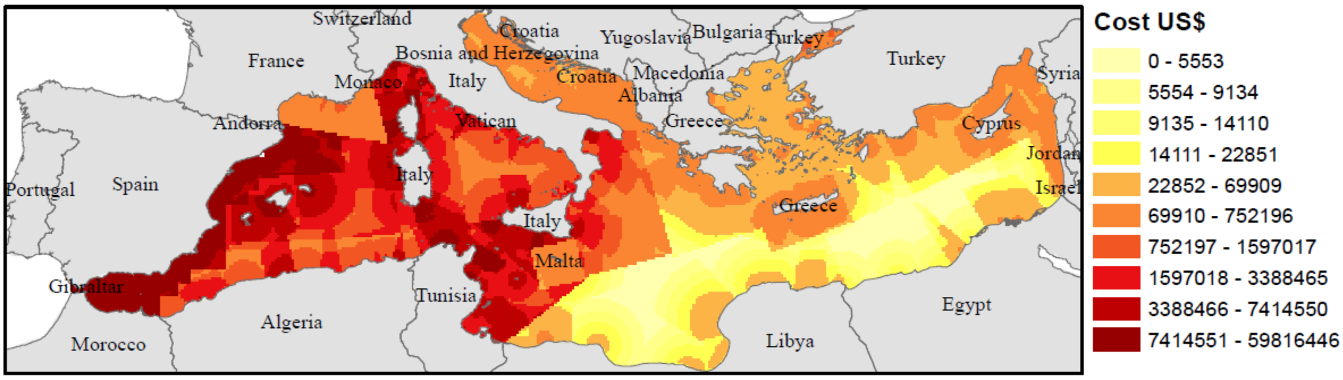
Table S2.3. Average cost of ten best solutions (1 best solution derived from 1000 Marxan runs) for each country and the three collaboration scenarios: fully coordinated scenario (all countries collaborate), partly coordination scenario (countries from each continent collaborating), and uncoordinated scenario (countries act independently).

	Average cost of 10 best solution (US$ million)		
Country	Fully coordinated scenario	Partly coordinated scenario	Uncoordinated scenario
Albania	1.86	2.68	4.36
Algeria	65.80	141.41	110.89
Bosnia and Herzegovina	0	0	-
Croatia	10.82	10.27	6.20
Cyprus (EU)	3.66	5.39	3.23
Egypt	3.90	1.71	6.09
France (EU)	113.23	122.40	28.02
Gibraltar (EU)	0	0	-
Greece (EU)	22.06	29.58	16.72
Israel	11.40	151.59	77.62
Italy (EU)	138.93	217.59	324.25
Lebanon	1.12	42.89	11.62
Libya	27.54	14.66	4.81
Malta (EU)	19.80	27.16	5.90
Monaco	0.43	0.45	-
Montenegro	2.17	1.58	1.89
Morocco	104.11	121.78	178.34
Slovenia (EU)	0	0	-
Spain (EU)	66.65	169.41	1,119.93
Syria	0.52	5.58	4.10
Tunisia	16.47	17.87	213.63
Turkey	3.43	55.56	17.85

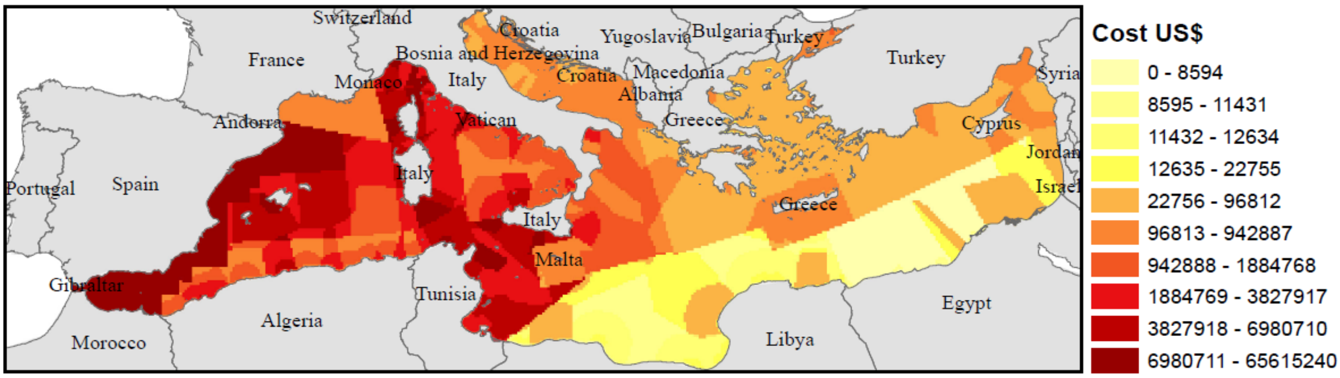
Table S2.4. Cost of marine conservation if a country acts independently to protect 10% of the distribution of 77 threatened marine vertebrate species (IUCN 2012). This table compares four different cost layers that represent opportunity cost of commercial and non-commercial (recreational and subsistence) fishing.

Country	**Average cost of 10 best solution (US$ million)**			
	Cost Layer (a) Commercial fishing (decay rate α = 0.001) and non-commercial fishing uses GDP values	**Cost Layer (b)** Commercial fishing (decay rate α = 0.01) and non-commercial fishing uses GDP values	**Cost Layer (c)** Commercial fishing (decay rate α = 0.05) and non-commercial fishing uses GDP values	**Cost Layer (d)** Commercial fishing (decay rate α = 0.01) and non-commercial uses an equal value ($50) for all countries.
Albania	4.43	4.36	4.10	7.98
Algeria	123.58	110.89	39.16	110.69
Croatia	6.76	6.20	2.50	5.93
Cyprus (EU)	3.25	3.23	1.48	2.85
Egypt	6.38	6.09	5.17	14.95
France (EU)	28.3	28.02	19.99	23.99
Greece (EU)	19.08	16.72	4.28	15.75
Israel	77.62	77.62	77.27	46.3
Italy (EU)	363.55	324.25	53.325	318.59
Lebanon	11.80	11.62	11.08	13.44
Libya	4.78	4.81	0.98	4.02
Malta (EU)	8.99	5.90	0.99	6.19
Montenegro	1.82	1.89	1.46	2.26
Morocco	186.94	178.34	104.02	183.31
Spain (EU)	1,090.55	1,119.93	789.04	1,111.09
Syria	4.12	4.10	3.83	12.21
Tunisia	201.34	213.63	87.16	204.58
Turkey	18.92	17.85	13.12	20.19
Total Cost	**2,162**	**2,540**	**1,219**	**2,104**

(a)



(b)



(c)

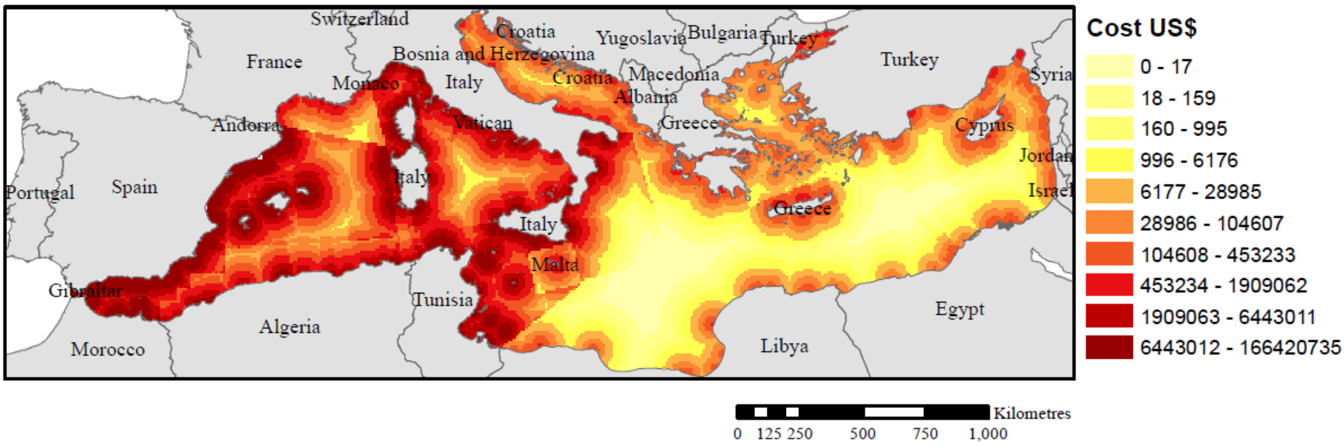
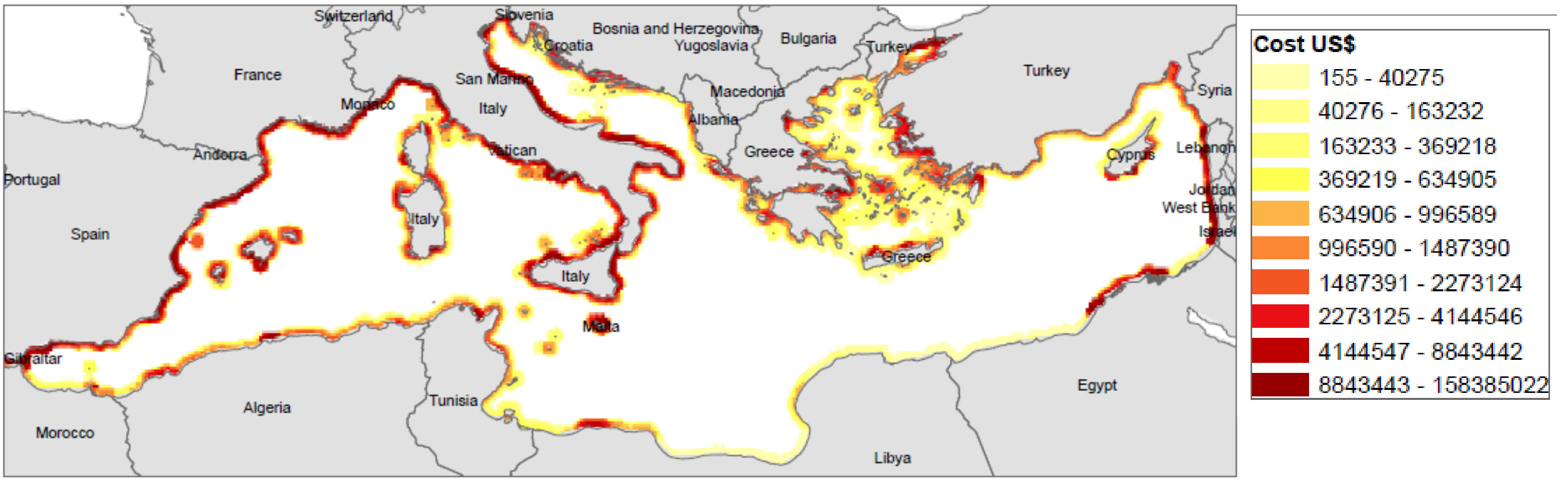


Figure S2.1. Cost layers (annual revenue in US$) of commercial fishing within the Mediterranean Sea. The cost values are displayed by quantile range at a spatial resolution of 100 km^2: (a) commercial fishing cost layer with decay rate $\alpha = 0.001$ from ports, (b) commercial fishing cost layer with decay rate $\alpha = 0.01$ from ports and (c) commercial fishing cost layer with decay rate $\alpha = 0.05$ from ports. Port data were from the National Geospatial-Intelligence Agency (2005).

a)



b)

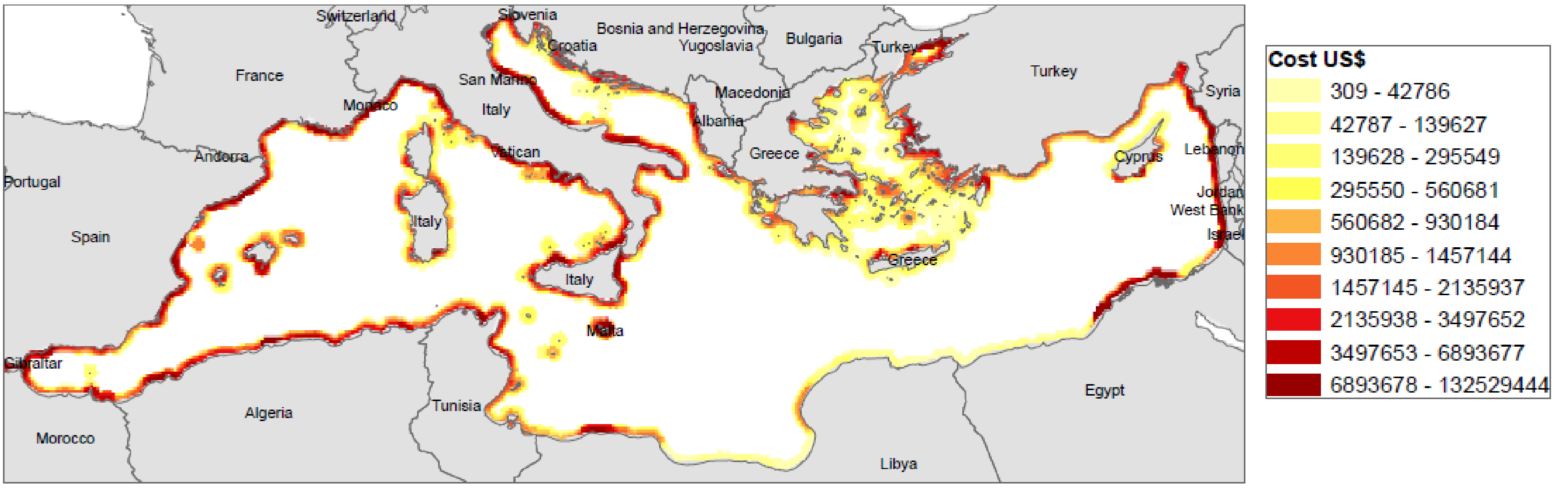


Figure S2.2. Cost layers (annual revenue in US$) of non-commercial fishing (recreational and subsistence fishing) within the Mediterranean Sea. The cost values are displayed by quantile range at a spatial resolution of 100 km^2: (a) non-commercial fishing layer (using GDP per country) and (b) non-commercial fishing layer (using a constant of US$50 per country).

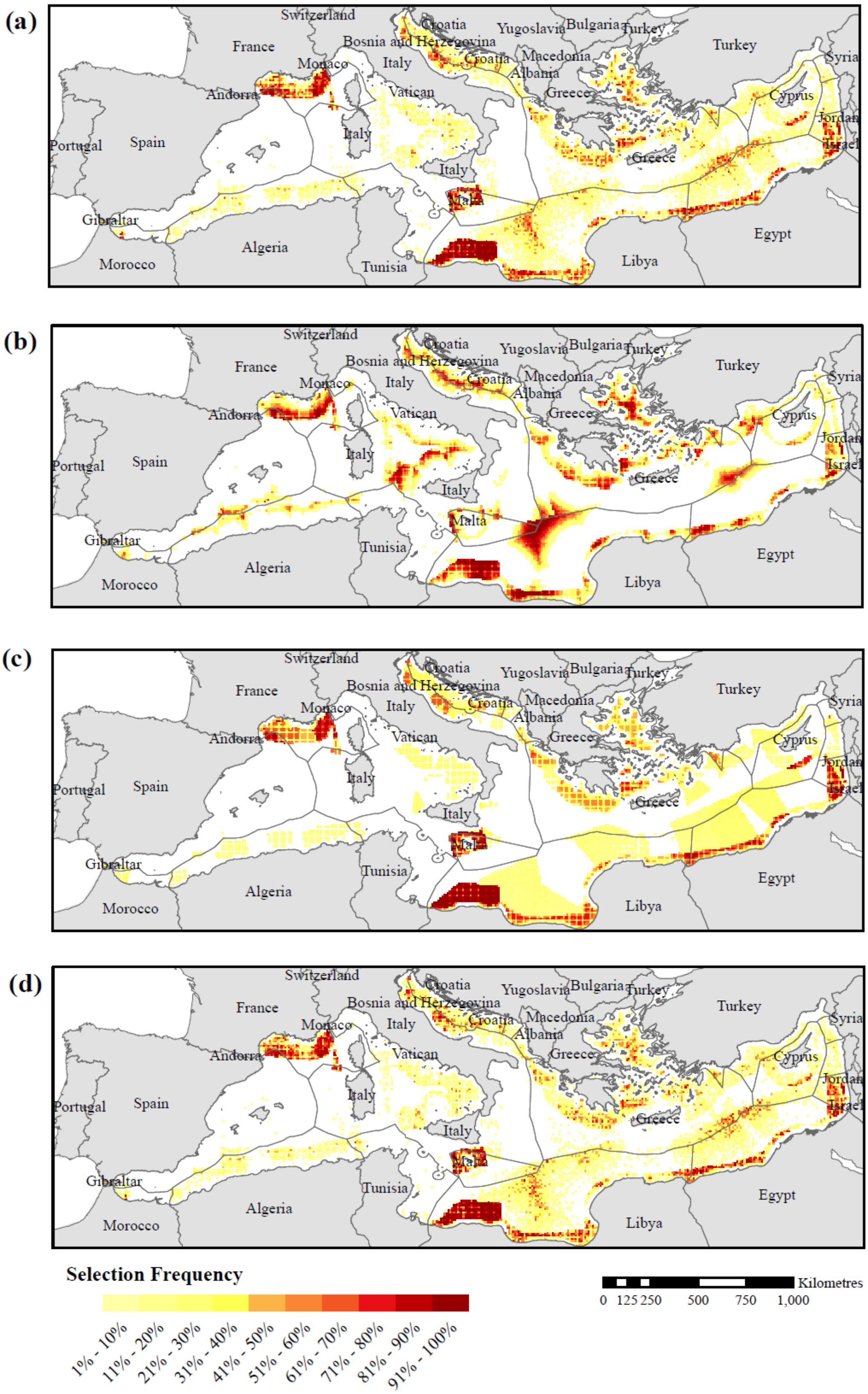


Figure S2.3. Comparison of the selection frequency outputs (from Marxan) of different cost layers a) Commercial fishing ($\alpha = 0.01$) and non-commercial fishing uses GDP values b) Commercial fishing ($\alpha = 0.05$) and non-commercial fishing uses GDP values c) Commercial fishing ($\alpha = 0.001$) and non-commercial fishing uses GDP values d) Commercial fishing ($\alpha = 0.01$) and non-commercial fishing uses an equal value ($50) for all countries.

Chapter 3

Large-scale conservation planning in a multinational marine environment: cost matters

Jaffa Port, Israel. Photo credit: T.Mazor

Ecological Applications (2014) **24**, 1115–1130

3.1 Abstract

Explicitly including cost in marine conservation planning is essential for achieving feasible and efficient conservation outcomes. Yet, spatial priorities for marine conservation are still often based solely on biodiversity hotspots, species richness, and/or cumulative threat maps. This study aims to provide an approach for including cost when planning large-scale Marine Protected Area (MPA) networks that span multiple countries. Here, we explore the incorporation of cost in the complex setting of the Mediterranean Sea. In order to include cost in conservation prioritization, we developed surrogates that account for revenue from multiple marine sectors: commercial fishing, non-commercial fishing, and aquaculture. Such revenue can translate into an opportunity cost for the implementation of an MPA network. Using the software Marxan, we set conservation targets to protect 10% of the distribution of 77 threatened marine species in the Mediterranean Sea. We compared nine scenarios of opportunity cost by calculating the area and cost required to meet our targets. We further compared our spatial priorities with those that are considered consensus areas by several proposed prioritization schemes in the Mediterranean Sea, none of which explicitly considers cost. We found that for less than 10% of the Sea's area, our conservation targets can be achieved while incurring opportunity costs of less than 1%. In marine systems, we reveal that area is a poor cost surrogate and that the most effective surrogates are those that account for multiple sectors or stakeholders. Furthermore, our results indicate that including cost can greatly influence the selection of spatial priorities for marine conservation of threatened species. Although there are known limitations in multinational large-scale planning, attempting to devise more systematic and rigorous planning methods is especially critical give that collaborative conservation action is on the rise and global financial crisis restricts conservation investments.

3.2 Introduction

An important and often overlooked component of marine conservation planning is the inclusion of conservation cost. Incorporating cost is necessary for delivering feasible conservation outcomes and for ensuring the successful implementation of Marine Protected Areas, MPAs (Lundquist & Granek 2005; Stewart & Possingham 2005; Ban & Klein 2009). However, cost is by no means a new concept in conservation planning. The well-known framework of systematic conservation planning enables us to incorporate cost and other social, economic, and political aspects (Pressey & Bottrill 2009; Micheli et al. 2013b). Previous studies have also presented methods for integrating cost into planning for the selection of marine conservation priorities (e.g., Klein et al. 2008a; Ban et al. 2009; Klein et al. 2010; Giakoumi et al. 2011). These methods enable us to make more achievable conservation plans that improve conservation efficiency by maximizing biodiversity and reducing cost. Despite this, to date there are still numerous plans for marine reserves and priority marine conservation areas that are produced without a measure of cost (Naidoo et al. 2006; Ban et al. 2011; Micheli et al. 2013b).

There are several types of cost that can be included in marine conservation planning. These include management cost (Balmford et al. 2004; Klein et al. 2010), monitoring cost (Gerber et al. 2005), transaction cost (Naidoo et al. 2006), and opportunity cost (Giakoumi et al. 2011). The most commonly accounted for and significant cost in marine planning is opportunity cost (Ban & Klein 2009). Opportunity cost is the forgone cost (or in other words, the lost benefit) when an activity takes place where another has occurred or can occur (e.g., fishing profits that are forgone when an area is made a closed/no take MPA; Cameron et al. 2008). There are also several forms of opportunity cost to consider, including commercial and recreational activities such as diving, boating, tourism, and fishing, as well as infrastructure cost such as offshore oil and gas production (Naidoo et al. 2006). The opportunity cost that is most commonly accounted for when planning marine conservation is related to fishing (Ban & Klein 2009). Yet, few studies, if any, have attempted to deal with opportunity cost over large-scale marine environments with multiple countries characterized by high heterogeneity in data availability.

The absence of cost data within many marine conservation plans is partly due to the challenge of quantifying and incorporating this component. This is especially the case in data-poor regions, large areas, and multinational environments. One of the first hurdles is to utilize and translate data related to human economic activities into cost values (when such values are absent) for use within conservation plans. Indeed, this can be a difficult task for biologists, ecologists, and conservation planners who may have no formal education in the field of economics (Naidoo et al. 2006). Second, finding economic data that are spatially explicit can be difficult; such data are often non-existent, especially over large scale areas (Naidoo et al. 2006; Ban & Klein 2009). In these circumstances, we must often turns towards developing surrogates for cost (Ando et al. 1998; Ban et al. 2009; Giakoumi et al. 2011). Thirdly, other challenges emerge when we explore conservation planning across different states, national jurisdictions, or countries (Kark et al. 2009; Mazor et al. 2013b). The ability to find explicit cost data that are compatible and comparable between various jurisdictions or countries with different socioeconomic status becomes more difficult. Nevertheless, as marine conservation planning begins to expand to larger spatial scales for the development of marine protected networks that encompass several countries (Miclat et al. 2006; Douvere 2008), we cannot ignore cost, the socioeconomic context in which our biodiversity goals exist (Polasky 2008).

3.2.1 Mediterranean Sea conservation planning

Large-scale conservation plans are arising in the marine realm, particularly for waters shared by multiple countries, such as the Mediterranean Sea (e.g., Notarbartolo di Sciara & Agardy 2009; CIESM 2011; Oceana 2011). The multiple number of large-scale plans for the Mediterranean Sea that have recently emerged focus mostly on identifying priority areas for protecting threatened species or habitats that span across multiple countries (Micheli et al. 2013b). Yet, no large-scale conservation plans for the Mediterranean Sea have explicitly included cost (Giakoumi et al. 2012b; Micheli et al. 2013b). Only several small-scale Mediterranean studies have addressed the cost of marine conservation within the framework of systematic conservation planning (Fraschetti et al. 2009; Maiorano et al. 2009; Giakoumi et al. 2011; 2012a). Large-scale planning is important for the Mediterranean Sea (Portman et al. 2013), but without incorporating cost, the ability of plans to aid decision makers can only go so far. To better direct and inform decision makers, there is a need for systematic methods that are driven by explicit objectives and translate into actions and costs (Margules & Pressey 2000; Moilanen et al. 2009).

The Mediterranean Sea supports the livelihood of millions of people via the exploitation of its living marine resources (Abdulla et al. 2008; Madau et al. 2009). The gross value of marine resources from lagoon and marine fishing and aquaculture in Mediterranean countries was estimated at US$6.3 billion for 2008 (Sacchi 2011). Fishing also has a great social and cultural value for most Mediterranean countries (Farrugio et al. 1993). Therefore, when we aim to protect biodiversity in the Mediterranean Sea, we must take into account the importance of this prevalent economic and cultural activity. A possible reason that this has never been accounted for at a whole-basin scale before is that the Mediterranean Sea is a collection of different countries with huge differences in socioeconomic status, political regimes, languages, governance, and cultures (Badalamenti et al. 2000; Giakoumi et al. 2012b). A major challenge is the standardization of data at a basin level, because there is a striking imbalance of available information. Data availability itself presents a challenge, as there is a negative gradient from the north to the south as well as from the west to the east of the Mediterranean Basin (Abdulla et al. 2008; Coll et al. 2012; Micheli et al. 2013a).

Here, we aim, for the first time at the scale of the whole Mediterranean Sea, to explicitly account for cost in conservation planning. We develop an approach for incorporating opportunity cost of exploitation of marine resources at large spatial scales within heterogeneous systems. We address three major sectors of marine exploitation: commercial fishing (including industrial and artisanal fishing), non-commercial fishing (recreational and subsistence), and aquaculture. Our objective is to provide an approach that allows one to include cost when planning large-scale MPA networks that span multiple countries. We aim to explore how the explicit consideration of cost alters conservation priority areas across the Mediterranean Sea. Furthermore, we will compare our results with consensus areas of 12 Mediterranean prioritization schemes that did not account for cost (Micheli et al. 2013b).

3.3 Methods

3.3.1 Spatial extent and species information

Our study area comprised the entire Mediterranean Sea. We divided the area into 10 x 10 km planning units (26,946 in total). This resolution was chosen to comply with the EU guidelines on the use of a Pan-European grid of 10 x 10 km for spatial planning (Directive 2007/ 2/EC of the European Parliament and of the Council of 14 March 2007 establishing an Infrastructure for Spatial Information in the European Community [Inspire]) and based on our previous work in the Mediterranean Basin (Kark et al. 2009; Mazor et al. 2013b). Spatial distribution data were available for 77 threatened marine vertebrate species in the Mediterranean Sea. These included marine fishes, sharks and rays, marine mammals, sea birds, and sea turtle nesting sites (Appendix S3.1: Table S3.1; see IUCN 2012). We projected all available species data into Albers Equal Area Projection at the planning unit scale, using ArcGIS software (Appendix S3.1: Fig. S3.1).

3.3.2 Opportunity cost surrogates

We derived surrogate cost layers to represent the opportunity cost (cost of establishing an MPA in a given area) of different marine sectors including fisheries and aquaculture activities. The three sectors included were: commercial (both industrial and artisanal) fishing, non-commercial (recreational and subsistence) fishing, and aquaculture. For each of these three sectors we developed equations to calculate opportunity cost in monetary terms (€) for each planning unit (Fig. 3.1). We used these opportunity cost layers separately or in combination (summed together) to give a total of nine scenarios of cost (see Table 3.1). These scenarios were used to quantify the benefit of planning conservation by using area (area of planning unit) as a cost and including single vs. multiple sectors cost. Each opportunity cost layer in these scenarios will be described.

Table 3.1. Nine opportunity cost scenarios (the cost of establishing an MPA in a given area) used in this study of conservation planning in the Mediterranean Sea.

Opportunity cost scenarios	**Scenario name**	**Opportunity cost layers included in scenario (summed layers)**
Scenario 1	Area as cost	Area of planning unit (km^2)
Scenario 2	Commercial fishing GFCM	Commercial fishing GFCM (General Fisheries Commission for the Mediterranean; FAO 2011)
Scenario 3	Commercial fishing SAUP	Commercial fishing SAUP (Sea Around Us Project 2011)
Scenario 4	Non-commercial fishing A	Non-commercial fishing A (cost of expenditure parameter b = 0.5)
Scenario 5	Non-commercial fishing B	Non-commercial fishing B (cost of expenditure parameter b = 1)
Scenario 6	Combined sectors A	Commercial fishing GFCM + Non-commercial A (b = 0.5) + Aquaculture
Scenario 7	Combined sectors B	Commercial fishing SAUP + Non-commercial A (b = 0.5) + Aquaculture
Scenario 8	Combined sectors C	Commercial fishing GFCM + Non-commercial B (b = 1) + Aquaculture
Scenario 9	Combined sectors D	Commercial fishing SAUP + Non-commercial B (b = 1) + Aquaculture

3.3.2.1 Commercial fishing layers

Here we developed two different cost layers to represent the opportunity cost of commercial fishing, using data provided from two different sources (Appendix S3.2: Fig. S3.2). The first cost layer is based upon biomass of fish caught over 28 different geographical regions, data provided by the General Fisheries Commission for the Mediterranean, GFCM (FAO 2011). The second cost layer uses fish landings in monetary values for 22 counties, with data provided by the Sea Around Us Project, SAUP (Sea Around Us Project 2011).

i) GFCM cost layer

The opportunity cost of commercial fishing was calculated as the combined cost of (1) small-scale fishing that occurs close to the coast and (2) large-scale fishing in deeper waters (for full methods, see Appendix S3.2; Table S3.2). To calculate the opportunity cost of this sector, we developed an equation (a simplified approach was used in Mazor et al. 2013b) where C_i is the annual value, and thus opportunity cost of commercial fishing in each planning unit i. We assumed that the opportunity cost is proportional to the size of the nearest port, PS (for port sizes, see National Geospatial-Intelligence Agency 2005), and decreases exponentially with distance d from port weighted exponentially by a constant α (0.01) and the area A of planning unit i. We used annual tonnage data regarding total fishing from 28 Geographical Sub-Areas (GSAs) as reported by the General Fisheries Commission for the Mediterranean GFCM for 2008 (FAO 2011). This is the most current, spatially available data for the entire Mediterranean Sea on fish catch. To ensure that the total value of catch in each region (28 GSA regions) sums to its real value, we normalized the cost of commercial fishing in each planning unit by a measure of total regional effort E_{R}:

$$E_{\mathrm{R}} = \sum_{i=1}^{m} \mathrm{PS}\, e^{-\alpha d} A_i ,$$

where m is the number of planning units in a given region. We multiplied the final value by the total production of fish in the region (in metric tons) B_{R}, multiplied by the value of fish (€ per metric ton) V_{fish}, such that the final expression for an estimate of the opportunity cost for commercial fishing C_i in each planning unit i is

$$C_i = \left(\frac{\mathrm{PS}\, e^{-\alpha d} A_i}{E_{\mathrm{R}}}\right) V_{\mathrm{fish}} B_{\mathrm{R}} .$$

For small-scale (artisanal) commercial fishing, we included data on the tonnage of fish extracted via small-scale vessels without engines (vessels <12 m long), small-scale vessels with engines (>6 m and 6–12 m long) and polyvalent (i.e., multipurpose) vessels (>12 m long) (FAO 2011). Boat length was not considered an absolute criterion because in most countries, polyvalent vessels longer than 12 m that use longline and gillnet fishing can be considered as practicing artisanal fishing (Sacchi 2011). The total value was multiplied by the average price (V_{fish} = 12.61 €/kg for 2010; prices available online)[3] of five fish species (*Mullus surmuletus*, *Sparus aurata*, *Serranus cabrilla*, *Scorpaena scrofa*, *Sarda sarda*) that compose the majority of artisanal fisheries catch as reported by Lloret and Font (2013). Although the catch composition may vary throughout the Mediterranean Sea, we consider that the average value of the estimated fish catch is representative for most Mediterranean countries. Small-scale commercial fishing takes place within a country's 12 nautical mile territorial waters (IEEP 2002; Morales-Nin et al. 2005); therefore, we only included planning units (8964 planning units) that were between the coastline and a distance of 22 km (~12 nautical miles).

For large-scale (semi-industrial and industrial) commercial fishing, we calculated the total tonnage of large-scale commercial fishing vessels in 2008 as reported by the FAO (2011). We used the price of five fish species that are major species targeted in commercial fishing (Lleonart & Maynou 2003; European Commission 2008b) in the Mediterranean Sea and relate to four particular fishing gear types: trawlers and dredgers (*Merluccius merluccius*, $V_{\text{fish}} = 7.02$ €/kg; Garcia-Vazquez et al. 2011); purse seiners and pelagic trawlers (>6 m long) (*Engraulis encrasicolus* and *Sardina pilchardus*, average $V_{\text{fish}} = 2.38$ €/kg; FAO 2010); long-liners (>6 m long) (*Xiphias gladius*, $V_{\text{fish}} = 5.40$ €/kg; FAO 2010); and tuna seiners (<12 m long) (*Thynnus thynnus*, $V_{\text{fish}} = 17.25$ €/kg; FAO 2010).

ii) SAUP cost layer

Here we used data provided by the Sea Around Us Project (SAUP) on annual landings (US$ per ton) for each county (22 countries) surrounding the Mediterranean Sea for the year 2006 (Sumaila et al. 2007; Sea Around Us Project 2011). These data are the most current, available data for the entire Mediterranean Sea reporting monetary values of fish landings at the country level. We assume here that each country's landings are from its own Exclusive Economic Zone, EEZ (as defined by VLIZ 2012), although a small amount of this catch may come from nearby geographical areas due the lack of supervision across marine boarders or permission from other countries to fish in their waters (FAO 2011). We use an equation similar to that in the previous cost layer by assuming that the opportunity cost is proportional to the size of the nearest port, decreases exponentially with distance, and is weighted by area. However, here we divided this by a country effort, E_{C}, rather on a regional effort. Also, because we have the value of the annual landings (US$) per country we multiplied our effort by the reported value V of each country. Thus, the opportunity cost for commercial fishing C in each planning unit i is defined as:

$$C_i = \left(\frac{\text{PS}\, e^{-\alpha d}\, A_i}{E_{\text{C}}}\right) V \,,$$

[3] http://en.fishprices.net/home

where

$E_C = \sum_{i=1}^{m} \text{PS}\, e^{-\alpha d} A_i$.

To make our opportunity cost layer comparable to other cost layers, we converted our resulting values from U.S. dollars to Euros using the average annual exchange rate for the year 2006 as reported by the International Monetary Fund (available online)[4].

3.3.2.2 Non-commercial fishing layers

Here we developed an equation where C_{Ni} is the annual value (thus opportunity cost) of non-commercial fishing in each planning unit *i*. We summed the cost of expenditure of fishing per year C_{exp} with the value of catch V_{catch}. The C_{exp} was used to estimate the value that recreational fishers give to recreational fishing through their purchases in the related markets, e.g., recreational vessel purchases and recreational fishers participating in this activity through their revealed preference (hedonic method; see Gaudin & De Young 2007). The V_{catch}, which can be considered a benefit increasing the value of the recreational fishing, was calculated by multiplying the number of fishing days per year, the total number of kilograms of fish per day, and the value of fish (€ per kilogram). Because the cost of recreational fishing includes both V_{catch} and C_{exp}, it may be argued that one of these is more or less important than the other for determining the value of recreation fishing. Therefore, we introduce parameter *b*, where $0 \le b \le 1$, to allow us to test different weightings. The resulting value is multiplied by the number of fishers per year N_f. The opportunity cost for non-commercial fishers per planning unit *i* is

$$C_{Ni} = \sum (b\, C_{exp} + (1-b) V_{catch})\, N_{fi}$$

and the number of fishers per planning unit, where SS is settlement size, is equal to

$$N_{fi} = \left(\frac{\text{SS}\, e^{-\alpha d} A_i}{E_R}\right) * N_{fR} \, ,$$

where,

$E_R = \sum_{i=1}^{m} \text{SS}\, e^{-\alpha d} A_i$.

We assume that opportunity cost is proportional to the size of the nearest settlement, SS (using 2011 data from Columbia University's Center for International Earth Science Information Network, available online)[5] and that it decreases exponentially with distance *d* from the settlement by a constant α and the area *A* of planning unit *i*. Due to the unavailability of data for non-commercial fishers per country, we used surrogates. In our study, C_{exp} was 1376 € (Ünal et al. 2010) and was adjusted for each country based on purchasing power parity (PPP) rates reported by The World

[4] http://www.imf.org/external/np/fin/ert/GUI/Pages/CountryDataBase.aspx
[5] http://sedac.ciesin.columbia.edu/data/set/grump-v1-settlement-points

Bank (available online)[6]. Conversion from US$ to Euro was via the average annual exchange value for 2010 by the IMF. For V_{catch}, we used a constant of 60 fishing days per year (Ünal et al. 2010) and multiplied this by 5 kg per day which is the maximum allowed mass in most Mediterranean countries (Gaudin & De Young 2007). We used the price of 5.95 €/kg, which is the average price of 10 fish species that compose 99% of the recreational catch, as reported by Tunca et al. (2012). Two values, 0.5 and 1.0, were used for *b*, thus producing two cost layers for non-commercial fishing: non-commercial fishing A and non-commercial fishing B, respectively (Appendix S3.2: Fig. S3.3). For N_{f}, we used a method used by Mazor et al. (2013b), assuming that 10% of the population goes fishing (CFCM 2010; Ünal et al. 2010; Herfaut et al. 2013). We also limited our spatial extent to planning units within 12 nautical mile territorial waters, as performed in the small-scale commercial fishing layer, giving a total of 8964 planning units in our layer.

3.3.2.3 Aquaculture layer

To spatially represent the cost of aquaculture in the Mediterranean Sea, we used data from Trujillo et al. (2012). This is currently the best available data that exist for aquaculture locations in the Mediterranean Sea. Here we calculated the area (in square kilometers) occupied by aquaculture pens, A_{AQ}, in each country using ArcGIS software (ESRI 2010). This was further divided by the sum of each country's total surface area A_{AQC} dedicated to aquaculture. The resulting value was then multiplied by the annual aquaculture production P_{AQ} (in metric tons) in 2006 as reported by Trujillo et al. (2012) for each country. To retrieve monetary values, for each country we multiplied its production *P* by the cost *C* of the two primary aquaculture species in the Mediterranean: seabream *Sparus aurata* (4.25 €/kg; FAO 2010) and seabass *Dicentrarchus labrax* (4.75 €/kg; FAO 2010; Trujillo et al. 2012). Following Trujillo et al. (2012), we have excluded tuna cages due to the relatively small number of cages in the Sea and because their productivity success is not well established in this region. The overall equation for estimating the opportunity cost for aquaculture C_{AQ} in each planning unit *i* is

$$C_{\mathrm{AQ}i} = \frac{A_{\mathrm{AQ}}}{\sum A_{\mathrm{AQC}}} PC.$$

To validate our resulting cost surrogate for the year 2006, we compared our results with those of the closest year we could find, reported in 2008 by FAO (Sacchi 2011; see Appendix S3.2; Table S3.3). Our resulting cost layer is similar to that of 2008 and seems to be an underestimation rather than an overestimation of aquaculture production (Appendix S3.2; Table S3.3).

[6] http://data.worldbank.org/indicator/NY.GNP.PCAP.PP.CD

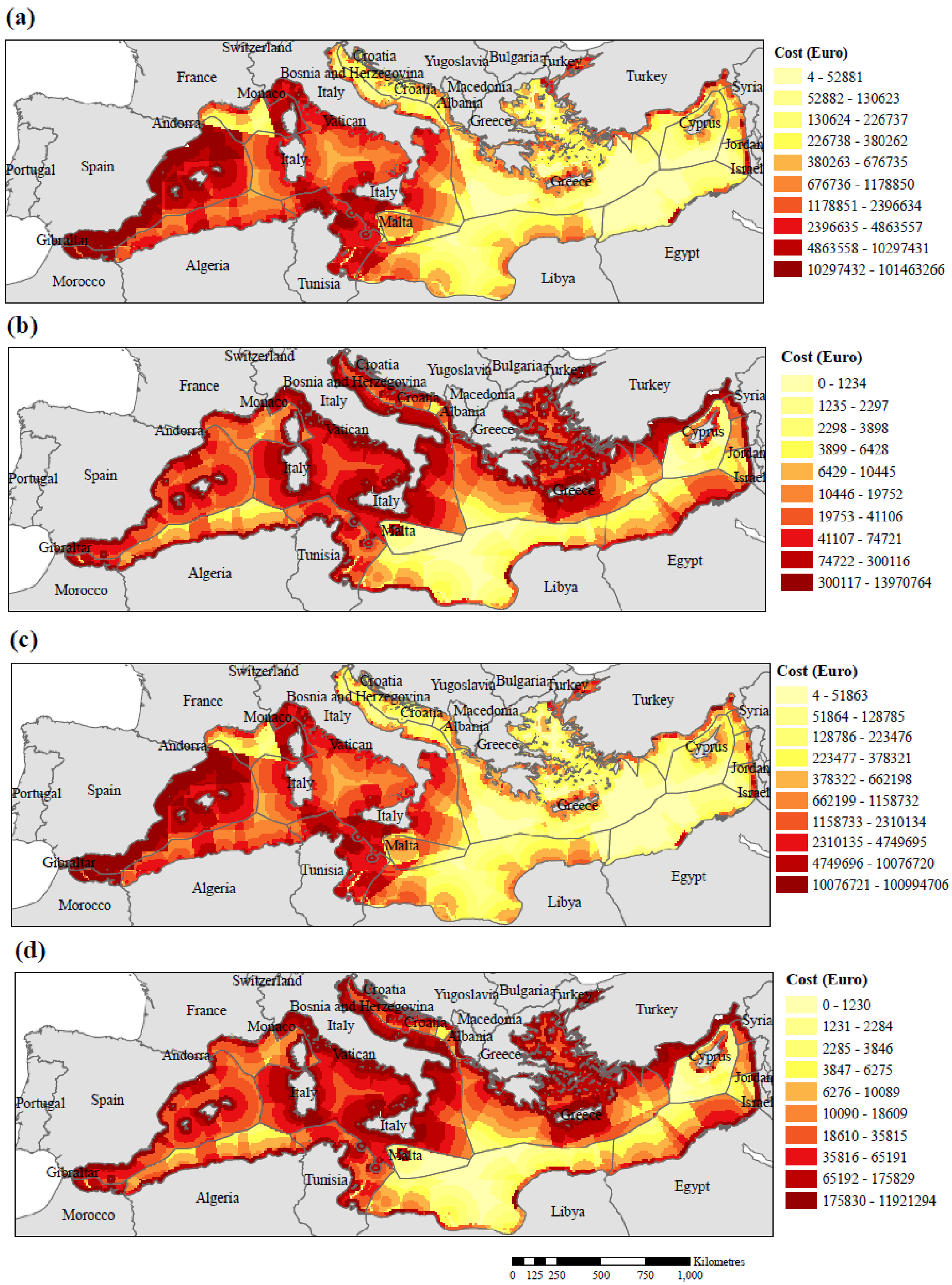


Figure 3.1. The final combined sector cost layers (opportunity cost in €/10 km^2 displayed by a quantile range) when all marine sectors (commercial, non-commercial, and aquaculture) in the Mediterranean Sea are combined: (a) combined sectors A (scenario 6), (b) combined sectors B (scenario 7), (c) combined sectors C (scenario 8), and (d) combined sectors D (scenario 9). See Table 3.1 for scenario details. Opportunity cost is the cost of establishing a marine protected area

(MPA), measured as lost income from restricting fishing and aquaculture in the conservation area. The quantile range divides the range of possible values into unequal sized intervals so that the number of values is the same in each class.

3.3.3 Systematic conservation planning using Marxan

For the identification of priority areas in our study, we used a systematic conservation planning tool, Marxan (Ball et al. 2009). Marxan uses a simulated annealing algorithm to solve the problem of meeting biodiversity targets for the least cost. Here we set a target to protect 10% of each of 77 threatened marine species (following Mazor et al. 2013b). This target was set as a realistic, achievable target for the region, considering that so little has been done in the Mediterranean Sea (Giakoumi et al. 2012b). However, given that these species are threatened, targets should ideally be set higher. We also ran our analysis with a 30% target for each species. Our objective was to meet targets for the minimum opportunity cost. To enable comparison between our different types of cost, we did not preferentially cluster our planning units (the Boundary Length Modifier was set to 0). We performed a Marxan analysis with 1000 runs on each of the eight opportunity cost scenarios. Our resulting Marxan outputs from each scenario were compared by analyzing the selection frequency (number of runs in which a planning unit was selected among the 1000 runs) and single best solution outputs (the solution that best reaches targets and minimizes cost). For all combined scenarios (scenarios 6–9; Fig. 3.1), we ran a Spearman's rank correlation on the selection frequency outputs to test how similar the outputs were.

3.3.4 Comparing our spatial priorities with consensus conservation areas

Here, using ArcGIS (ESRI 2010), we compared our resulting priority areas, which are the first attempt at explicitly including cost at a whole-basin scale, with those that have been recently proposed as conservation consensus priority areas in the Mediterranean Sea (Micheli et al. 2013b). These consensus areas are areas where 12 proposed prioritization schemes for the Mediterranean Sea overlap (for further information, see Micheli et al. 2013b). Because these consensus areas do not aim to meet biodiversity targets or build a representative reserve network, we used spatial overlap as a means of comparison. The comparison was made by calculating the percentage of overlap of our spatial priorities (planning units that had a selection frequency > 50%) in our outputs from the most plausible combined scenarios, scenario 8 (combined sector C) and scenario 9 (combined sector D), with the consensus areas. The opportunity cost scenarios used for this comparison were chosen because they represent all sectors of marine exploitation and include the two different approaches for estimating commercial fishing.

3.4 Results

3.4.1 Comparing opportunity cost scenarios

Our results indicate that ~10% of surface area of the Mediterranean Sea is required to implement a solution that meets our 10% conservation target (Table 3.2). When we increased our target to 30% for each species, we found that ~30% of Mediterranean Sea surface area is required to be protected (Appendix S3.3: Table S3.4). Similarly, the cost also increased with a higher target for each scenario. For the four combined cost scenarios (scenarios 6–9), we can meet a 10% target for a cost less than 1% (for minimized sectors), whereas a cost of 3–6% is needed to reach a 30% target. Despite the area and cost requirements, we found that the relative changes in cost were similar for the scenarios under the different targets (Table 3.2; Appendix S3.3; Table S3.4). Due to this, the following results will discuss only the 10% target. We found that the total cost and area were the lowest when multiple marine sectors were included in the opportunity cost (scenarios 6–9; Table 3.2; Fig. 3.2). Although scenarios that included only one marine sector (scenarios 2–5) incurred higher costs and area than combined scenarios, they all performed better than using area as a cost surrogate (scenario 1). The combined sector scenarios had a total cost of 1.1–3.73%, single sector scenarios between 3.85% and 8.67%, and area of 12.25% (Table 3.2). In all cases in which opportunity cost from a particular marine sector was included in the scenario, we found that the percentage of annual income required to meet our objectives was minimized to 0–2.36%. However, in scenarios (excluding scenario 1) in which particular marine sectors were not included in the opportunity cost, it could result in costs up to 11.59% (Table 3.2). We also found that when a marine sector is minimized alone (e.g., scenario 2–5), it does not actually benefit (no substantial cost differences) any more than if it were included with other marine sectors in a combined scenario.

We found that spatial conservation priorities selected were sensitive to the different opportunity costs considered (Fig. 3.3; Appendix S3.3: Figs. S3.4). This is also due to the high flexibility of achieving our 10% conservation target in the Mediterranean Sea. In the four combined sector scenarios (scenarios 6 to 9; Fig. 3.3), we notice that commercial fishing is an important determinant of our spatial priorities due to the selection of similar priority areas for cost layers that used the same commercial fishing cost. Comparing all four combined scenarios (scenarios 6–9), we found that areas that are highly selected in all scenarios are: waters of Malta, coastal waters of western Libya, coastal waters of Egypt, waters of the Adriatic, parts of Greece's EEZ, and waters of France and Monaco. Our Spearman's correlation coefficient showed that there was some correlation between the spatial patterns of the reserve selections. Scenarios with the same commercial fishing surrogate (GFCM or SAUP) were most similar; moreover, scenario 6 was most similar to scenario 8 (*P* = 0.97), and scenario 7 was most similar to scenario 9 (*P* = 0.98). We found that a moderate correlation (*P* = 0.52; *P* = 0.53) exists between scenarios consisting of different commercial fishing cost layers (Table 3.3; scenarios 6 and 8, using GFCM, correlated with scenarios 7 and 9, using SAUP). Overall, conservation priorities were largely dominated by the commercial fishing sector (Fig. 3.3).

3.4.2 Comparing spatial priorities with consensus areas

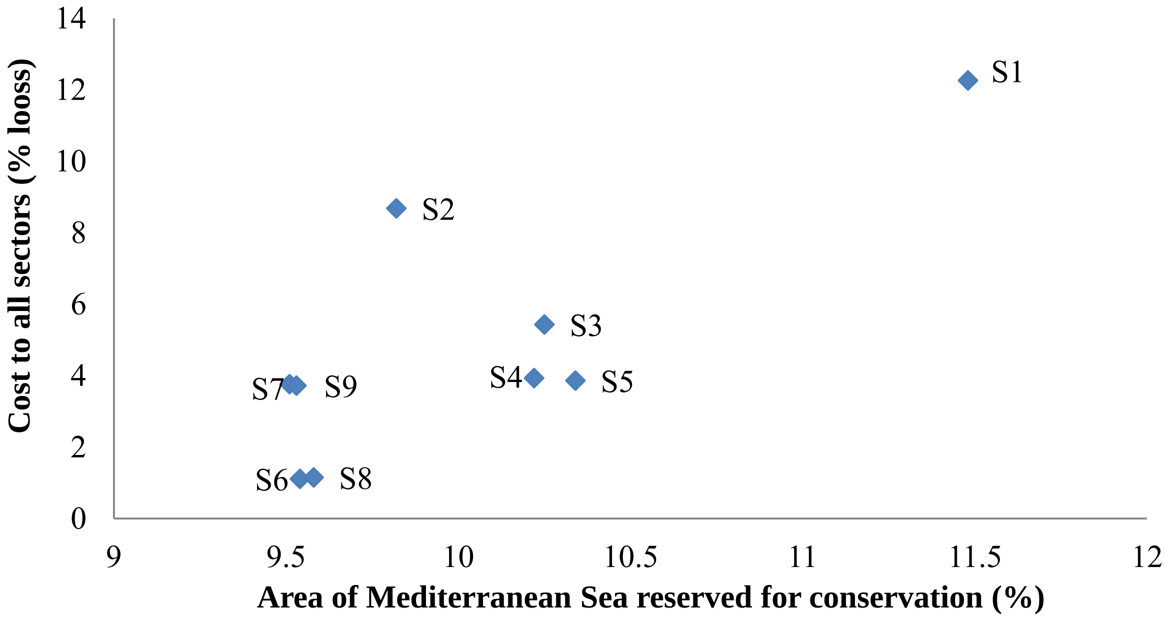
We found that some priority conservation areas identified in our study matched with areas found to be consensus areas among multiple conservation plans by Micheli et al. (2013b; Fig. 3.4). When comparing scenario 8 (combined sectors C; Fig. 3.4b) with the consensus areas (Fig. 3.4a), we found there was a 25% (37,978.44 km^2) overlap of matching priority areas, and an 18% (21,572.78 km^2) overlap of priorities with scenario 9 (combined sectors D; Fig. 3.4c). These matching priority areas include the waters of France and Monaco, parts of the Adriatic Sea, waters of Malta, and coastal areas of western Libya (Fig. 3.4). Other similarities exist, such as the selection of the Aegean Sea, although Micheli et al. (2013b) has priorities in the south of the Aegean Sea (Fig. 3.4a), whereas our results show priorities in the north of the Aegean Sea as well (Fig. 3.4b, c). However, we also identified different priority areas that were not considered priority consensus areas in Micheli et al. (2013b). In our study, we identified a large priority area along the coast of Libya and another one that extends from the Egyptian coastline toward the EEZ boarder with Greece (Fig. 3.4b, c). Other priorities in our study that were not identified as consensus areas include: parts of Algerian waters, southern Greece extending toward Egypt, and waters around Cyprus. Our resulting outputs (Fig. 3.4b, c) show more priority areas within eastern waters of the Mediterranean Sea compared with Micheli et al. (2013b), where most are predominantly in the western basin. This is probably due to greater sampling efforts, availability, and accessibility to information on biodiversity and habitats from western areas. Some consensus areas from Micheli et al. (2013b), shown in Fig. 3.4a, were not present as priority areas in our results (Fig. 3.4b, c). These areas included most of the Alboran Sea, the Ligurian Sea, and the Tunisian Plateau. The exclusion of such areas in our study is probably due to the high cost associated with these areas, because common species were used within in analysis; the biodiversity features of these areas can be represented (with a target set at 10%) in areas of lower cost.

Table 3.2. Results from Marxan best solution outputs (best solution from 1000 Marxan runs) for 9 scenarios of opportunity cost. Each scenario is compared by the percentage of cost the sector will lose from its annual revenue in order to implement the best solution and the surface area (%) the solution will take up in the Mediterranean Sea. Values in gray are marine exploitation sectors that were minimized in the scenario.

	Marine Exploitation Sectors							
Nine Scenarios of Opportunity Cost	**Commercial fishing**		**Non-commercial fishing**		**Aquaculture**	**Cost of all sectors**	**Cost of minimized marine sectors**	**Area of the Mediterranean Sea**
	GFCM	**SAUP**	**A** *(b = 0.5)*	**B** *(b = 1)*				
Scenario 1. Area as cost	11.87%	10.48%	15.88%	15.92%	7.83%	**12.25%**	**9.99% (area)**	**11.48%**
Scenario 2. Commercial fishing GFCM	1.05%	10.49%	3.41%	3.26%	5.72%	**8.67%**	**1.05%**	**9.82%**
Scenario 3. Commercial fishing SAUP	11.45%	1.89%	11.59%	11.57%	6.32%	**5.42%**	**1.89%**	**10.25%**
Scenario 4. Non-commercial fishing A* *(b = 0.5)*	1.86% (4.18%*)	2.22% (4.69%*)	1.01%	0.99%	8.19%	**2.01% (3.92%*)**	**1.01%**	**3.40% (10.22%*)**
Scenario 5. Non-commercial fishing B* *(b = 1)*	1.84% (4.14%*)	2.21% (4.67%*)	1.01%	0.96%	6.54%	**1.96% (3.85%*)**	**0.96%**	**3.44% (10.34%*)**
Scenario 6. Combined sectors A *(GFCM + Non-commercial A +Aquaculture)*	0.96%	4.99%	0.95%	2.01%	0.02%	**1.10%**	**0.89%**	**9.54%**
Scenario 7. Combined sectors B *(SAUP + Non-commercial A + Aquaculture)*	4.22%	2.36%	0.07%	0.06%	0%	**3.73%**	**0.33%**	**9.51%**
Scenario 8. Combined sectors C *(GFCM + Non-commercial B + Aquaculture)*	0.98%	5.05%	2.37%	2.27%	0.01%	**1.14%**	**0.92%**	**9.58%**
Scenario 9. Combined sectors D *(SAUP + Non-commercial B + Aquaculture)*	8.36%	2.35%	0.12%	0.11%	0%	**3.71%**	**0.46%**	**9.53%**

*8,964 planning units were used for these cost layers

a)



b)

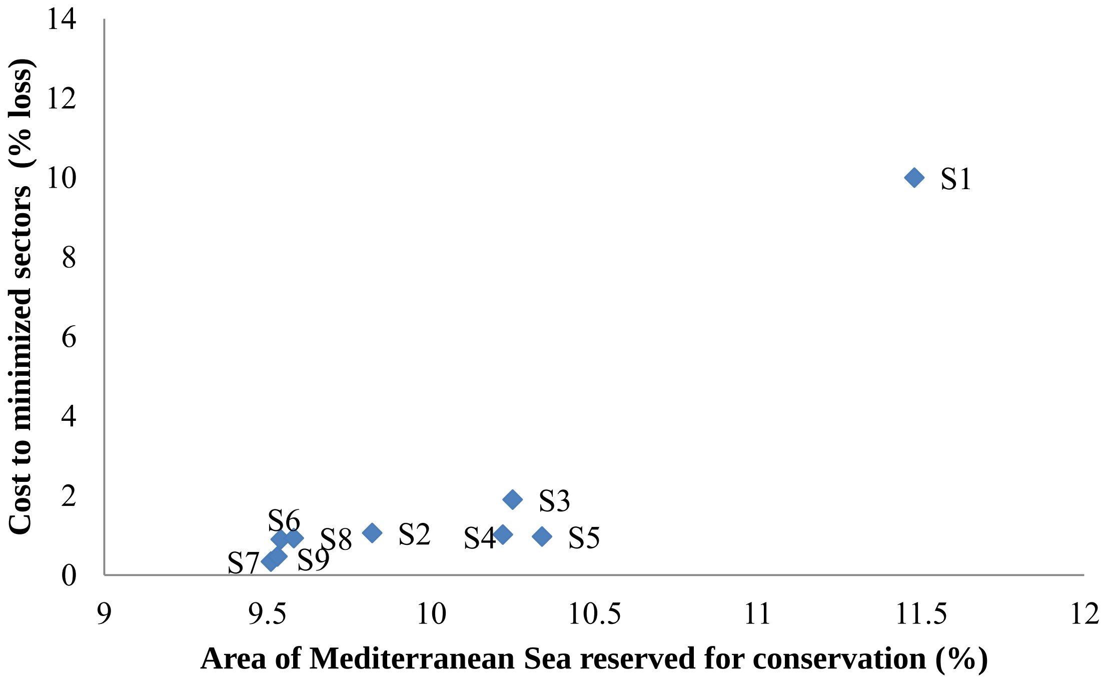


Figure 3.2. These graphs compare the cost (percentage of annual loss to marine sectors) and area (percentage of Mediterranean Sea surface area needed to be reserved) to reach our targets for each of our nine scenarios (S1-S9; see Table 3.1 for a full description of each scenario) of opportunity cost: (a) cost to all sectors vs. area reserved, and (b) cost to minimized sectors vs. area reserved.

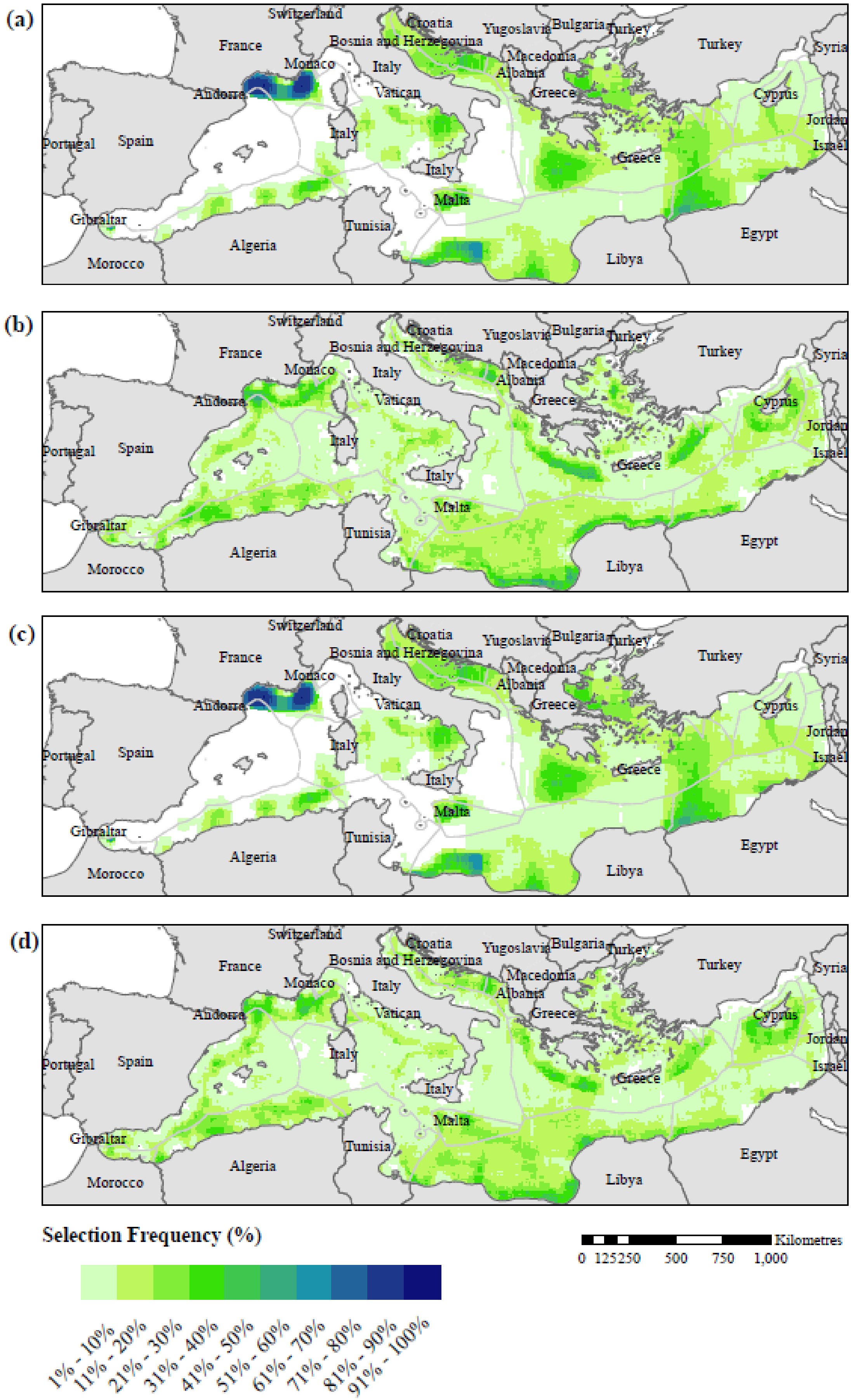


Figure 3.3. Selection frequency of four combined fishing layers (combined costs from commercial, non-commercial, and aquaculture sectors) for: (a) combined sectors A (scenario 6), (b) combined sectors B (scenario 7), (c) combined sectors C (scenario 8), (d) combined sectors D (scenario 9).

Selection frequency is the percentage of times that an area is selected, from 1000 Marxan runs, as a priority area for conservation as a Marine Protected Area (MPA).

Table 3.3. Spearman rank correlation coefficient of the selection frequency output for each of the combined scenarios. All scenarios show significant $P < 0.001$.

Scenarios of opportunity cost	**Scenario 6**	**Scenario 7**	**Scenario 8**	**Scenario 9**
Scenario 6. Combined sectors A *(GFCM + Non-commercial A +Aquaculture)*		0.53	0.97	0.53
Scenario 7. Combined sectors B *(SAUP + Non-commercial A + Aquaculture)*	0.53		0.52	0.98
Scenario 8. Combined sectors C *(GFCM + Non-commercial B + Aquaculture)*	0.97	0.52		0.52
Scenario 9. Combined sectors D *(SAUP + Non-commercial B + Aquaculture)*	0.53	0.98	0.52	

3.5 Discussion

Here, we show that including the cost of implementing marine conservation in the form of opportunity cost, especially within a multinational setting, can greatly influence the selection of priority conservation areas. By using nine different opportunity cost scenarios, we demonstrated how the incorporation of different cost layers can result in spatial conservation plans that have different priority areas (Fig. 3.3) and different cost and area requirements (Table 3.2; Fig. 3.2). The spatial priorities identified in this study met conservation targets while minimizing the opportunity cost for multiple exploitation sectors of marine resources (Fig. 3.3). In addition, areas considered spatial priorities (e.g., EBSAS [ecologically or biologically significant marine areas]) by other studies, e.g., the Alboran Sea and the Ligurian Sea, where the Pelagos Sanctuary for marine mammals is located (as identified by WWF [World Wildlife Fund], Greenpeace, and ACCOBAMS [Agreement on the Conservation of Cetaceans of the Black and Mediterranean Seas]; see Micheli et al. 2013b) were actually found to be inefficient areas for conservation due to their high cost. Providing conservation plans that fit within economic constraints and budgets is critical for achieving viable conservation outcomes (Naidoo et al. 2006).

We achieved greater conservation efficiency in identifying priority areas for the establishment of MPAs when combining the opportunity cost from different marine sectors (Fig. 3.2). Moreover, the percentage of cost to the marine sectors and the spatial requirements for an MPA network were reduced. By only accounting for commercial fishing opportunity cost, our results would produce less efficient solutions than combing this cost with opportunity cost for non-commercial fishing and aquaculture. Moreover, planning for a single sector would produce higher costs for other users (Table 3.2). In the Mediterranean Sea, which is exploited by a composite of marine users from developing and developed nations with diverse socioeconomic, political, and cultural characteristics, it is important that we set multiple objectives when planning conservation to reflect this diversity of marine users. For example, the impact of recreational fishing is often overlooked compared to its counterpart, commercial fishing (Cooke & Cowx 2006). Thus, only considering the cost of commercial fishing when planning conservation may cause our resulting spatial priorities to diverge from ones that are realistically achievable. Not only are there quantifiable benefits (cost and area) but also combining costs from various socioeconomic interests (marine sectors) can build a greater understanding of feasible spatial options that serve multiple objectives rather than encountering future conflicting interests (Cameron et al. 2008; Klein et al. 2008a). Providing options that minimize impacts on multiple marine users is pivotal for convincing stakeholders to cooperate in marine conservation and MPA implementation.

Area is a poor cost surrogate in marine systems. In conservation planning, area is sometimes used to represent cost in spatial reserve design (Naidoo et al. 2006). However, monetary costs are considered preferable for decision makers and planners (Naidoo et al. 2006; Carwardine et al. 2008). Nevertheless, in some terrestrial cases it has been suggested that area sometimes may be just as effective as a cost surrogate, or more effective than a poor cost surrogate (Adams et al. 2010). In marine systems, this is not the case. In our study, we see that area as a cost performs poorly at delivering outcomes that minimize the cost for multiple marine sectors (Table 3.2; Fig. 3.2). Not only are there less efficient outcomes for marine sectors, but also conservation priorities can be misleading (Ban & Klein 2009). Coastal areas are highly utilized by humans; therefore we know that opportunity cost will be much greater along the coast. This is especially the case in the Mediterranean, where fishing practices are mostly confined to a narrow continental shelf (Papaconstantinou & Farrugio 2000). Similarly, in the Mediterranean Sea the high heterogeneity of wealth and culture between countries means that opportunity costs are far from uniform, which is often considered the case when using area for cost. Although we acknowledge that an inaccurate cost layer will bias results, we emphasize the need for better cost surrogates and approaches for their development and evaluation in the marine realm.

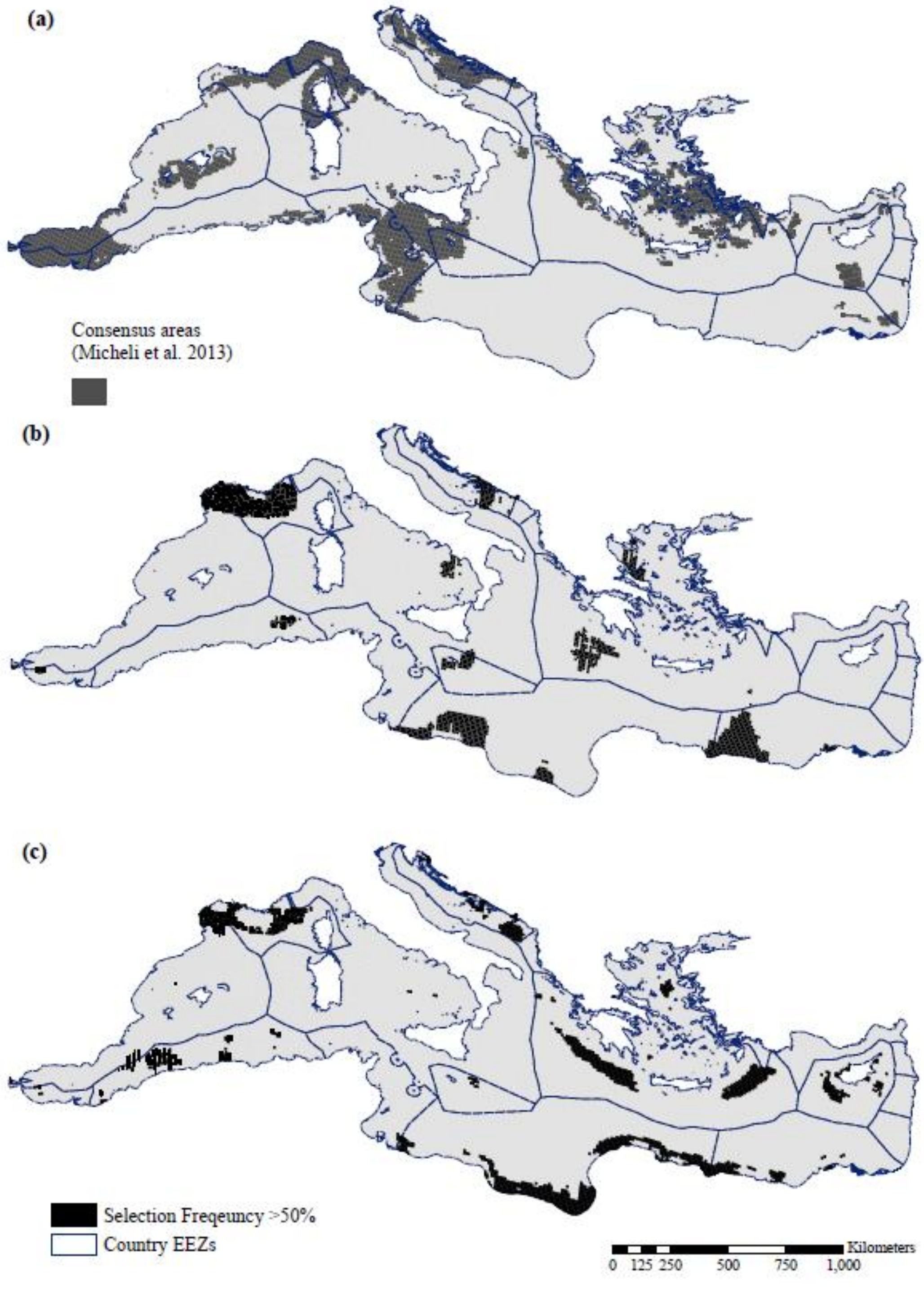


Figure 3.4. (a) Consensus areas of prioritization schemes that do not consider costs (dark gray) as presented in Micheli et al. (2013b), compared with our resulting priority areas (areas that are selected more than 50% of the time; in black) for (b) combined sectors C (scenario 8), commercial fishing GFCM + non-commercial B (b = 1.0) + aquaculture and (c) combined sectors D (scenario 9), commercial fishing SAUP + non-commercial B (b = 1.0) + aquaculture. Country EEZ is the Exclusive Economic Zone of each country, with boundaries shown by thin lines in all three panels. SAUP is the Sea Around Us Project (2011); GFCM is the General Fisheries Commission for the Mediterranean (FAO 2011). Refer to Table 3.1 for full scenario details.

Currently, Mediterranean countries face major economic and political challenges. Cost is an important component of a conservation plan's feasibility, but there are also other issues that determine feasibility, e.g., law enforcement in territorial waters where priority areas have been identified. In the Mediterranean Sea, some countries in the northern part of the basin are on the verge of bankruptcy and those in the east and south are experiencing societal instability and shifts in political regimes (Gaiser & Hribar 2012). As a result, resources for conservation are more limited than ever and wise decisions should be made for their allocation. We propose that future conservation plans for the Mediterranean Sea apply systematic plans where costs and benefits can be explicitly estimated and, hence, can appropriately guide decision-making. Moreover, spatial priorities should be coupled with specific conservation actions and return on investment should be estimated to facilitate informed decision-making.

The surrogates provided in this study indicate the lack of knowledge and comparable data we have when planning large-scale marine areas that span multiple countries. In areas that encompass several countries with great economic, political, and cultural heterogeneity, it becomes difficult to find data that are in a compatible format, are spatially refined, temporally comparable or that even exist. We have attempted to keep data temporally consistent where possible, and to account for the variance between countries using PPP adjustment in our cost metrics. However, the ability to validate our surrogates is impossible with the lack of detailed information on commercial and non-commercial fishing in the Mediterranean Sea. Although it may be argued that our results are based on coarse surrogate data, previous studies in terrestrial landscapes (Ando et al. 1998; Moore et al. 2004) and small-scale marine settings (Stewart & Possingham 2005) show that the use of opportunity cost data can substantially improve efficiency in selecting priority conservation areas beyond a study that used area as a cost factor or completely ignored cost. Distance from port or coast is a representative measure of fishing pressure according to numerous studies, especially for small-scale fishing (Cabrera & Omar 1997; Caddy & Carocci 1999; Gelchu & Pauly 2007; Stelzenmueller et al. 2008). Moreover, some studies have applied it to prioritization schemes (Sala et al. 2002; Stewart et al. 2010; Giakoumi et al. 2011) and it has been proven to perform well in comparison to other cost surrogates, e.g., population pressure (Weeks et al. 2010a). However, we acknowledge that large-scale fishing, mainly industrial fishing, is driven by specific features, e.g., the migratory paths of commercial pelagic species. The availability of data on Vessel Monitoring Systems applied in large-scale fisheries in most Mediterranean countries would improve the estimation of the spatial distribution of such commercial fisheries (Maiorano et al. 2009; Giakoumi et al. 2012a). We propose that future studies address these shortages of data in the Mediterranean

Sea, and as information and data become readily available our priorities can be validated and appropriately adjusted.

Our approach for large-scale conservation planning provides a platform for future expansion. This includes other types of cost involved with implementing an MPA network. These costs specifically include: monitoring cost, transaction cost, and management cost (Naidoo et al. 2006). Future considerations should include issues such as illegal fishing, political stability, variation in law enforcement among countries, and the ability for countries to collaborate (Levin et al. 2013). Our study used coarse species distribution data from the IUCN (2012); however, building a better database of species and habitat distribution for the Mediterranean Sea, which is consistent between countries and at a finer spatial resolution, will help to better determine spatial priorities that reach conservation targets.

This work contributes and builds upon a growing body of literature (see Naidoo et al. 2006; Bode et al. 2008; Ban et al. 2009) that demonstrates the benefits of including cost when planning conservation. Moreover, our findings support evidence from previous studies showing that the identification of priority areas is more sensitive to the inclusion of cost data than biodiversity data, highlighting the necessity to consider both ecological and economic data in prioritization schemes (Bode et al. 2008). We demonstrated that priority areas for conservation can be selected to be spatially compatible with multiple sectors of marine users, even in a data-poor system. Our approach is also relevant and applicable to other marine regions that are shared between various geographic jurisdictions such as states, territories, or countries (e.g., the Black Sea, the Gulf of Mexico, the Baltic Sea, the Caribbean Sea). Overall, the inclusion of cost when setting spatial conservation priorities can help to provide better investment decisions and advance conservation efforts in a timely and efficient manner.

3.6 Supplementary material

Appendix S3.1. Detailed information of threatened species used in this study.

Table S3.1. List of species used in this study, taxa group, scientific name, common name and IUCN red list status in both the Mediterranean and the Globe (IUCN 2012).

Taxa group	Scientific name	Common name	IUCN Red List status	
			Mediterranean Red List	**Global Red List**
Marine mammals	*Balaenoptera borealis*	Sei whale	EN	EN
	Balaenoptera physalus	Fin whale	EN	EN
	Delphinus delphis (Mediterranean subpopulation)	Short-beaked common dolphin	EN	
	Eubalaena glacialis (vagrant in the Mediterranean Sea)	North Atlantic right whale	EN	EN
	Monachus monachus	Monk Seal	CR	CR
	Physeter macrocephalus	Sperm whale	VU	VU
	Tursiops truncatus ssp. ponticus	Common bottlenose dolphin	EN	
Native fishes (jawless and bony fishes)	*Dentex dentex*	Common *dentex*	VU	
	Dicentrarchus labrax	European seabass	NT	
	Epinephelus aeneus	White grouper	NT	NT
	Epinephelus marginatus	Dusky Grouper	EN	EN
	Hippocampus guttulatus	Long-snouted seahorse	NT	
	Hippocampus hippocampus	Short-snouted seahorse	NT	
	Labrus viridis	Green wrasse	VU	VU
	Merluccius merluccius	European hake	VU	
	Opeatogenys gracilis		VU*	VU*
	Platichthys flesus	European flounder	NT	
	Pleuronectes platessa	European plaice	NT	
	Pomatoschistus microps	Common goby	CR	
	Pomatoschistus minutus	Sand goby	VU	
	Pomatoschistus tortonesei		EN*	EN*
	Psetta maxima	Turbot	NT	
	Sciaena umbra	Brown meagre	VU	
	Scomber colias	Atlantic chub mackerel	NT	
	Syngnathus acus	Greater pipefish	NT	
	Syngnathus taenionotus	Dark-flank pipefish	EN*	EN*

	Syngnathus tenuirostris	Narrow-snouted pipefish	NT*	NT*
	Syngnathus typhle	Broad-nosed pipefish	NT	
	Thunnus thynnus	Atlantic bluefin tuna	EN	
	Umbrina cirrosa	Shi drum	VU	
	Xiphias gladius	Swordfish	NT	
Seabirds	*Larus audouinii*	Audouin's Gull	NT	NT
	Pelecanus crispus	Pelican	VU	VU
	Puffinus griseus	Sooty Shearwater	NT	NT
	Puffinus mauretanicus	Balearic Shearwate	CR	CR
	Puffinus yelkouan	Yelkouan Shearwater	NT*	NT*
Sea turtles	*Caretta caretta*	Loggerhead sea turtle	EN	EN
	Chelonia mydas	*Green sea turtle*	*EN*	*EN*
Sharks and rays (cartilaginous fishes)	*Alopias vulpinus*	Long-tailed, common thresher shark	VU	VU
	Carcharhinus plumbeus	Sandbar shark	EN	VU
	Carcharias taurus	Sand tiger shark	CR	VU
	Carcharodon carcharias	Great white shark	EN	VU
	Centrophorus granulosus	Gulper shark	VU	VU
	Cetorhinus maximus	Basking shark	VU	VU
	Dasyatis centroura	Roughtail stingray	NT	
	Dasyatis pastinaca	Common stingray	NT	
	Dipturus batis	Common skate	CR	CR
	Dipturus oxyrhynchus	Longnosed skate	NT	NT
	Gymnura altavela	Spiny butterfly ray	CR	VU
	Heptranchias perlo	Harpnose sevengill shark	VU	NT
	Hexanchus griseus	Bluntnose sixgill shark	VU	NT
	Isurus oxyrinchus	Ahortfin mako shark	CR	VU
	Lamna nasus	Porbeagle shark	CR	VU
	Leucoraja circularis	Sandy skate	CR	VU
	Leucoraja fullonica	Shagreen ray	NT	NT
	Leucoraja melitensis	Maltese skate	CR*	CR*
	Leucoraja naevus	Cuckoo ray	NT	
	Mobula mobular	Giant devil ray	EN	EN
	Mustelus asterias	Starry smooth-hound	EN	
	Mustelus mustelus	Common smooth-hound	EN	VU
	Myliobatis aquila	Common eagle ray	NT	
	Odontaspis ferox	Smalltooth sand	VU	VU

		tiger, bumpytail ragged-tooth		
	Oxynotus centrina	Angular roughshark	CR	VU
	Pteroplatytrygon violacea	Pelagic stingray	NT	LC
	Pristis pectinata	Small-tooth sawfish	CR	CR
	Pristis pristis	Common sawfish	CR	CR
	Raja clavata	Thornback ray	NT	NT
	Raja undulata	Undulate Ray	EN	EN
	Rhinobatos cemiculus	Black-chin guitarfish	EN	EN
	Rhinobatos rhinobatos	Common guitarfish	EN	EN
	Rostroraja alba	Bottlenose Skate, Spearnose Skate, White Skate	CR	EN
	Scyliorhinus stellaris	Nursehound	NT	NT
	Sphyrna zygaena	Smooth hammerhead	VU	VU
	Squalus acanthias	Spiny dogfish, spurdog, mud shark, or piked dogfish	EN	VU
	Squatina aculeata	Sawback Angelsharks	CR	CR
	Squatina oculata	Smoothback angelshark	CR	CR
	Squatina squatina	Angelshark	CR	CR

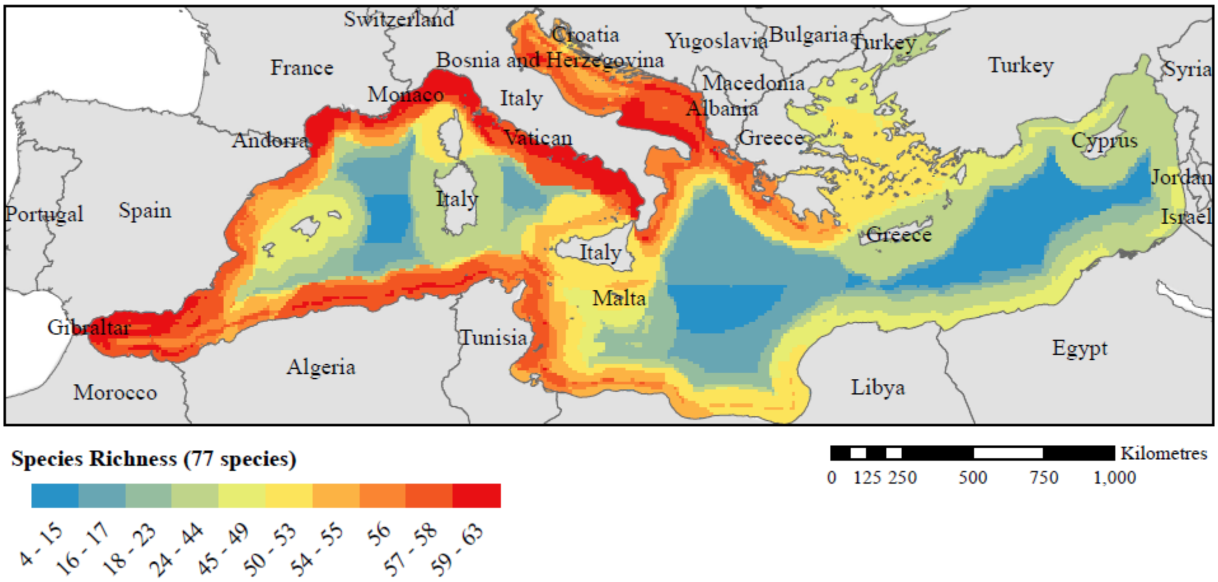


Figure S3.1. Species richness of 77 threatened marine species (IUCN 2012) in the Mediterranean Sea.

Appendix S3.2. Detailed descriptions of input data and resulting cost layers.

Table S3.2. Categories used to separate small-scale and large-scale commercial fishing. Small scale (artisanal fleet) refers to categories A, B, C and M. Large-scale (semi-industrial and industrial fleet) refers to all other categories (within FAO 2011).

Category	Description
A	small scale vessels without engine (<12 m)
B	small scale vessels with engine (<6 m)
C	small scale vessels with engine (6-12 m)
D	Trawlers (<12 m)
E	Trawlers (12 -24 m)
F	Trawlers (>24 m)
G	Purse Seiners (6-12 m)
H	Purse Seiners (>12m)
I	Long liners (>6 meters)
J	Pelagic Trawlers (>6 meters)
K	Tuna Seiners (<12 meters)
L	Dredgers (>6 m)
M	Polyvalent vessels (>12 m)

Table S3.3. Here we compare the total cost ($US million) retrieved from aquaculture as calculated in our methods for each country in 2006 with values reported by FAO in 2008 (Sacchi 2011). Differences could be due to the fact that aquaculture of shell fish and tuna species were not included in the 2006 data and the expansion of aquaculture farm within a two year period. Conversion from US Dollars to Euros was via the annual average exchange rate for 2006 as reported by the International Monetary Fund (http://www.imf.org/).

Country	Calculated Cost (€ million) 2006	Cost conversion to $US million (1€ = US$1.26)	Cost (US$ million) 2008
Croatia	15.7	19.8	29.8
Cyprus	10.7	13.5	38.4
France	23.5	29.6	102.0
Greece	355.3	447.7	522.3
Israel	11.6	14.6	16.6
Italy	57.5	72.5	307.2
Libya	1.1	1.4	1.1
Malta	4.5	5.7	9.9
Slovenia	0.14	0.2	1.3
Spain	94	118.4	149
Tunisia	5.1	6.4	18.9
Turkey	313	394.38	449.4

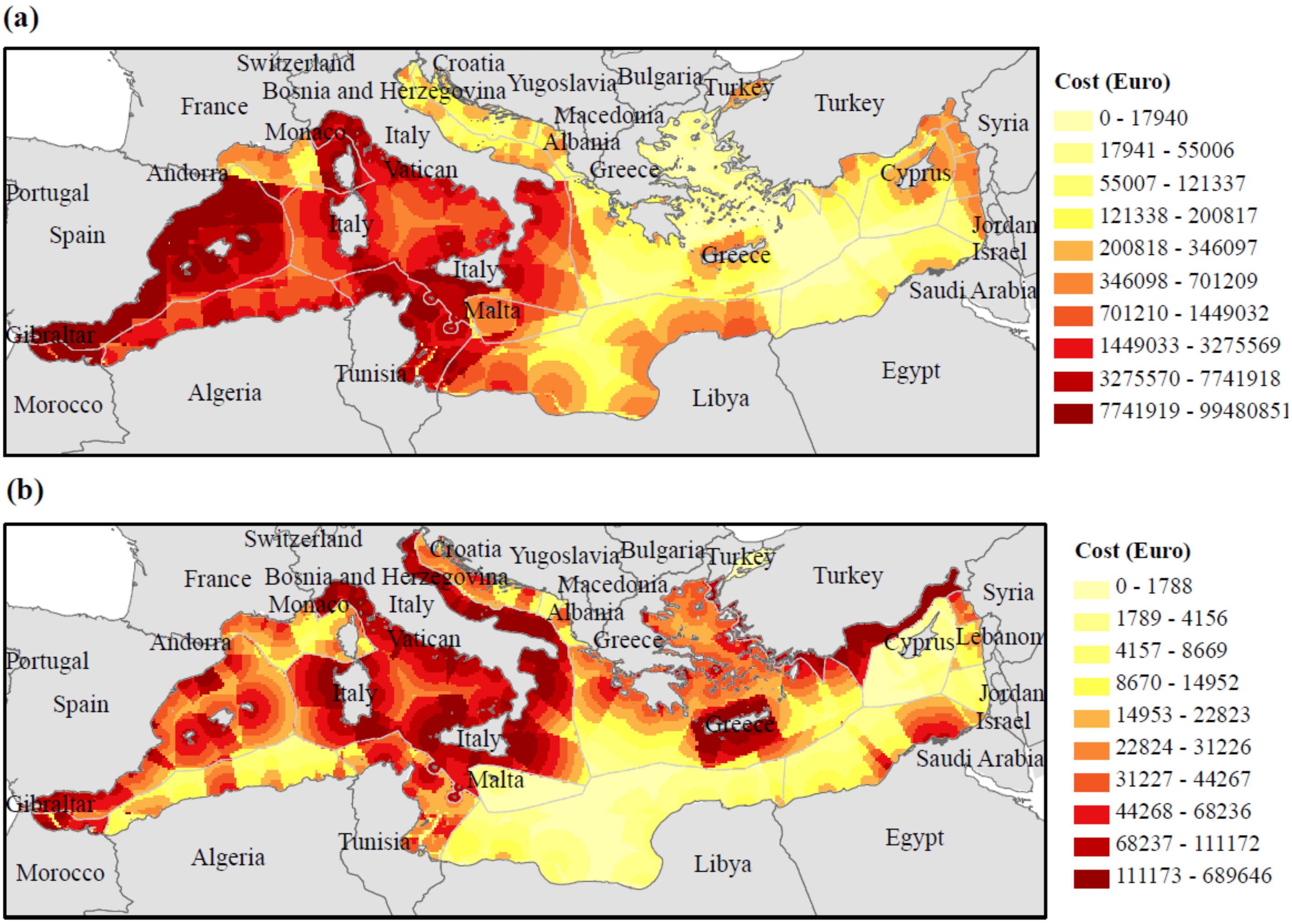
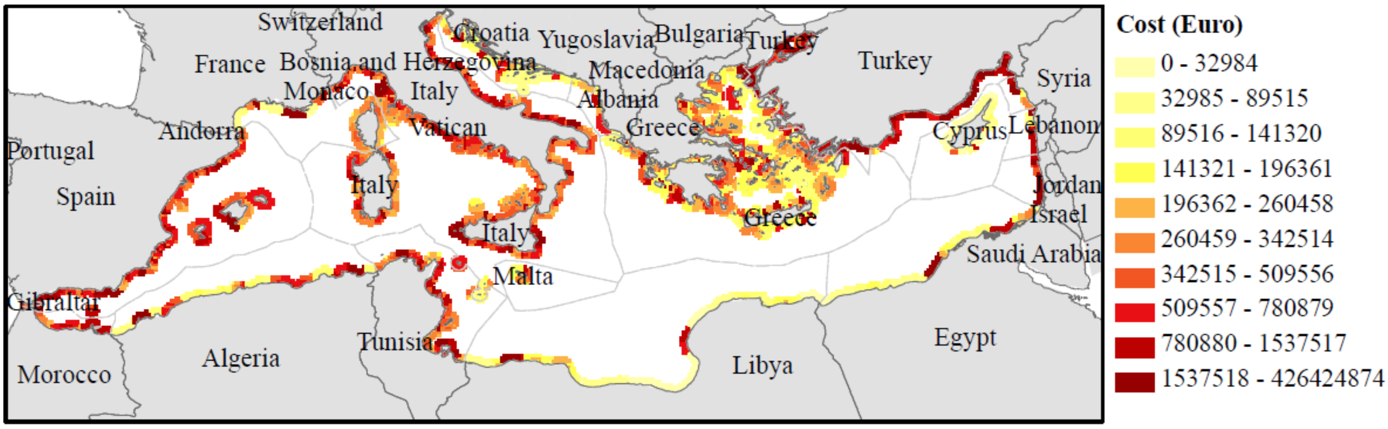


Figure S3.2. Cost layers (opportunity cost in €/10 km^2 displayed by a quantile range), of a) Commercial fishing GFCM and b) Commercial fishing SAUP.

(a)



(b)

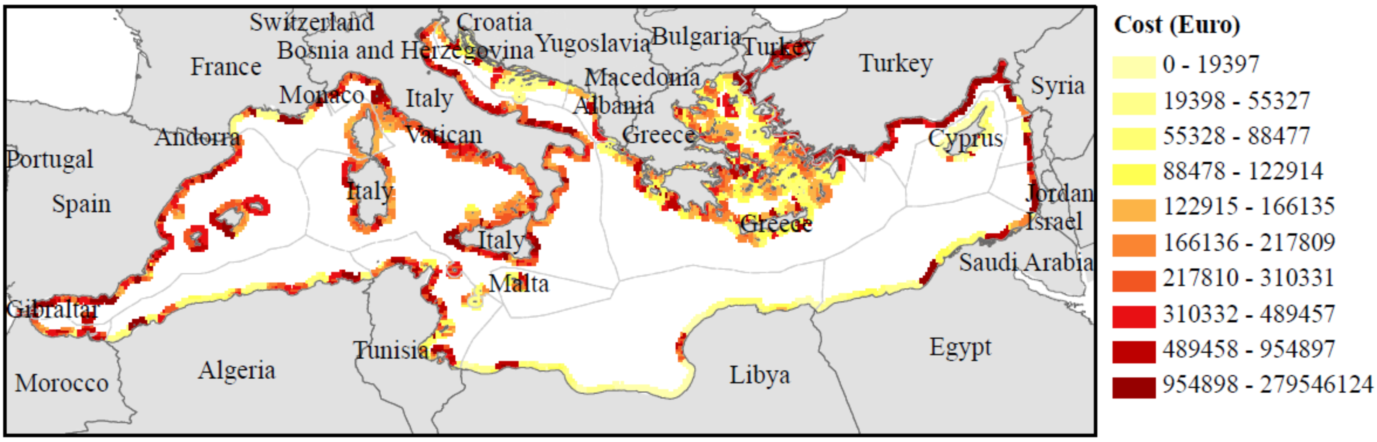


Figure S3.3. Cost layers (opportunity cost in €/10 km^2 displayed by a quantile range) of a) Non-commercial fishing A (with constant b = 0.5) and b) Non-commercial fishing B (with constant b = 1).

Appendix S3.3. Results of 30% target and selection frequency outputs from Marxan analysis.

Table S3.4. This table displays the cost (annual loss) to each marine sectors and the area of the sea required when implementing a reserve system for the whole Mediterranean Sea. We compare between four scenarios of opportunity cost. In our analysis a 30% target we set for each threatened species (77 species; see Table S3.1; Fig. S3.1). These results compare with Table 3.2 which uses a target of 10%.

	Marine Exploitation Sectors							
	Commercial fishing		**Non-commercial fishing**		**Aquaculture**	**Cost of all sectors**	**Cost of minimised marine sectors**	**Area of the Mediterranean Sea**
Scenarios of Opportunity Cost	**GFCM**	***SAUP***	***A*** *(b = 0.5)*	***B*** *(b = 1)*				
Scenario 1. Area as cost	30.58%	30.74%	33.61%	33.77%	33.76%	**31.03%**	**20.99% (area)**	**30.90%**
Scenario 2. Commercial fishing GFCM	6.57%	20.07%	12.91%	12.74%	21.50%	**7.89%**	**6.57%**	**29.32%**
Scenario 3. Commercial fishing SAUP	17.82%	9.68%	30.63%	30.65%	33.98%	**19.92%**	**9.68%**	**29.56%**
Scenario 4. Non-commercial fishing A* (*b* = 0.5)	9.07% (20.37%)	9.56% (20.25%)	4.86%	4.76%	3.58%	**8.49% (16.23%)**	**4.86%**	**9.56% (28.70%)**
Scenario 5. Non-commercial fishing B* (*b* = 1)	9.11% (20.47%)	8.89% (18.82%)	4.85%	4.61%	2.81%	**8.51% (16.26%)**	**4.61%**	**9.36% (28.12%)**
Scenario 6. Combined sectors A (GFCM + Non-commercial A +Aquaculture)	6.23%	20.01%	10.89%	10.63%	22.47%	**7.12%**	**6.18%**	**28.68%**
Scenario 7. Combined sectors B (SAUP + Non-commercial A + Aquaculture)	18.75%	12.45%	1.50%	1.53%	34.81%	***16.28%***	**2.68%**	**28.61%**
Scenario 8. Combined sectors C (GFCM + Non-commercial B + Aquaculture)	6.21%	19.75%	12.12%	11.77%	30.43%	***7.23%***	**6.20%**	**28.82%**
Scenario 9. *Combined sectors D (SAUP + Non-commercial B + Aquaculture)*	18.07%	11.97%	1.86%	1.75%	22.07%	***15.74%***	**3.22%**	**28.71%**

*Grey shaded cells means that cost was minimised in the analysis and values in brackets used 8,964 planning units for calculations.

(a)

(b)

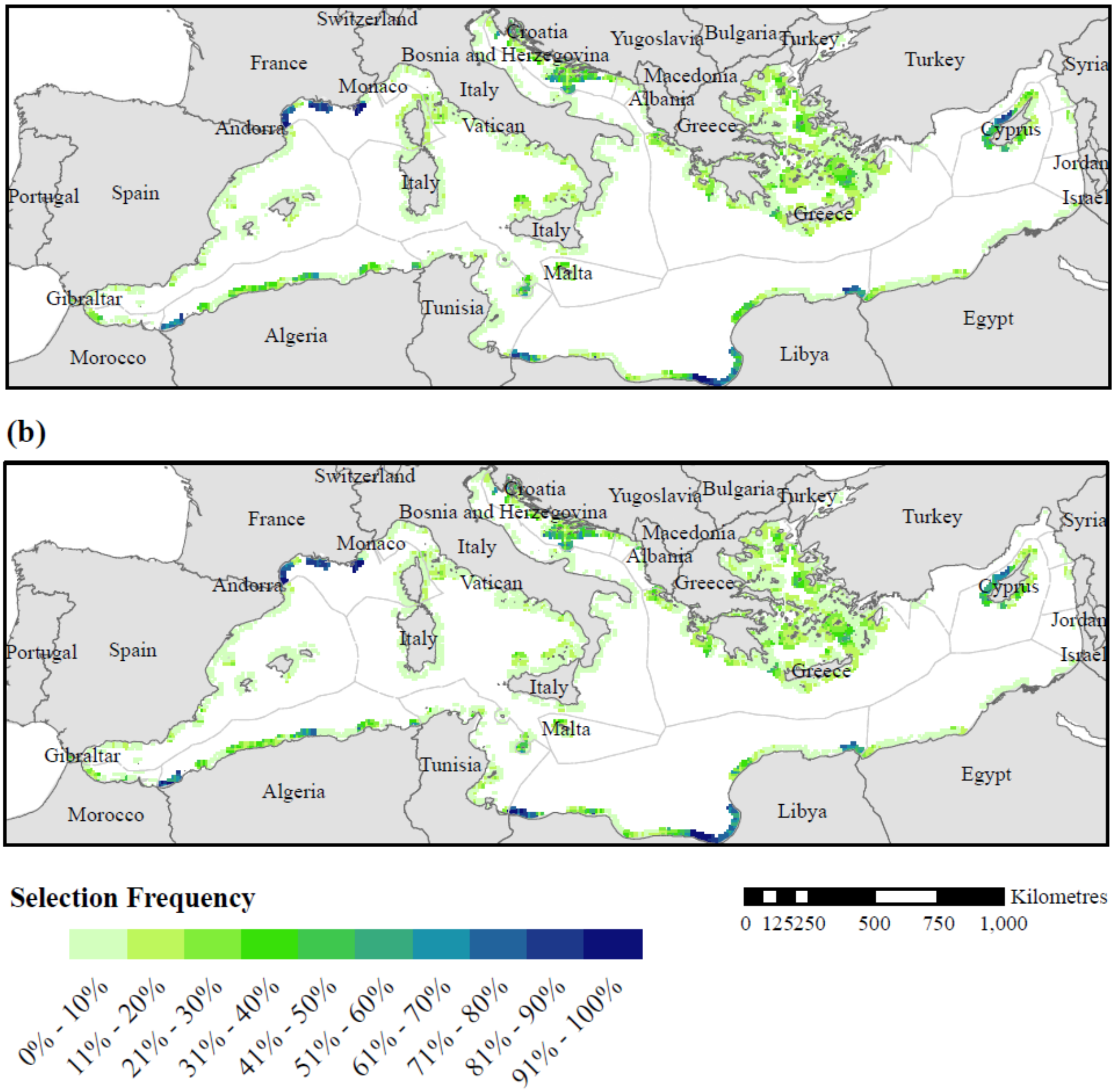


Figure S3.4. Non-commercial fishing (subsistence and recreational fishing) selection frequency (from 1000 Marxan runs) a) Non-commercial fishing A (with constant *b* = 0.5) and Non-commercial fishing B (with constant *b* = 1).

Chapter 4

Incorporating breeding, feeding and migration information for prioritising sea turtle conservation

Loggerhead sea turtle in the Eastern Mediterranean. Photo credit: T.Mazor

4.1 Abstract

Sea turtles are globally threatened, yet, large-scale conservation plans that explicitly incorporate their entire migratory life cycle are lacking. Conserving mobile marine species that use both the land and sea is challenging given the large distances they travel across international borders and between different habitats. Given our knowledge gap in sea turtle movement patterns, no attempts have been made to investigate the potential value that existing sea turtle information can provide for conservation, or how much of this information is needed for robust conservation decision making. Here we aim to incorporate breeding, feeding and migration information to prioritise loggerhead sea turtle *Caretta caretta* conservation in the Mediterranean Sea. We examine the value of available spatial information using the decision support tool Marxan, comparing four prioritisation approaches that incorporate increasing amounts of information: 1) the broad species distribution range (representative of IUCN data), 2) multiple habitat types that include foraging, nesting and inter-nesting habitats, 3) mark-recapture migration information, and, 4) tracking data. We discovered that turtle conservation priorities are sensitive to the inclusion of sea turtle migration information, and even a small number of tracks can substantially help capture migratory links. Our results suggest that in order to convey sea turtle habitat connectivity in conservation plans, efforts should focus on collecting a heterogeneous sample of tracking data over quantity. Synthesising our results, we identify for the first time, priority conservation regions for loggerhead sea turtles at the Mediterranean Sea scale across coastal and marine habitats while minimising opportunity cost. These high conservation value hotspots include: the Adriatic Sea, coastal sections of the Levantine Sea, coastal areas of southern Turkey and southern Greece, and coastal regions around the Gulf of Sidra, Libya. Our findings underpin the importance of cross-country collaboration to protect far-ranging sea turtle species, and the value of tracking data to represent species migration in spatial conservation plans.

4.2 Introduction

Sea turtles are a globally threatened species group, with five of seven species listed as Endangered or Critically Endangered by the IUCN (IUCN 2013b). Migration is essential for the persistence of marine turtles, which rely on movement between nesting and feeding areas (Miller 1997). The vast distance (thousands of kilometres) these species travel across international borders and between land and sea habitats makes them highly vulnerable to an array of anthropogenic threats (Godley et al. 2002; Plotkin 2003; Witt et al. 2008; Shillinger et al. 2010; Maxwell et al. 2011). These threats include, disturbance to nesting beaches from coastal development (Margaritoulis 2005), turtle egg harvesting, incidental catch in fishing gear, collision with boats, and the digestion of plastic material, amongst others (Casale & Margaritoulis 2010). Contributing to vulnerability of marine turtles is their long life spans, reproductive age (e.g., loggerheads ~ 25-30 years old; Heppell et al. 2005; Casale et al. 2011; Avens & Snover 2013) and non-annual nesting patterns (usually nesting every 2-4 years; Broderick et al. 2003). Given the irrefutable need for sea turtle protection and conservation, large-scale conservation plans that explicitly incorporate their entire life cycle and migratory behaviours are scarce.

Previously, sea turtle conservation efforts have focused on protecting nesting sites (Witherington & Martin 1996; Garcıa et al. 2003; Casale & Margaritoulis 2010). The central aim of these recovery efforts has been to protect eggs, emerging hatchlings and breeding females (Dalzell et al. 2011). Presumably because nesting beaches are the best studied sea turtle habitats and are easier to access and manage. However the limitations of an approach that focuses on a sub-set of the life-history of a species has been recognised as sea turtle populations have continued to decline (Spotila et al. 2000; Witherington et al. 2009; Dalzell et al. 2011). Population models indicate that conserving sea turtle nesting habitats alone without considering other key habitats is insufficient for species recovery (Heppell et al. 1996; Spotila et al. 2000; Lazar et al. 2004). Currently, there are very limited efforts to conserve sea turtles within marine ecosystems (e.g., turtle exclusion devices, Hamann et al. 2010). The absence of offshore conservation efforts is particularly concerning given that sea turtles spend most of their life at sea (Chan et al. 1988; Spotila et al. 2000; Casale et al. 2004; Lewison et al. 2007). Thus, conservation planning for sea turtles needs to explicitly protect all the life-stages and the habitat those life-stages depend on (i.e. both terrestrial and marine habitats). One of the major impediments for minimising mortality in the sea is that information on the offshore distribution and movements of sea turtles is limited (Luschi et al. 2003; James et al. 2005; Casale et al. 2007a; Hamann et al. 2010).

Various methods have been trialled to further understand sea turtle movement in offshore habitats. Since the 1950s, the most common method has been mark-recapture approaches, where tags are affixed to sea turtles at nesting sites and their location of recapture is documented (Carr & Giovannoli 1957; Hendrickson 1958; Harrisson 1959; Caldwell et al. 1962). Recaptured sea turtles have given initial insight into our understanding of sea turtle migratory pathways and movements (recaptures at sea; Limpus et al. 1992; Casale et al. 2007b; Casale et al. 2013), but have most significantly contributed to our knowledge of nesting populations and growth rates (recaptures at the same nesting beaches; Dutton et al. 2005; Monk et al. 2010). Mark-recapture approaches have been refined over the years via the application of a variety of tagging types (e.g., flipper tags; Passive Integrated Transponder (PIT) tags, genetic tags, living tags; sea turtle.org 2013), nevertheless, this approach remains labor-intensive (Stewart et al. 2013), characterised by low recapture rates (Wyneken et al. 2013), so knowledge accumulates slowly (Godley et al. 2008). For acquiring movement information this approach is limited, providing release and re-encounter locations with no insight of the route taken by an individual turtle (Casale et al. 2007b).

In recent decades, with the expansion of telemetry systems such as radio trackers, satellite transmitters and GPS loggers, sea turtle tracking programs have proliferated (Godley et al. 2008). These technologies actively improve our understanding of sea turtle migration pathways at sea (Hughes et al. 1998; Schofield et al. 2007; Bentivegna 2002; Griffin et al. 2013). While there is a strong global emphasis on these technologies to improve our understanding of sea turtles distribution, physiology and behaviour (e.g., Hochscheid et al. 2007; McCarthy et al. 2010), there is comparatively less attention paid to how this knowledge can improve conservation and identify priority conservation areas.

Currently, we have no knowledge of the quantity or type of data needed to adequately capture sea turtle movements in conservation plans for generating robust spatially explicitly priorities. Sea turtle tagging and telemetry programs are rarely explicitly shaped by conservation planning objectives, and their execution is logistically difficult and expensive (satellite transmitters range from US$2000-5000 each; Godley et al. 2008; seaturtle.org 2013). Such information often remains confined within sea turtle behavior and ecological literature without further application towards broad-scale conservation action (Godley et al. 2008). Furthermore, conservation plans are being made for mobile species such as sea turtles often without considering the potential input that migration information could contribute (Martin et al. 2007; Runge et al. 2014). Presently, no attempts have been made to investigate the potential value that sea turtle migration information may provide to enhance spatial conservation planning in a Marxan analysis.

Here we aim to improve sea turtle protection by incorporating breeding, feeding and migration information into conservation prioritisation. We aim to present a broad starting point for large-scale conservation planning of Endangered loggerhead sea turtles *Caretta caretta* in the Mediterranean Sea (Marine Turtle Specialist Group 1996). The Mediterranean Sea provides an excellent system to explore, as it is unique semi-enclosed sea, with a distinct loggerhead sea turtle population segment (Casale & Margaritoulis 2010), and substantial amount of sea turtle research (Margaritoulis et al. 2003). Given current data limitations, we examine the value of available spatial information (migration and habitat information) for identifying the location of sea turtle priority areas. We synthesise our findings to identify important conservation priority areas for loggerhead sea turtles at a whole Mediterranean scale while minimising lost fishing profits and management cost from protected areas. Such spatial prioritisations could greatly assist decision makers in determining the allocation of conservation resources within a complex multinational shared sea and provide direction to render future turtle research more applicable to inform conservation initiatives (Casale & Margaritoulis 2010).

4.3 Methods

4.3.1 Study area and habitat representation

The study area included the entire Mediterranean Sea, excluding the seafloor deeper than 1,000 m. Areas deeper than 1,000 m were excluded because a) most important foraging habitats for sea turtles are generally classified in shallow waters along the continental shelf, b) anthropogenic threats are mainly concentrated along the coast and c) the General Fisheries Commission for the Mediterranean (GFCM) recommended the prohibition of towed dredges and trawl nets fisheries at depths beyond 1000 m (Recommendation GFCM/2005/1 on the ''management of certain fisheries exploiting demersal and deep-water species'') which has been adopted by the EU (Regulation 1967/2006). We divided the Mediterranean Sea into planning units of 10 x 10 km, including coastal land areas with important nesting beaches. This resolution was chosen following EU guidelines on the use of a pan-European 10 x 10 km grid for spatial planning (Directive 2007/2/EC), and follows the resolution of other large-scale regional planning studies (Giakoumi et al. 2013; Levin et al. 2013; Mazor et al. 2013b; Mazor et al. 2014).

We compiled available sea turtle data into a database of three sea turtle habitat types (Fig. 4.1a) and turtle movement tracks. First, the locations of 131 loggerhead nesting beaches were collated from over thirty published resources (see Supplement material Table S4.1 for a complete list). We did not aim to predict potential additional (unreported) locations of beaches because female sea turtles display natal homing (Bowen et al. 1994) and factors that affect their site selection within this homing range are not well known (Garcon et al. 2009). Planning units along the beach within a 10 km radius from each known nesting site were designating as nesting beach. We created inter-nesting habitat data using a 10 km buffer from nesting beaches (Tucker et al. 1995; Waayers et al. 2011). These areas are important areas for female sea turtles during the time between laying clutches (Zbinden et al. 2007) as well as juvenile turtles making their way to the ocean post-hatching (Bolten 2003).

Given that sea turtle foraging habitat is not well known in the Mediterranean, we incorporated a modelling approach to determine current foraging habitats. We used MaxEnt (Version 3.3.3k; http://www.cs.princeton.edu/~schapire/maxent/ Phillips et al. 2004, 2006; Elith et al. 2011) to model the distribution of sea turtle foraging grounds from presence-only species data. We collated sea turtle sighting locations from EurOBIS (2014), several scientific papers and location and telemetry data contributed by seaturtle.org (see Table S4.2). Telemetry data points that were spatially aggregated exhibiting high sinuosity on the continental shelf (defined by the 200 m isobaths; Kallianiotis et al. 2000; Sardà et al. 2004) were included, because such patterns indicate foraging (Benhamou et al. 2004; McCarthy et al. 2010; Dodge et al. 2014). Thus, all straight linear movements (and those off the continental shelf) were excluded, resulting in a total of 9,058 data points. These point data were combined with 22 environmental variables (for a list of variables see Table S4.3). The resulting model was validated by a random sub-sampling method that was repeated 15 times and used 25% of the data (Phillips et al. 2004, 2006). To create a distribution map of suitable foraging habitat we used the 10 percentile training presence logistic threshold (>0.36). By using this threshold, we defined suitable habitat to include 90% of the data we used to develop the model because 'true' absence data were not available. For further details on the MaxEnt distribution model see Appendix S4.1.

To represent sea turtle migration movements we compiled available satellite tracking data from EurOBIS (2014) and seaturtle.org (Table S4.4 for full references). A total of 34 tracks from individual sea turtles were collected from a variety of sources across the Mediterranean Sea and were used in this study (Fig. 4.1b – individual tracks cannot be shown due to data protection).

4.3.2 Determining the value of sea turtle information for conservation

Here we examined the value of different sorts of information about sea turtles using systematic conservation planning, which provides a platform for achieving explicit objective driven conservation goals (Margules & Pressey 2000). Conservation targets are a key component of this method and specify how much of each conservation feature (species or habitat) to protect within the study area. We applied Marxan, a commonly used spatial planning tool to determine a range of near-optimal spatial plan configurations of planning units that satisfy a suite of conservation targets for the least possible cost (McDonnell et al. 2002; Ball et al. 2009). All planning scenarios were run with 1000 repetitions and a uniform planning unit cost so we could focus on the effects that different kinds of information have on spatial priorities.

We explored conservation priorities for sea turtles with increasing complexity of input data. The changes in spatial priorities signify the potential gain to be derived from investing in the gathering of additional (and more complex) information to improve the adequacy of conservation priority sites to protect the entire turtle life cycle. First, we prioritised using the extant distribution range of sea turtles, then by multiple habitat types (foraging, nesting and inter-nesting), followed by movement information extracted from mark-recapture data and finally, the incorporation of satellite tracking data. Our targets vary according to approach (Table 4.1; Appendix S4.2).

Table 4.1. The four planning approaches compared in this study. These approaches include increasing amounts of data and information on the distribution and movement of sea turtles. Each plan aims to derive conservation priorities for loggerhead sea turtles (*Caretta caretta)* in the Mediterranean Sea, and uses systematic conservation decision tool Marxan. For details on how we determined targets see Appendix S4.2.

Approach for sea turtles conservation planning	Targets	How connectivity was incorporated
1. Species Distribution Range	The distribution of sea turtles as a whole (not per habitat type) overall target = 20%	Not at all
2. Habitat Differentiation	Nesting = 60% Inter-nesting habitat = 40% Foraging habitat = 20%	Targets for habitats used in different life-stages
3. Mark-Recapture	Nesting = 60% Inter-nesting habitat = 40% Foraging habitat = 20%	Connections between the priority habitats
4. Tracks	Nesting = 60% Inter-nesting habitat = 40% Foraging habitat = 20%	Connections between each track is prioritized

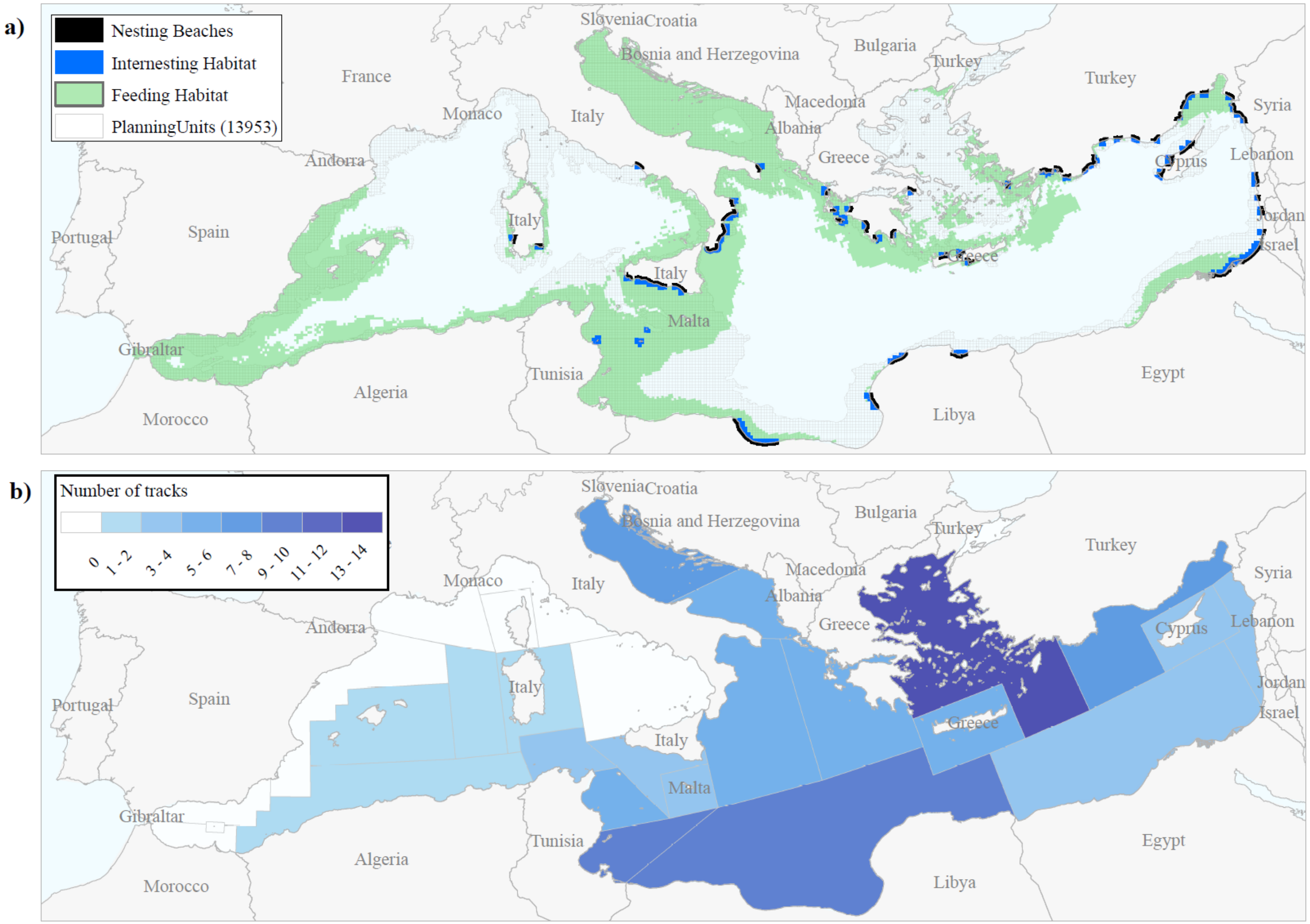


Figure 4.1. a) Three types of loggerhead sea turtle (*Caretta caretta*) habitat: nesting beaches, inter-nesting habitat and foraging habitat. b) Map of the Mediterranean Sea divided by geographical sub areas as determined by the General Fisheries Commission of the Mediterranean Sea (GSCM). The total number of sea turtles tracks that cross each sub area were calculated and represented in this map. Individual tracks were unable to be displayed due to data protection, see Appendix S4.2 for further information.

We parameterised Marxan both without representing any connections between planning units (used for Approach 1: species distribution range and Approach 2: habitat differentiation; Ball et al. 2009; Table 4.1) and by incorporating ecological flows into the objective function (used for Approach 3: mark-recapture and Approach 4: tracks; Beger et al. 2010a; 2010b; Table 4.1). When including flows, we calibrated the Connectivity Strength Modifier (CSM – for methods see Beger et al. 2010b) to 50 (Fig. S4.1). Below we explain each approach to identify turtle conservation priorities. We compared the four approaches by Spearman Rank Correlations of each selection frequency output from Marxan, and by mapping the resulting spatial conservation priorities.

Approach 1: Species distribution range

In this approach we represented the overall distribution of loggerhead sea turtles by a single broad distribution range in the Mediterranean Sea (combining nesting, inter-nesting and foraging habitat data into one single distribution range). This is a basic approach that is commonly used in conservation planning given the common paucity of fine-scale spatial habitat data (e.g., IUCN distribution ranges).

Approach 2 Habitat differentiations

For this approach we set specific conservation targets for nesting, inter-nesting and foraging habitats, simulating a situation where the three main habitats used by turtles are known. Dividing the broad distribution range into specific habitats with set targets ensures that priority conservation areas will be selected for each habitat type.

Approach 3: Mark-recapture information

Mark-recapture studies define at least two points on a turtle's travel, its start (tagging location) and end points (recapture location). To represent this type of information in conservation planning, we targeted the three habitats used by turtles while also ensuring a link between nesting and foraging sites. Here, we simulated mark-recapture data by using tracking routes to select planning units that linked nesting beaches with foraging habitat. To do this we used the selection tool in ArcGIS (ESRI 2010) and identified tracks which traversed planning units and demonstrated obvious foraging behavior (e.g., zigzagging in one spot known as track sinuosity; McCarthy et al. 2010). Tracks that were unclear or did not move across more than 50 planning units were discarded from the analysis, because Mediterranean loggerhead sea turtles typically move more than 600 km between nesting and foraging grounds (Zbinden et al. 2008). Nesting beaches were identified by tracks and also complemented by information about the nesting beaches which were usually the location where the satellite transmitter was attached to the turtle. This analysis enabled us to allocate connectivity values (assigned in the boundary length file) between foraging and nesting planning units at either end of the track, representing non-directional connectivity in Marxan (Beger 2010b).

Approach 4: *Tracks*

Mark-recapture data provides information to help link habitats at either end of turtle migrations. However, this method does not capture information about the pathways turtles take to cross vast distances. To incorporate links between habitats along the entire journey, we applied a method that incorporates telemetry-derived movement information into Marxan (Beger et al. 2013). Here we assigned connectivity values by developing a connectivity matrix created in MATLAB (2013) that connects all planning units along the sea turtle track. By symmetrically linking all planning units along an individual turtle's pathway, this method allows for spatial dependencies to exist between places that are not adjacent to each other (Beger 2010b). Planning units that are travelled through by more than one individual turtle are deemed increasingly important sea migration pathways and contribute more to the connectivity of the solutions.

We tested how the number of telemetry tracks altered the resulting conservation plan. To investigate the value of information (for identifying conservation priorities) when increasing the number of tracks, we randomly selected a fixed number of tracks from the pool of known tracks; 0 (no tracks), 5, 10, 15, 20, 25, 30, 34. The Marxan analysis was repeated ten times for each group of tracks to account for variability in the selected tracks. From these solutions we calculated the Spearman rank correlation of the selection frequency outputs and compared it with that of a solution that includes all 34 tracks. To further examine the increased inclusion of telemetry tracks, we used a Bray-Curtis dissimilarity matrix method as described in Linke et al. (2012) and displayed our results in a dendrogram. This method compared the Marxan best solution outputs (solution with the lowest objective function score) when run with different numbers of tracks.

4.3.3 Synthesising our findings to identify conservation priorities

Here we derived conservation priorities for the loggerhead sea turtles of the Mediterranean Sea using the most informative planning approach that included all available information. When identifying real-world conservation priorities it is important to incorporate the associated cost of actions (Naidoo et al. 2006; Wilson et al. 2007; Carwardine et al. 2008; Mazor et al. 2014). Thus, we combined this approach with a cost layer (Fig. 4.2) based on the opportunity cost of fishers and management cost of employing rangers for protecting nesting beaches (see Appendix S4.3 for detailed methods). We determined priority areas for sea turtle conservation and ensured all targets were met (here the CSM was set to 500; Fig. S4.1). We considered areas with planning units that were selected in over 50% of solutions as high priority areas for sea turtle conservation.

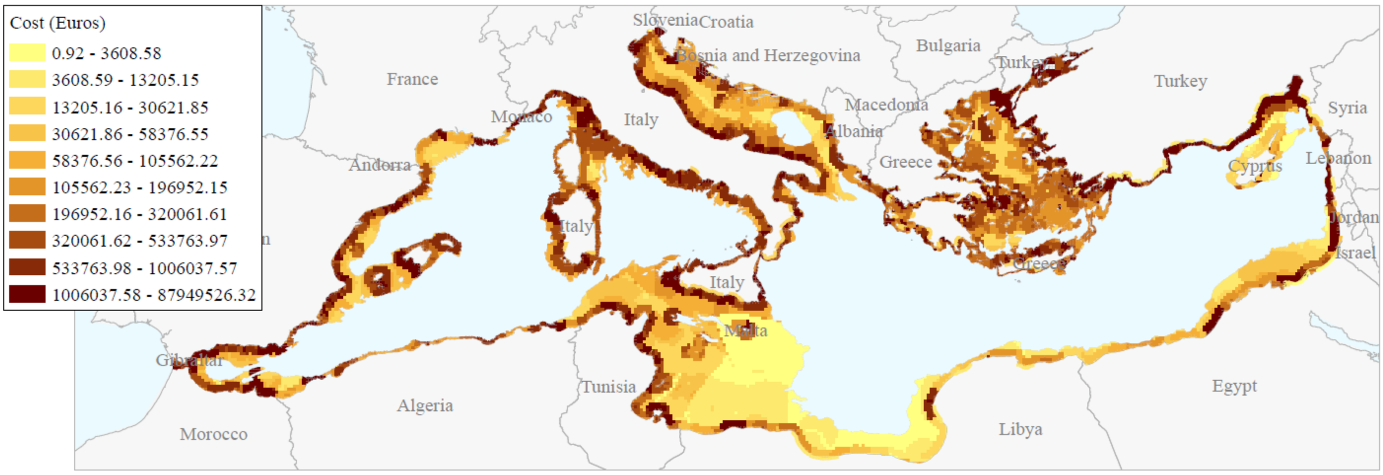


Figure 4.2. Cost layer used to determine priority areas for sea turtle conservation in the Mediterranean Sea. The cost in Euros is defined in each 10 x 10 km planning unit. This cost layer includes the opportunity cost of fishers when reserving marine waters for sea turtle conservation, and the management cost of rangers to protect nesting beaches based on a human disturbance metric. Full methods of this cost layer are described in Appendix S4.3.

4.4. Results

4.4.1 Conservation priorities

We identified six major conservation priority areas for loggerhead sea turtles in the Mediterranean Sea when incorporating breeding, feeding, migration (Approach 4 – Tracks) and cost information (Fig. 4.3). These high conservation value areas include: the Adriatic Sea (no. 1), coastal areas of southern Aegean, Greece (no. 2), coastal sections of southern Turkey (no. 3), coastal parts of the Levantine Sea (no. 4), and coastal regions around the Gulf of Sidra, Libya (no. 5, 6). Comparing these priorities in Figure 4.3 with sea turtle habitat in Figure 4.1, we can understand the sea turtle information type that drives the selection of these areas. One of these priority areas (no.1) captures the entire Adriatic Sea as a critical area for the protection of foraging and migrating sea turtles. Three other priority areas emphasise nesting beach protection (no.3 Turkey, no. 5, 6 Libya). The priority in eastern Egypt combines foraging and nesting habitats (no. 4), and the area in the southern Aegean, Greece, was selected as an important migration pathway (no. 2).

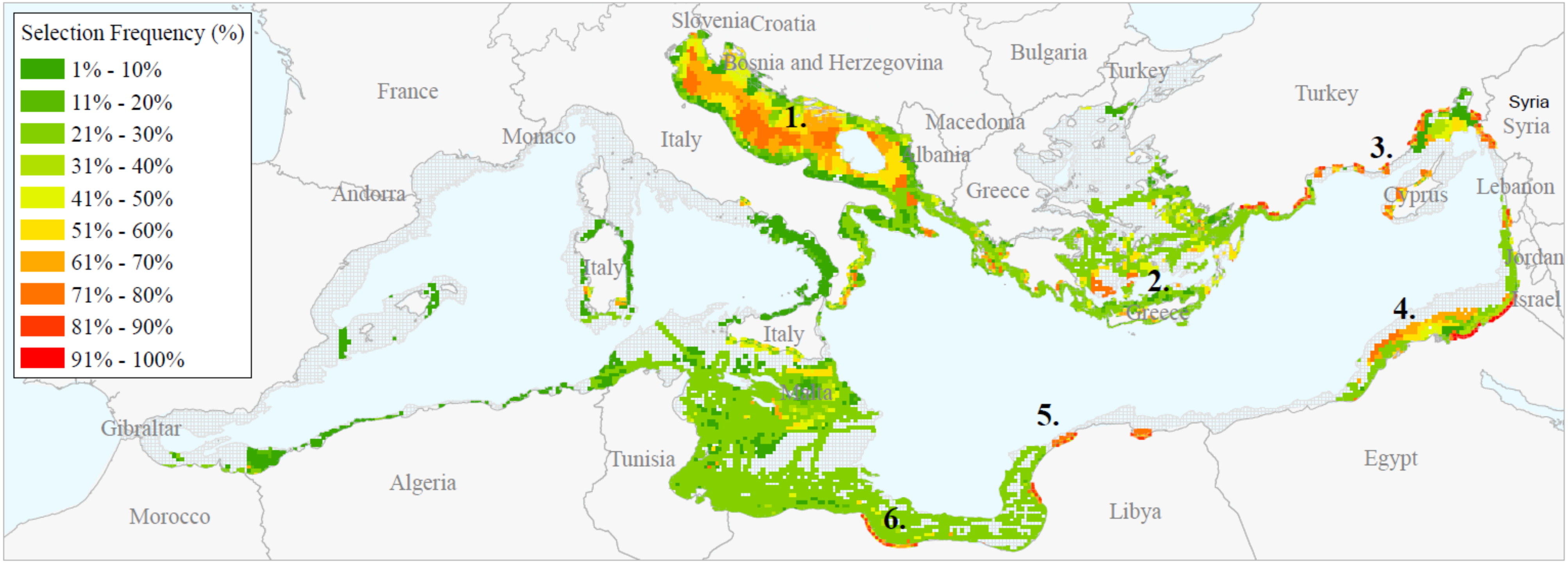


Figure 4.3. Priority areas for conservation of sea turtles in the Mediterranean Sea: the Adriatic Sea (no. 1), coastal areas of southern Aegean, Greece (no. 2), coastal sections of southern Turkey (no. 3), coastal parts of the Levantine Sea (no. 4), and coastal regions around the Gulf of Sidra, Libya (no. 5, 6).

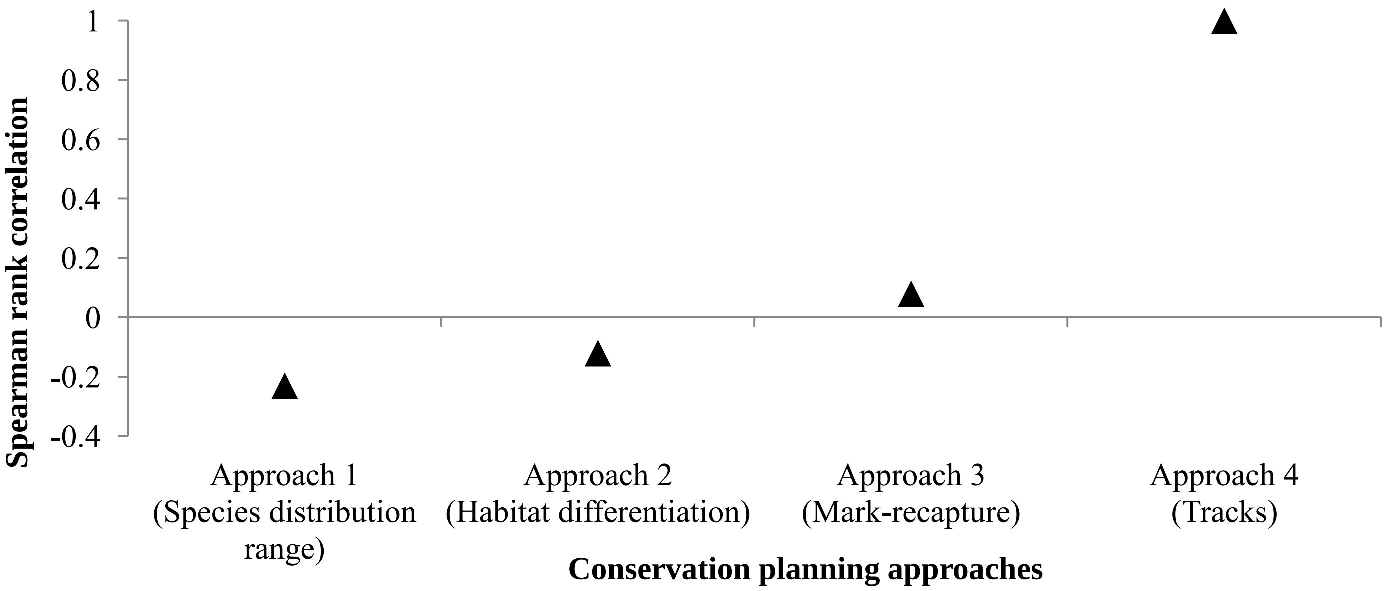
4.4.2 The value of sea turtle information for conservation

Conservation priorities that were evident in Approach 4 (Tracks) were not well represented in the other three approaches. For example, Approach 3 (Mark-Recapture Information), which had the highest Spearman rank correlation coefficient of the three scenarios when compared with a plan that incorporates tracking data (Approach 4 – Tracks), indicated that the spatial prioity areas from the plans do not significantly overlap (rho = 0.08). Thus, results show that links between habitats are not protected by chance when protecting sea turtle habitat, but need to be separately represented.

We found that conservation priorities substantially changed as we changed the approach (Fig. 4.4a; Fig. 4.5). Despite the weak correlations, we found that approaches that incorporated more habitat and movement information (e.g., Approach 2 habitat differentiation rho = -0.12 and Approach 3 Mark-Recapture Information rho = -0.23) than a broad species distribution rang (Approach 1 rho = -0.08), were more successful at capturing migration pathways (comparison with Approach 4 – Tracks) in the resulting spatial plans. Including movement data can also increase the cost of conservation plans (see Table S4.5) as movement corridors may be more costly to protect.

Even a small number of tracks (~5) can substantially increase the correlation (rho = 0.6) with plans that include all thirty-four tracks (Fig. 4.4b). We discovered that the largest Bray-Curtis dissimilarity was between conservation plans that did include sea turtle tracks and those that did not (see Group A vs. Group C in Fig. 4.6). The second largest dissimilarity was between plans that had a low number of tracks (Group B and Group D in Fig. 4.6) and a corresponding low spearman rank correlation (~ rho <0.7 Table S4.6) when compared with solutions that included ≥20 tracks and resulted in a higher spearman rank correlation (~ rho >0.7; Group C in Fig. 4.6). This dissimilarity was due to the low number of tracks (5-15 tracks) included in the plans and because the spatial variability captured was insufficient for the entire region. Given these results it seems that plans with >20 tracks were sufficient at capturing the spatial heterogeneity of turtle movement across the Mediterranean Sea. Thus, plans with over twenty tracks did not vary considerably to those with 34 tracks.

a)



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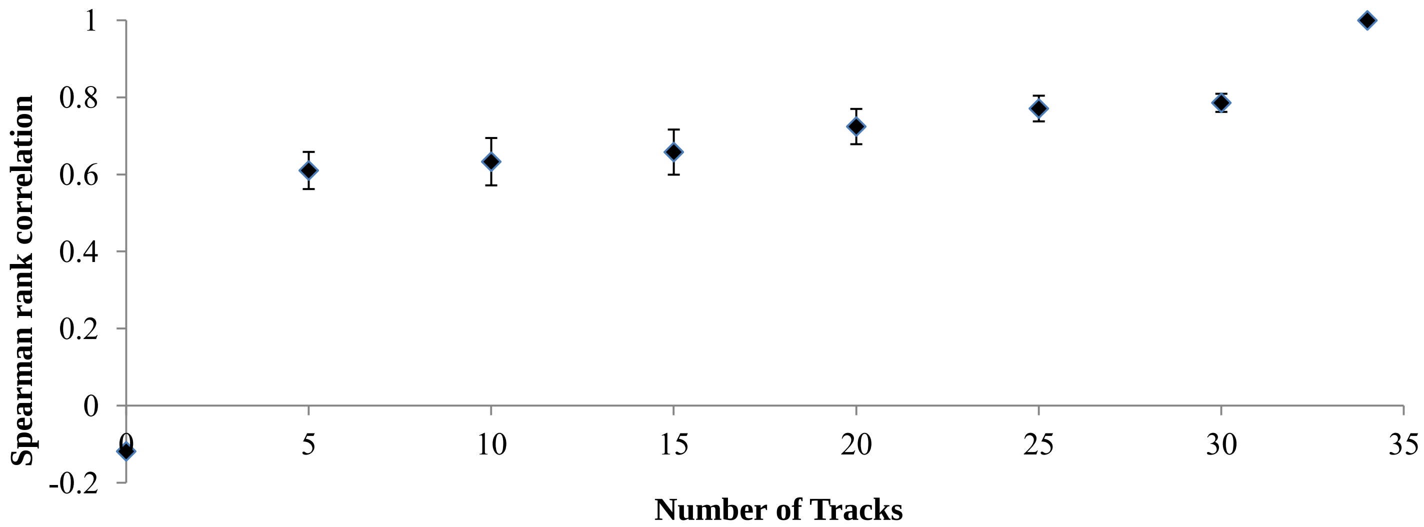

Figure 4.4. a) Spearman rank correlation of selection frequency outputs, comparing four conservation plans with increasing data complexity on sea turtle movement and habitat: Approach 1 - single species distribution range, Approach 2 - habitat differentiation (nesting, inter-nesting, foraging), Approach 3 – three habitat types and movement information from mark-recapture data, and Approach 4 – three habitat types and movement information from 34 sea turtle tracks. b) Graph of the average Spearman rank correlation of selection frequency outputs, comparing scenarios with a subset of tracks vs. scenarios with all 34 tracks. The standard deviation is shown for each scenario (calculated from ten repeated Marxan runs). This analysis used an equal cost for each planning unit.

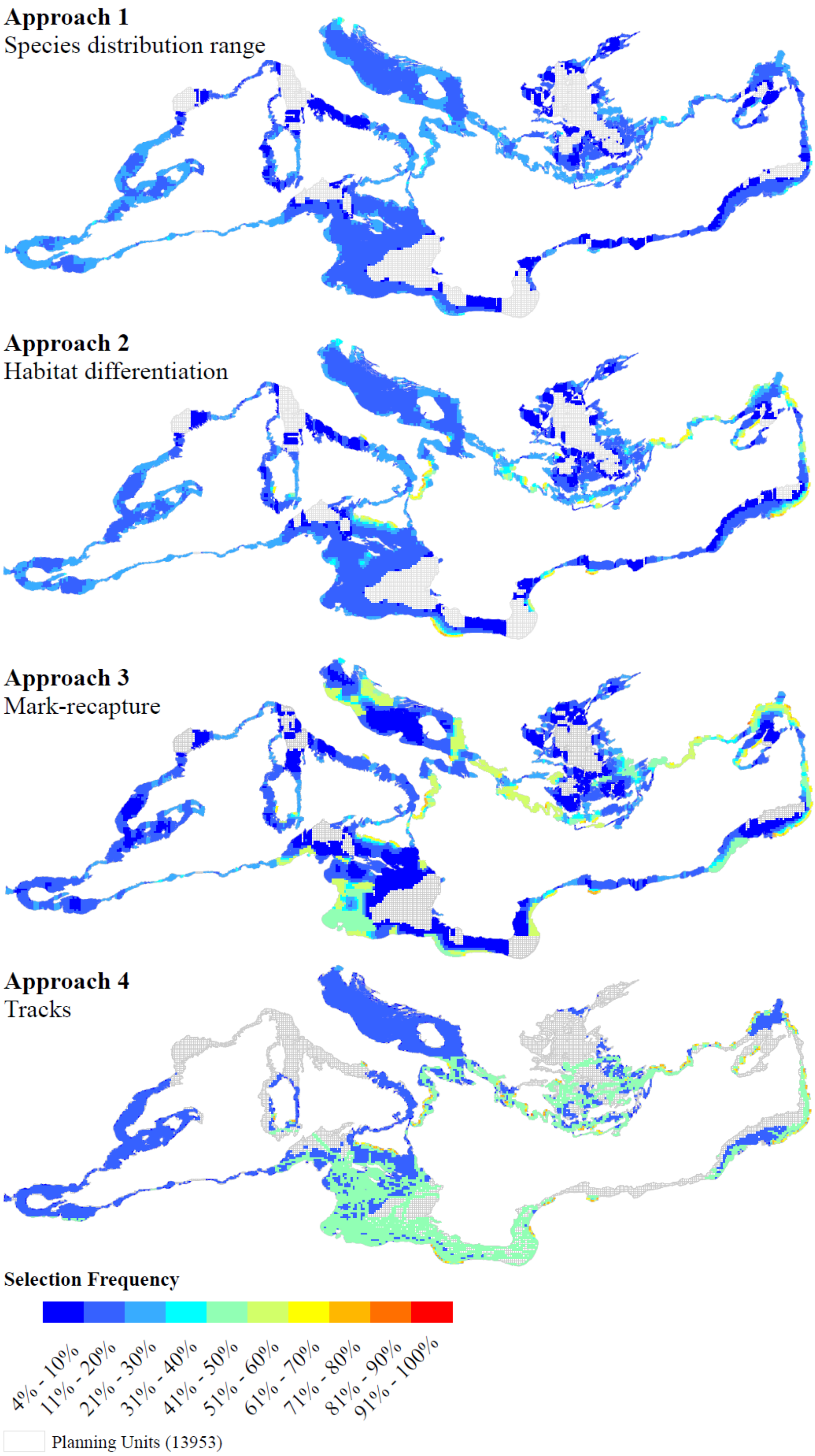

Figure 4.5. Maps of four conservation plans in the Mediterranean Sea with increasing data complexity for sea turtle movement: Approach 1 - single species distribution range, Approach 2 - habitat differentiation (nesting, inter-nesting, foraging), Approach 3 – three habitat types and movement information from mark-recapture data, and Approach 4 – three habitat types and movement information from 34 sea turtle tracks. Priority areas are those planning units that have a high percentage of selection (selection frequency).

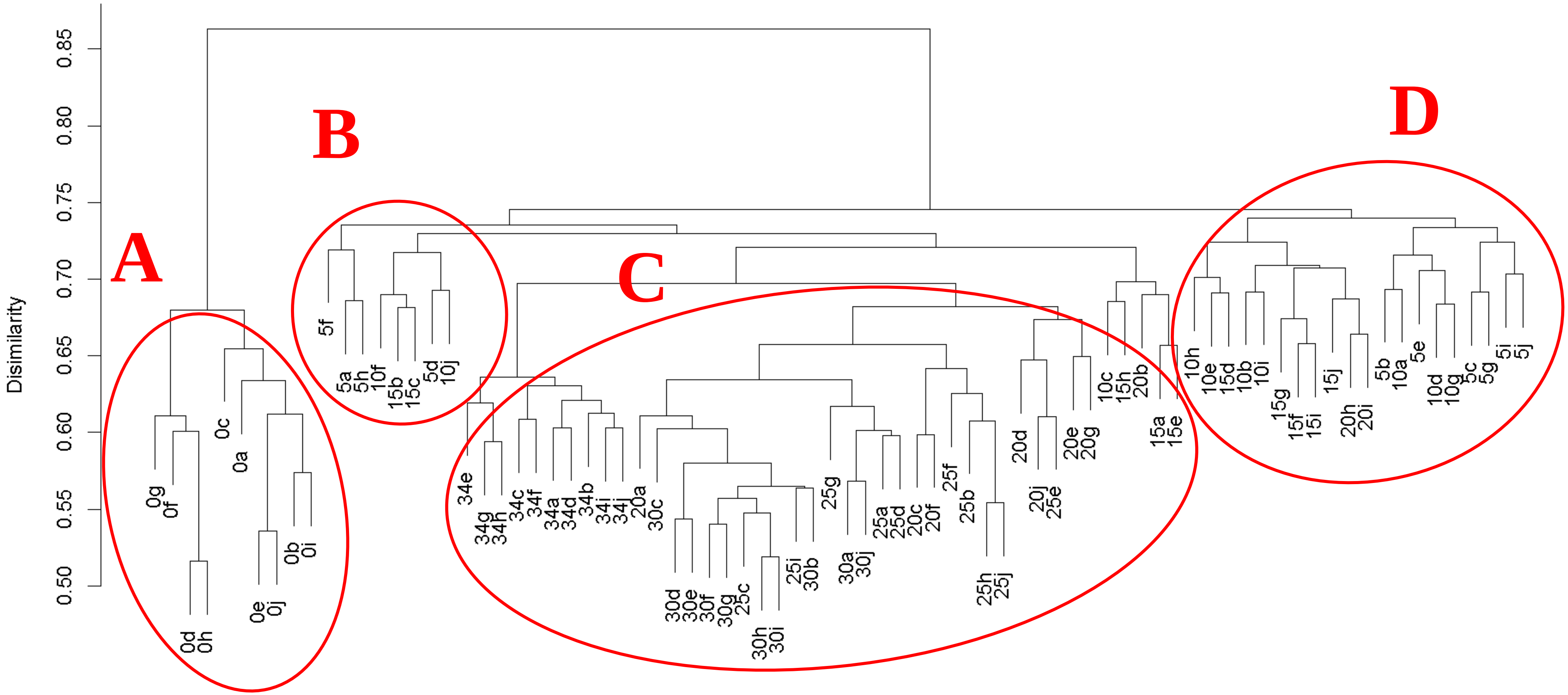

Figure 4.6. Dendrogram comparing the dissimilarity of solutions (Bray-Curtis dissimilarity matrix method; Linke et al. 2012) with increasing numbers of tracks (0, 5, 10, 15, 20, 25, 30, and 34 tracks, number of individual tracks represented by the number after the "dash"). The main split between solutions is between analyses without tracks and those that include tracks (Group A and B). The letters after each tracking group number are to enable the relation with Table S4.6.

4.5 Discussion

In this study we identified six priority areas for conserving loggerhead sea turtles at a whole Mediterranean Sea scale (Fig. 4.3). Our approach incorporated information about sea turtle breeding and feeding habitats, migratory connections via telemetry data and the conservation cost of reducing both sea and land threats. All priority areas were within the eastern part of the Mediterranean Sea indicating that loggerhead conservation efforts should be focused in this region. One of the most important priority areas was the Adriatic Sea; this is strongly supported by literature as an important area for foraging and migration (Lazar et al. 2004; Casale et al. 2004; Zbinden et al. 2008; Hays et al. 2010). The priority areas identified in this study (Fig. 4.3) can help guide the allocation of sea turtle conservation effort in the region as well as assist the recognition of areas that require between-country collaboration (Mazor et al. 2013b).

The migratory nature of sea turtles renders its conservation an inter-governmental and regional issue (Margaritoulis et al. 2003). Our study highlights important land and sea areas for sea turtle conservation. Given that we have incorporated migration connectivity in this study, the priority areas here are connected. We found that the Egyptian coastline was an important conservation hotspot for loggerhead sea turtles. This priority area includes both nesting beaches on the Sinai Peninsula (supported by Kuller 1999; Clarke et al. 2000) and foraging habitat near the Nile Delta due to shelf sea grass *Posidonia oceanica* beds (supported by Laurent et al. 1996; Clark et al. 2000). Studies indicate that this foraging area is connected to rookeries in Turkey, Cyprus and Greece (Broderick et al. 2007; Casale et al. 2008). Similarly, the Adriatic Sea feeding habitat (no. 1; Fig. 4.3) is essential for nesting populations from Greece (Lazar et al. 2004), and our priority region in the southern Aegean (no. 2; Fig. 4.3) represents an important sea turtle migration pathway region (high number of tracks crossing this region Fig. 4.1b, and absence of important habitat features Fig. 4.1a) for travel from Greece to Turkey (Schofield et al. 2010). Thus, this study highlights the importance for incorporating sea turtle movements in conservation prioritisation and the need for collaborative efforts between countries to protect loggerhead sea turtle populations.

This study highlights the value of incorporating critical habitat and migration information into sea turtle conservation planning. Our results showed significant changes in spatial priorities when increasing the amount of sea turtle information (see four approaches; Fig. 4.4; Fig. 4.5). Sea turtle migration was best represented by incorporating the entire movement track, because critical habitat information (Approach 2) or mark-recapture data (start and end points of movements; Approach 1) were not able to achieve the same outcome (Fig. 4.4; Fig. 4.5). We managed to access 34 sea turtle tracks in this study and we discovered that even a small number of spatially heterogeneously distributed tracks (e.g., five) can help robustly assign conservation priority sites that encompass the migratory life cycle of sea turtles (Fig. 4.4b; Fig. 4.6). Thus, we suggest that future conservation plans for sea turtles should attempt to incorporate available habitat and telemetry data where possible.

Our results suggest that in order to convey sea turtle habitat connectivity in conservation plans the heterogeneity of tracks across the study area is perhaps more important than the number of tracks (Fig. 4.6). However, given our limited sample size (34 tracks) and the difficulty of performing further analysis this could also be attributable to the length of the tracks (Fig. S4.2). Given this case in the Mediterranean Sea, we found that >20 sea turtle tracks that were widely sampled across the study region were sufficient at deriving sea turtle movement. In terms of explaining sea turtle movements we suggest that perhaps only a relatively small well-sampled number of tracks are required, because similar migratory trajectories are observed from sea turtles that nest at the same beach. For example, three tracks from Dalyan beach (Turkey) headed to the Gulf of Gabes (Tunisia). Moreover, having two or more tracks from the same location that travel along the same migration path, will not contribute new connectivity information for conservation plans as presented here. We suggest that future efforts should be made to track sea turtles from a great diversity of geographic locations in the Mediterranean, particularly targeting turtles nesting in the eastern part of the basin (Lebanon, Israel, Egypt) where less tracking data have been collected (Fig. 4.1b).

Sea turtle telemetry studies provide a wealth of connectivity information that is not often applied to conservation planning. Godley et al. (2008) presented concerns over the use of expensive satellite tracking technology (estimated at US$1.25–5 million per annum) for sea turtles when resources for conservation are limited. Their paper refers to a lack of clear management and conservation applications from the growing body of tracking literature. Our study provides a possible answer to these concerns. To guide the collection of further telemetry data, we found that heterogeneity of data may be more valuable than the quantity. This result could perhaps provide better direction for the timely and costly collection of telemetry data. Future efforts should aim to extract all available turtle telemetry data where possible, perhaps using monetary incentives or intellectual safeguards, and compile databases for the incorporation of turtle migration into conservation plans. We propose that already established collaborative frameworks such as the EU, or the IUCN, could be a potential starting point.

Another challenge in addressing sea turtle movement is determining how much connectivity is needed. Relying on too few tracks means there is also a risk of over-fitting to a limited number of data tracks. As an attempt to overcome these challenges, this study used a calibration method where planning units that contained a track were selected over 50% of the time (Fig. S4.1). The method ensures that connectivity is represented, but it does not necessarily mean that 50% of all migration links are captured in the solution. Determining the level of connectivity that is needed will largely depend on the species of interest as well as the conservation budget and data availability. For example, connectivity is especially important for sea turtles that exhibit high mortality rates within movement pathways (Lewison et al. 2004; Casale et al. 2011). However, connectivity may not be particularly useful for species that are less threatened during the movement/migration phase or those that have large dispersal patterns without clear migration trajectories. Importantly, the cost of a conservation plan increases as the importance of connectivity is increased (high cost areas are forced to be chosen; Table S4.5). Hence, we suggest that the level of connectivity should be well determined and perhaps a measure of minimum connectivity should be set per species.

Our study is based upon adult loggerhead movement patterns (mainly females post-nesting), whereas juvenile movements may be quite different (UNEP 2011). The integration of genetic information within conservation planning (Grivet et al. 2008; Sork et al. 2009; Ndobe et al. 2011; Beger et al. 2014) of sea turtles could also be a valuable factor to investigate due to their complex life history. In this study we examined two major threats (accidental bycatch and nesting beach disturbance) that can be mitigated by no-take protected areas. However there are other conservation actions to be taken as well as other threats to abate. Other actions that could be considered in future work include; implementing Turtle Exclusion Devices (TEDs), combing spatial prioritisation with Population Viability Analyses (PVA), restricting artificial night time lights along beach fronts, preventing coastal developments and implementing educational and learning programs.

This work aims to highlight a broad starting point for the future of large-scale conservation prioritisation of loggerhead sea turtles in the Mediterranean. Given current data limitations, we have presented a static conservation plan based upon annual distribution data. We have attempted to collate all possible freely available data, but we are aware that our findings may be affected by sample size, observational and reporting bias. Our migration pathways do however match well with a study by UNEP (2011) with a larger sample of tracks. With the expansion of future data availability, research should aim to expand this work to include seasonal variations, explore temporally dynamic plans (Grantham et al. 2011) or plans for various loggerhead life stages, as well as explore the incorporation of more sea turtle tracks to examine whether sea turtle priorities differ.

In summary, our study shows that the selection of priority areas for sea turtle conservation can be considerably improved by using available sea turtle information. Incorporating breeding, feeding and migration information, helps address the entire migration life cycle of sea turtles. Thus, this work highlights the importance of focusing sea turtle conservation efforts in the sea as well as on the land. When there is only a short widow of time to act for threatened species it is critical that decision makers invest and act in areas which will be most effective at ensuring species persistence (Bottrill et al. 2008). We recommend future research aims to examine the value of existing information for conservation plans of sea turtles around the world as well as other migratory species (Runge et al. 2014), especially those that are threatened. Only with the integration of available information into conservation planning tools that are transparent and systematic, will effective conservation decisions be achieved.

4.6 Supplementary material

Table S4.1. Nesting data compiled from literature. A total of 131 loggerhead (*Caretta caretta*) nesting beaches were recorded.

References for nesting data compiled from literature
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Table S4.2. References for data extracted from EurOBIS (2014), scientific literature and seaturtle.org to collect point data on sea turtle locations when foraging.

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seaturtle.org	RAC/SPA-SZN Tracking of Mediterranean Marine Turtles. A project of Stazione Zoologica Anton Dohrn in conjunction with the partners and sponsors detailed below. Data accessed online: http://www.seaturtle.org/tracking/index.shtml?project_id=358 • RAC/SPA - Regional Activity Centre for Specially Protected Areas • MEPA - Malta Environment and Planning Authority • EGA - Environmental General Authority, Libya • MBRC - Marine Biology Research Centre, Tajura • The Sea Turtle Rescue Center (DEKAMER) – Turkey

Table S4.3. Environmental Variables (Variables included in final model marked with *)

Data Type	Data Source	Citation
Ph	BIO-Oracle	Tyberghein et al. 2012
Chlorophyll *	BIO-Oracle	Tyberghein et al. 2012
Salinity *	BIO-Oracle	Tyberghein et al. 2012
SST minimum *	BIO-Oracle	Tyberghein et al. 2012
SST maximum	BIO-Oracle	Tyberghein et al. 2012
SST range	BIO-Oracle	Tyberghein et al. 2012
SST mean	BIO-Oracle	Tyberghein et al. 2012
Nitrate	BIO-Oracle	Tyberghein et al. 2012
Phosphorous *	BIO-Oracle	Tyberghein et al. 2012
Calcite *	BIO-Oracle	Tyberghein et al. 2012
Silicate *	BIO-Oracle	Tyberghein et al. 2012
Dissolved Oxygen	BIO-Oracle	Tyberghein et al. 2012
Bathymetry *	EMOD net	http://www.emodnet-hydrography.eu/
East/West aspect (biogeo01)	MARSPEC	http://www.marspec.org/
North/South aspect (biogeo02)	MARSPEC	http://www.marspec.org/
Plan Curvature (biogeo03)	MARSPEC	http://www.marspec.org/
Profile Curvature (biogeo04)	MARSPEC	http://www.marspec.org/
Distance to shore (biogeo05) *	MARSPEC	http://www.marspec.org/
Bathymetric Slope (biogeo06**)**	MARSPEC	http://www.marspec.org/
Concavity (biogeo07**)**	MARSPEC	http://www.marspec.org/
Near-surface currents a drifter-derived seasonal climatology of global near surface currents (nscurr)	NOAA	http://www.aoml.noaa.gov/phod/dac/dac_meanvel.php Citation: Lumpkin, R. & Johnson, G.C. (2013). Global Ocean Surface Velocities from Drifters: Mean, Variance, ENSO Response, and Seasonal Cycle. *J. Geophys. Res.-Oceans*, **118**, 2992–3006.
Tide	NASA Global Ocean Tide Model from TOPEX/POSEIDON Altimetry: GOT99.2	http://svs.gsfc.nasa.gov/stories/topex/tides.html Richard Ray, GSFC, NASA, author http://marinedataliteracy.org/examples/19990089548_1999150788.pdf

Table S4.4. Sea turtle tracking data sources. All data were obtained via EurOBIS (2014) and seaturtle.org. All data extracted from these sources is reference below.

Data Source	Reference
EurOBIS	Cañadas, A., Sagarminaga, R., de Stephanis, R., Urquiola E. & Hammond P.S. (2005). Habitat preference modelling as a conservation tool: Proposals for marine protected areas for cetaceans in southern Spanish waters. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **15**, 495-521.
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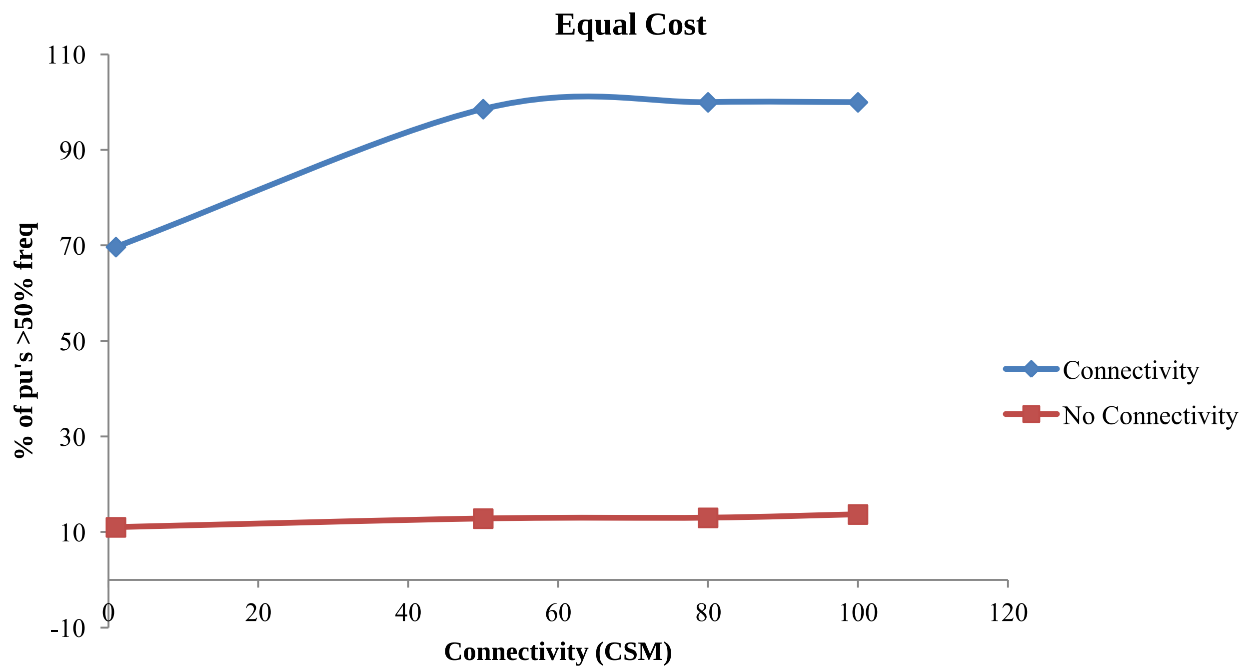
seaturtle.org Mediterranean group and partners	RAC/SPA-SZN Tracking of Mediterranean Marine Turtles. A project of Stazione Zoologica Anton Dohrn in conjunction with the partners and sponsors detailed below. Data accessed online: http://www.seaturtle.org/tracking/index.shtml?project_id=358 • RAC/SPA - Regional Activity Centre for Specially Protected Areas • MEPA - Malta Environment and Planning Authority • EGA - Environmental General Authority, Libya • MBRC - Marine Biology Research Centre, Tajura The Sea Turtle Rescue Center (DEKAMER) – Turkey

Table S4.5. The opportunity cost of each scenario when run with a cost layer and when cost is assumed equal. The Connectivity Strength Modifier (CSM; Beger et al. 2010b) was calibrated to 500 and 50 respectively (Fig. S4.1). All values in the table represent the average value when run in Marxan 1000 times. The "number of planning units" indicates the number of 10 x 10 km units needed for reservation to meet biodiversity targets.

Scenario	**Cost**	**Number of Planning Units**
(CSM = 500; Cost layer)		
Baseline	275,007,329	1353
Habitat Representation	432,474,291	1474
Connectivity by end points	764,774,031	1520
Connectivity by tracks	2,429,058,538	1917
(CSM = 50; Equal cost)		
Baseline	1,924	1,924
Habitat Representation	2,285	2,285
Connectivity by end points	2,624	2,624
Connectivity by tracks	2,413	2,413

Table S4.6. Spearman rank correlation coefficient when running conservation plans in Marxan with different numbers of sea turtle tracks (0, 5,10,15,20,25,30,34). The selection frequency outputs from Marxan were compared against a solution with all 34 tracks included. These values indicate the similarity between spatial priorities in the solutions. We tested the number of tracks with 10 repetitions to test for variation between selected tracks in our random samples (indicated by a letter).

No. of Tracks	a	b	c	d	e	f	g	h	i	j	Average	St. dev
0	-0.13	-0.12	-0.12	-0.12	-0.11	-0.12	-0.12	-0.12	-0.11	-0.13	-0.12	0.01
5	0.54	0.70	0.56	0.61	0.59	0.63	0.61	0.58	0.67	0.60	0.61	0.05
10	0.66	0.64	0.57	0.77	0.57	0.59	0.65	0.62	0.67	0.58	0.63	0.06
15	0.67	0.56	0.57	0.69	0.63	0.74	0.66	0.69	0.72	0.65	0.66	0.06
20	0.79	0.64	0.76	0.69	0.73	0.76	0.69	0.69	0.69	0.76	0.72	0.05
25	0.77	0.77	0.82	0.74	0.78	0.78	0.69	0.77	0.79	0.78	0.77	0.03
30	0.75	0.80	0.78	0.79	0.79	0.75	0.80	0.78	0.82	0.77	0.79	0.02
34	0.99	0.99	0.99	0.99	0.99	0.99	0.99	0.99	0.99	0.99	0.99	0.01



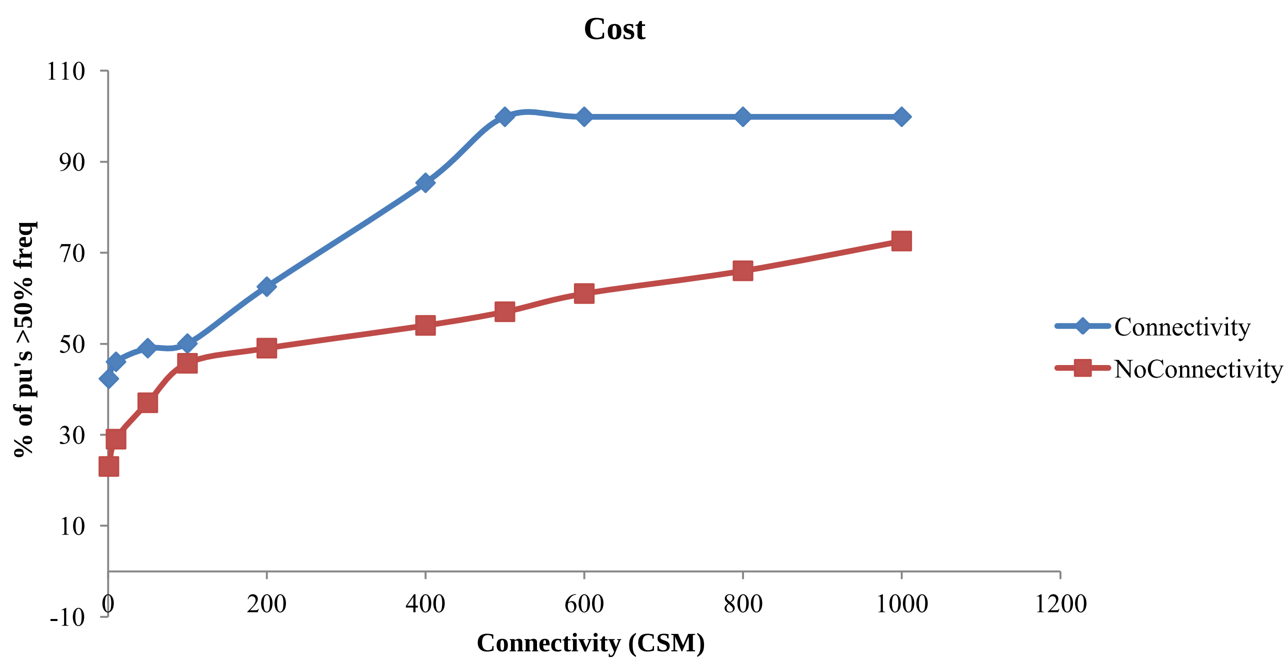


Figure S4.1. Graphs showing the trade-off curve of the connectivity strength modifier (CSM) with the number of connected planning units (those containing a sea turtle track). By assessing a trade-off curve with the number of planning units that overlap with tracking data we could determine the appropriate Connectivity Strength Modifier (CSM - Beger et al. 2010b). We aimed for planning units containing tracks to be selected >50% of the time when run 1000 times in Marxan. We used a CSM of 50 (equal cost) and 500 (cost).

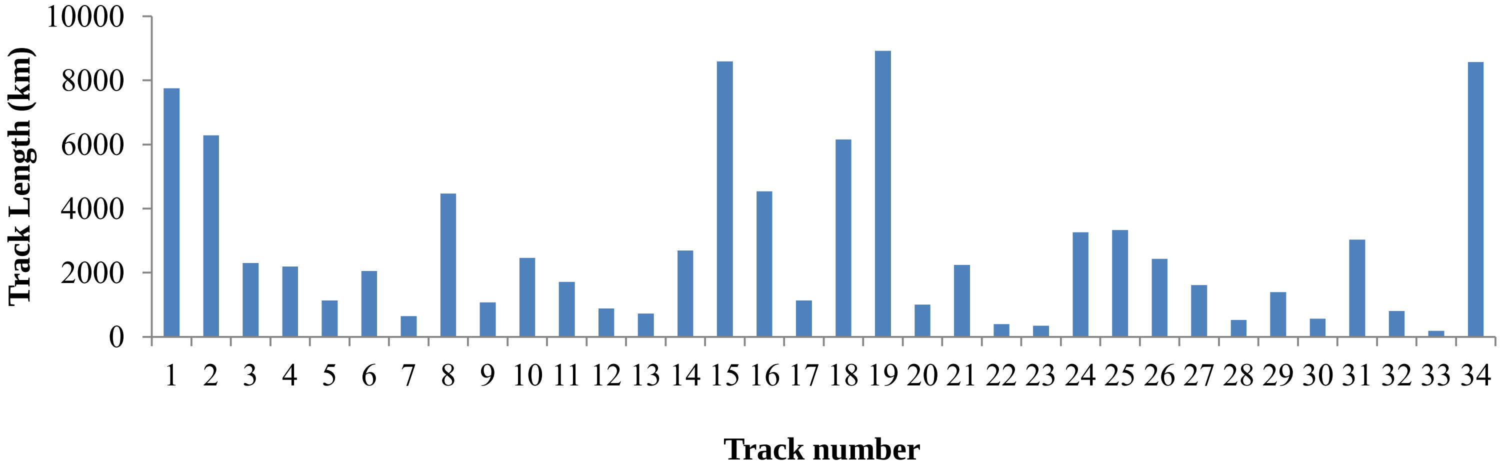


Figure S4.2. Graphs showing the length (km) of each of the 34 tracks used in this study. See Table S4.4 for the sources of the 34 tracks.

Appendix S4.1. Sea turtle foraging distribution model created using MaxEnt.

Predictor variables:

We considered 22 potential predictors (environmental variables; Table S4.3) which had some postulated connection to the ecological requirements of the sea turtle feeding habitat in the Mediterranean Sea. Pairwise Pearson correlations between variables in our resulting model were all less than 0.85 (Elith et al. 2006; Elith et al. 2010; Elith et al. 2011; see table below). For pairs of variables that were highly correlated we choose the variable that we considered more biologically meaningful according to sea turtle literature.

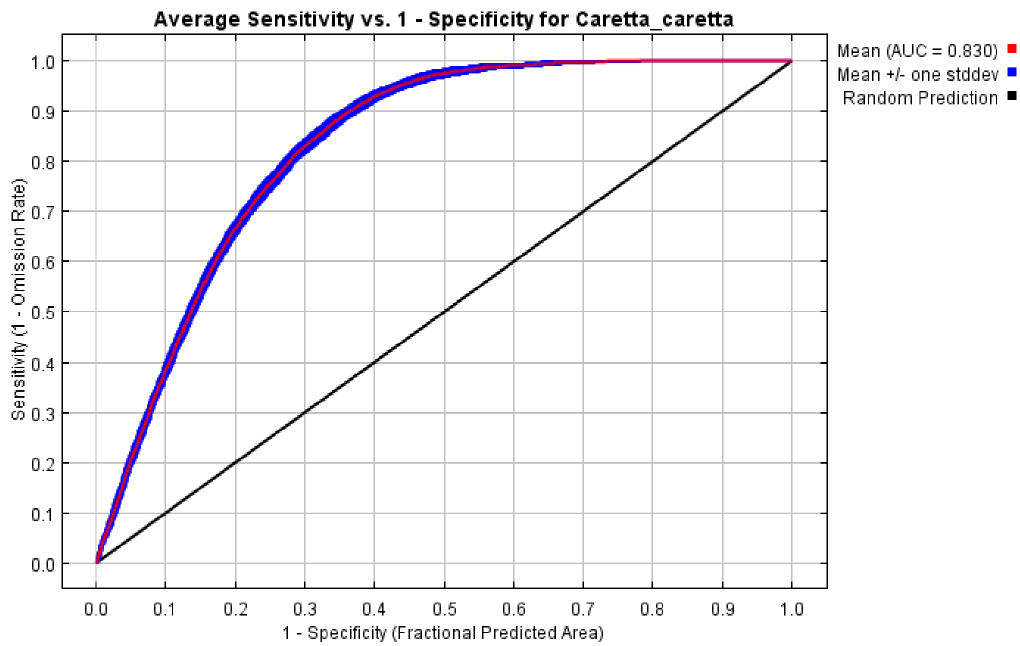
MaxEnt Settings:

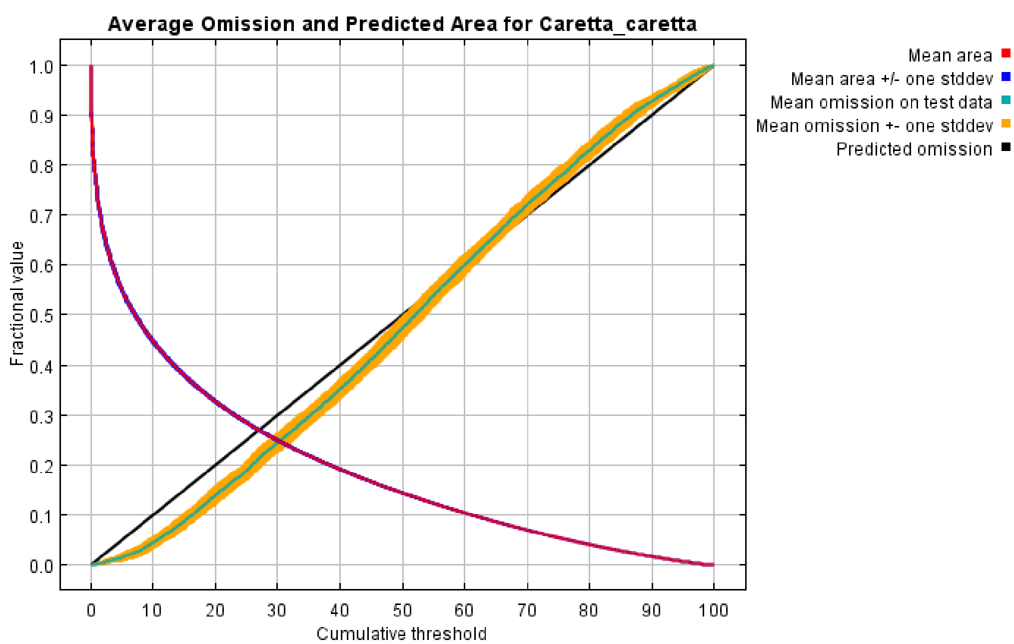
We used MaxEnt a maximum entropy based machine learning software for modelling species distributions from presence-only species records (Elith et al. 2011). Our settings in MaxEnt were: Hinge = 1 (smoothing), Bias grid (background), Replicates = 15, Random Test Percentage = 25 (sub-sampling method), Max Iterations = 5000. We attempted to correct for geographical sampling bias by incorporating a sampling bias grids with only countries that contain data. We also included a Jackknife test to compare important predictor variables in the resulting model. To validate the predictive performance of the resulting models we used a random sub-sampling method (75% training and 25% testing) that was run 15 times. We chose this approach to cross-validation to help prevent the models from being over fitted.

Resulting model:

To evaluate model performance we used two approaches, the omission rate and the area under the Receiver Operating Characteristic ROC curve (AUC). Any variable identified as not relevant at this point was removed, and MaxEnt was run following the above mentioned settings. This process was repeated until no non-relevant variable remained in the model (i.e. the decrease in training gain when any of the remaining variables was omitted from the full model was greater than 0.01). Our resulting model had eight predictor variables (see Table S4.3), an AUC of 0.83, and standard deviation of 0.006.

From our resulting model we used a 10 percentile training presence logistic threshold to define the minimum probability of suitable sea turtle foraging habitat. We used this threshold to create our distribution layer because the data we used may have some errors as we do not include true absence data. Using this threshold we defined the suitability of foraging habitat to include 90% of the data we used in making the model.





Pairwise Pearson correlations of variables used in resulting model.

Variables	sst_mi_clip2.asc	bath_clip2.asc	biogeo5_clip2.asc	calcite_clip2.asc	chlor_clip2.asc	phos_clip2.asc	sal_clip2.asc	silica_clip2.asc
sst_mi_clip2.asc		-0.171	0.251	-0.177	-0.277	-0.079	0.645	-0.364
bath_clip2.asc			-0.665	0.202	0.179	-0.147	-0.187	0.222
biogeo5_clip2.asc				-0.166	-0.164	-0.029	0.179	-0.162
calcite_clip2.asc					0.582	-0.085	-0.159	0.190
chlor_clip2.asc						0.022	-0.410	0.341
phos_clip2.asc							0.061	-0.044
sal_clip2.asc								-0.557
silica_clip2.asc								

Appendix S4.2. Setting conservation targets

These targets were set according to the EU guidelines from the Habitat Directive (Article 17 92/43/EEC as a species of Community interest in need of strict protection.). The Directive follows a broad approach with guidelines to protect between 20-60% of habitat or species distribution. We targeted 60% of nesting beaches, 40% of inter-nesting habitat and 20% of foraging habitat for the entire Mediterranean Sea. These targets are derived from EU additional guidelines for assessing sufficiency of Natura 2000 proposals (SCIs) for marine habitats and species (ETC Biological Diversity October 2009). For *Caretta caretta* these guidelines state that *"sites should be designated also for other life cycle stages than nesting where scientific evidence support regular presence in significant numbers"*.

Information for each sea turtle track. The start and end country that the tracks were found, starting positions were usually nesting sites. Further information is unable to be given due to data privacy.

Track	Country track started	Country track ended
1	Turkey	Greece
2	Malta	Italy
3	Libya	Tunisia
4	Libya	Algeria
5	Libya	Tunisia
6	Libya	Tunisia
7	Libya	Tunisia
8	Tunisia	Malta
9	Tunisia	Italy
10	Turkey	Turkey
11	Turkey	Tunisia
12	Israel	Turkey
13	Turkey	Tunisia
14	Turkey	Greece
15	Italy	Tunisia
16	Italy	Croatia
17	Italy	Slovenia
18	Italy	Italy
19	Italy	Italy
20	Italy	Italy
21	Croatia	Croatia
22	Greece	Croatia
23	Albania	Montenegro
24	Greece	Italy
25	Greece	Italy
26	Greece	Greece
27	Egypt	Turkey

28	Turkey	Turkey
29	Turkey	Tunisia
30	Turkey	Turkey
31	Turkey	Turkey
32	Turkey	Turkey
33	Egypt	Turkey
34	Turkey	Turkey

Appendix S4.3. Cost data

To determine spatial priorities for sea turtle conservation in the Mediterranean Sea we must identify the threats, the conservation actions and the costs involved (Wilson et al. 2007; Carwardine et al. 2008). One of the greatest causes of sea turtle mortality is bycatch; turtles caught unintentionally in fishing gear (Lewison & Crowder 2007; Alessandro & Antonello 2010; Casale 2011). The majority of sea turtle bycatch is caught at shallow depths: between ~10-30 m (Project Life Nature 2003-NAT/IT/000163), <60m (Laurent et al. 2001), 10-15 m (Rueda & Sagarminaga 2008), 11-30 m (FAO 2004). Longliners and trawlers are the fishing gear types accountable for most of this bycatch. We were unable to obtain biomass values for these specific fishing gears for each country; therefore we used the overall fishing industry as a proxy. We define the cost of protecting sea turtles from bycatch as the lost revenue of fishermen when a given area is made protected (opportunity cost). This opportunity cost was derived from biomass values obtained in the General Fisheries Commission for the Mediterranean (GFCM 2010) report and using an exponential weighting from port we derived monetary values. This opportunity cost also includes recreational fishing surrogate and includes aquaculture see Mazor et al. 2014 (we used Scenario 6) for a detailed explanation of the methods.

Another threat to sea turtles is anthropogenic disturbances along nesting beaches (Mazaris et al. 2009). These disturbances can be in the form of night light pollution, humans digging up nests, driving along beaches (Casale & Margaritoulis 2010). Such disturbance can affect the nesting females when finding a nesting beach, as well as the hatchlings when emerging and entering the sea, leading to mortality of sea turtles (Casale & Margaritoulis 2010). To prevent sea turtle mortality along nesting beaches our conservation action was to employ rangers to patrol and protect the beaches. To assign cost to the beaches we accounted for the cost of one ranger's salary and combined this with a metric which we term here the "Human Disturbance Index (HDI)". This HDI metric assumes that with greater anthropogenic activity the more cost (time and effort) is required to manage nesting beaches. Anthropogenic activity is measured by the weighted sum of human population density (raster grid for the year 2005; CIESIN 2005) and artificial night time lights (this accounts for commercial sites e.g., restaurants, ports, shopping precincts which would not be evident from only accounting for population density; night light data 30 arc second grids from year 2009 DMSP-OLS 2009). Nesting sites were buffered by a radius of 10 km, the approximate average distance people travel to the beach (Ünal & Williams 1999; Gale 2010). All planning units on the land (those that overlap with the 10 km buffer from a nesting site) were given the corresponding cost. We used a proxy for ranger cost, assuming that the ranger cost is different for each country and a reflection of the GDP per capita values (IMF 2013 http://www.imf.org/external/index.htm; Syria used Lebanon values as they are unavailable). All cost was converted to Euros to match the sea cost (fishing opportunity) by using the IMF annual exchange value for 2013.

Nesting beach cost = Ranger Cost + HDI (normalized between 0-100)

where,

HDI = a Population density + (1-α) Nightlight intensity

with α =0.6.

Thus, our resulting cost layer (Fig. 4.2) is comprised of a bycatch protection cost (Scenario 6 from Mazor et al. 2014) and a nesting beach conservation cost (Ranger cost and HDI metric).

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Chapter 5

Can satellite-based night lights be used for conservation? The case of nesting sea turtles in the Mediterranean

Night lights in the Mediterranean Basin region. Photo credit: NASA

Biological Conservation (2012) **159**, 63-72.

5.1 Abstract

Artificial night lights pose a major threat to multiple species. However, this threat is often disregarded in conservation management and action because it is difficult to quantify its effect. Increasing availability of high spatial-resolution satellite images may enable us to better incorporate this threat into future work, particularly in highly modified ecosystems such as the coastal zone. In this study we examine the potential of satellite night light imagery to predict the distribution of the Endangered loggerhead (*Caretta caretta*) and green (*Chelonia mydas*) sea turtle nests in the eastern Mediterranean coastline. Using remote sensing tools and high resolution data derived from the SAC-C satellite and the International Space Station, we examined the relationship between the long term spatial patterns of sea turtle nests and the intensity of night lights along Israel's entire Mediterranean coastline. We found that sea turtles nests are negatively related to night light intensity and are concentrated in darker sections along the coast. Our resulting GLMs showed that night lights were a significant factor for explaining the distribution of sea turtle nests. Other significant variables included: cliff presence, human population density and infrastructure. This study is one of the first to show that night lights estimated with satellite-based imagery can be used to help explain sea turtle nesting activity at a detailed resolution over large areas. This approach can facilitate the management of species affected by night lights, and will be particularly useful in areas that are inaccessible or where broad-scale prioritization of conservation action is required.

5.2 Introduction

Coastal zones are experiencing rapid population growth around the world (Turner et al. 1996) and attract increasing levels of tourism, trade and development (Shi & Singh 2003; Stancheva 2010). These anthropogenic pressures threaten biodiversity in the coastal environment, affecting the dynamics of flora and fauna populations and ecosystem processes (Chapin et al. 2000; Crain et al. 2009). While the effects of some human-caused threats have been examined in detail, our understanding of the consequences of artificial night lights on biodiversity in coastal areas, which have rapidly increased in both spatial extent and intensity in recent decades, remains limited (Longcore & Rich 2004).

Researchers have studied the effect of night lights on species for many years (Longcore & Rich 2004). Previous studies exploring the impact of artificial lights on organisms were mainly conducted by ecologists studying species of birds (e.g., Longcore 2010), sea turtles (e.g., Lorne & Salmon 2007), bats (e.g., Jung & Kalko 2010) and freshwater fish (e.g., McConnell et al. 2010). Results from these studies demonstrate that night lights can attract, repel, and disorientate organisms in their natural settings. These reactions can further alter behavioral patterns such as reproduction, foraging, migration, communication and predator–prey relationships (Longcore & Rich 2004). Such studies provide evidence that artificial lights often have adverse effects on organisms (Salmon 2003; Bird et al. 2004; Longcore & Rich 2004; Bourgeois et al. 2009; Kempenaers et al. 2010; Longcore 2010).

The threats of artificial night lights to biodiversity are rarely explored at a broad spatial scale. Previous studies were predominantly conducted at a local scale in field or laboratory settings (Witherington & Bjorndal 1991; Salmon et al. 1995b; Grigione & Mrykalo 2004). However, broader, regional spatial patterns of activities and processes that threaten the existence of species are important to examine, especially when management practices are applied at larger spatial scales, as is often the case in regional conservation planning for large marine and terrestrial mammals and reptiles (Watzold et al. 2006). Today, with our improved ability to estimate anthropogenic pressures and activities from advanced sources such as satellite imagery and remote sensing, we are able explore the impact of human-threats on species at various scales (Kerr & Ostrovsky 2003).

Few studies have used satellite night light data for the assessment of threats and impacts on species, biological or environmental factors. Of the limited studies, night light imagery has been used in conservation to derive an index for environmental sustainability (Sutton 2003), has been used to explore the temporal impact of light pollution on marine ecosystems (Aubrecht et al. 2010a) and has been incorporated into the management of protected areas (Aubrecht et al. 2010b). However, the effect of artificial light sources and the night environment has largely been neglected in reserve system or corridor designs (Bird et al. 2004; Longcore & Rich 2004). No studies, as far as we are aware, have explicitly examined the potential of using satellite night light imagery as a tool for examining the distribution of sea turtle nests and its further conservation application.

5.2.1 Sea turtles – threats and factors affecting nesting patterns

Sea turtle species *Caretta caretta* (Linneaus 1758, loggerhead turtle) and *Chelonia mydas* (Linneaus 1758, green turtle) are globally threatened (Calase & Margaritoulis 2010). Their worldwide conservation status underlines the importance of understanding factors that influence their distribution and vulnerability. Sea turtles display philopatry, where nesting turtles return to their original place of birth (Carr 1975; Bowen et al. 1994). This behavior is known to operate at a relatively coarse regional scale ~10 km – 50 km (Miller et al. 2003) and factors that drive nesting sea turtles within this coarse spatial-scale are poorly understood (Weishampel et al. 2003; Garcon et al. 2009).

One important factor that is known to affect sea turtle behavior is the presence of night lights. Ecologists have found artificial lights disrupt sea turtle behavior in two ways. First, night lights reduce the ability of sea turtle hatchlings to find the sea. Hatchlings are either attracted to the artificial light source or are disorientated (Salmon 2003; Tuxbury & Salmon 2005; Lorne & Salmon 2007; Kawamura et al. 2009). Disoriented turtle hatchlings may fail to find the sea, thereby reducing population viability (Lorne & Salmon 2007; McConnell et al. 2010).

Second, there is the poorly understood phenomenon of artificial beach-front lighting preventing turtles from nesting. Nesting females of *C. caretta* and *C. mydas* are deterred by artificial lighting (Witherington 1992; Salmon et al. 1995b; Witherington & Martin 2000; Bourgeois et al. 2009). The repellent effect could be dose dependent so that highly lit areas deter all nesting and poorly lit areas have a minor impact (Margaritoulis 1985; Witherington 1992). Most of these studies are on beach sites along the coast of Florida (Salmon et al. 1995b; Witherington & Martin, 2000; Salmon 2003; Weishampel et al. 2006; Aubrecht et al. 2010a). Sea turtle researchers along the coast of the Mediterranean Sea seldom investigate this relationship (Kaska et al. 2003; Aureggi et al. 2005) and very few studies have explored this issue at a regional or broad spatial scale. Overall, the relationship between night lights and its effect on sea turtle nesting is poorly understood.

Previous studies found that sea turtles nest in non-random patterns and their selection of nest site is influenced by specific factors (Mellanby et al. 1998; Weishampel et al. 2003). Besides night lights, variables that are considered to influence sea turtle nesting include: beach dimensions (Kikukawa et al. 1996; Mazaris et al. 2006), beach slope (Wood & Bjorndal 2000) sand characteristics (Le Vin et al. 1998; Kikukawa et al. 1999), beach nourishment (Brock et al. 2009), climate change (Van Houtan & Halley 2011), predation (Leighton et al. 2011), human settlements (Kikukawa et al. 1996) and coastal development such as seawalls (Rizkalla & Savage 2011). Understanding the impact of these variables on sea turtle nesting is important for setting spatial conservation priorities (Moilanen et al. 2009). In this paper we investigate whether night lights, as quantified using space-borne images, can be used to help predict the distribution of sea turtle nests and we discuss the potential application of this tool in future conservation applications. The major questions we test in this study are:

(1) Can night lights derived from satellite imagery help us explain the distribution of sea turtle nests?

(2) Do night lights remain important at predicting sea turtle nest activity when considering additional anthropogenic and environmental variables?

5.3 Materials and methods

5.3.1 Study area

The Mediterranean Sea coastline of Israel is ~190 km long and has a north–south orientation (with the exception of the Carmel and Haifa Bay; Schattner 1967; Fig. 5.1). The overall width of beaches in Israel is between 20 and 100 m, with wider areas at river mouths. Israel's southern beaches (south of Tel Aviv) are characterized by relatively wider, sandy beaches (compared with northern beaches) with transverse sand dune fields, which have formed behind the shore in the past 1000 years (Schattner 1967; Tsoar 2000). In comparison, northern beaches are generally narrower and bordered by aeolionite (kurkar) cliffs. There are 32 rivers and ephemeral streams that flow through this coastal stretch into the sea (Lichter et al. 2010) and tidal movements in Israel are limited to a range of 15–40 cm (Lichter et al. 2010). Rectangular spatial units along the Israeli coastline were designed to examine the relationship between turtle nesting sites, night lights and associated anthropogenic and environmental factors. A buffer of 500 m to the east and west of the coastline was constructed and 336 spatial units of 1 x 0.5 km were positioned in this space. The buffer was chosen to allow for longitudinal location errors, as sea turtle nest surveyors sometimes reported only the latitudes. The dimensions of the spatial unit were based on the resolution of available night light imagery and expert advice regarding nesting turtle behavior.

5.3.2. Sea turtle data

Sea turtle data for this study were provided by Israel's National Parks Authority (NPA). We used nesting data of the two sea turtle species, *C. caretta* and *C. mydas*, which nest on the Mediterranean beaches of Israel (Kuller 1999; Levy 2003). The annual number of sea turtle nests have been increasing exponential within the past two decades, however specific reasons for their increase are unknown (Levy 2011; see Supplementary material Fig. S5.1). Sea turtle surveys along the entire coast of Israel were performed by Israel's Nature and Parks Authority since 1993, during the turtle nesting season from May to August. At the start of the nesting season (May), surveys were conducted two or three times a week. During peak season (June–July), beaches were surveyed daily. Towards the end of the season (August), surveys were performed twice a week. For survey purposes, the Mediterranean coast of Israel was divided equally into seven survey sections. Beach sections from Herzliya to Tel Aviv (~8 km) were not surveyed due to high human population density and development. The beach sections were scanned at sunrise by Israel's Nature and Parks Authority rangers along with trained volunteers. Surveys were conducted with 4WD vehicles driven close to the water edge, with a minimum of two people searching from the windows. Turtle nests were identified by the sand tracks that the female turtle leaves behind after laying her eggs. The two turtle species can easily be identified via their large and unique imprints, nest depth and position on the sand. The nest position was recorded via Garmin GPS units. Turtle tracks that did not result in a nest (false crawl), but seem to clearly be a nesting attempt were also recorded. Hatchling emergence or success was not systematically recorded over the years.

We examined and mapped the turtle nest data using ArcGIS (ESRI 2011). We combined the two sea turtle species together due to their related choice of nesting beaches (Broderick & Godley 1996; Weishampel et al. 2003) and the low number of *C. mydas* turtle nests in our study (0.8% of all nests). We used two variables derived from the turtle nest surveys: (1) the total number of nests found in each spatial unit summed over 19 years (1993– 2011; Fig. 5.1a); (2) the occupancy (presence/absence) status of each spatial unit for turtle nests in each year and then summed over a 19 year period (1993–2011) – this will be referred to as turtle nest persistence (Fig. 5.1b). This was performed to limit influences from individual years (Fig. S5.1). When the total number of turtle nests was summed per spatial unit for this time frame, there was a mean of 9.63 ± 15.5, a median of 3.5 and a range from 0 to 169 individual turtle nests. Twenty-six percent of the surveyed spatial units in our study had no turtle nests (absences).

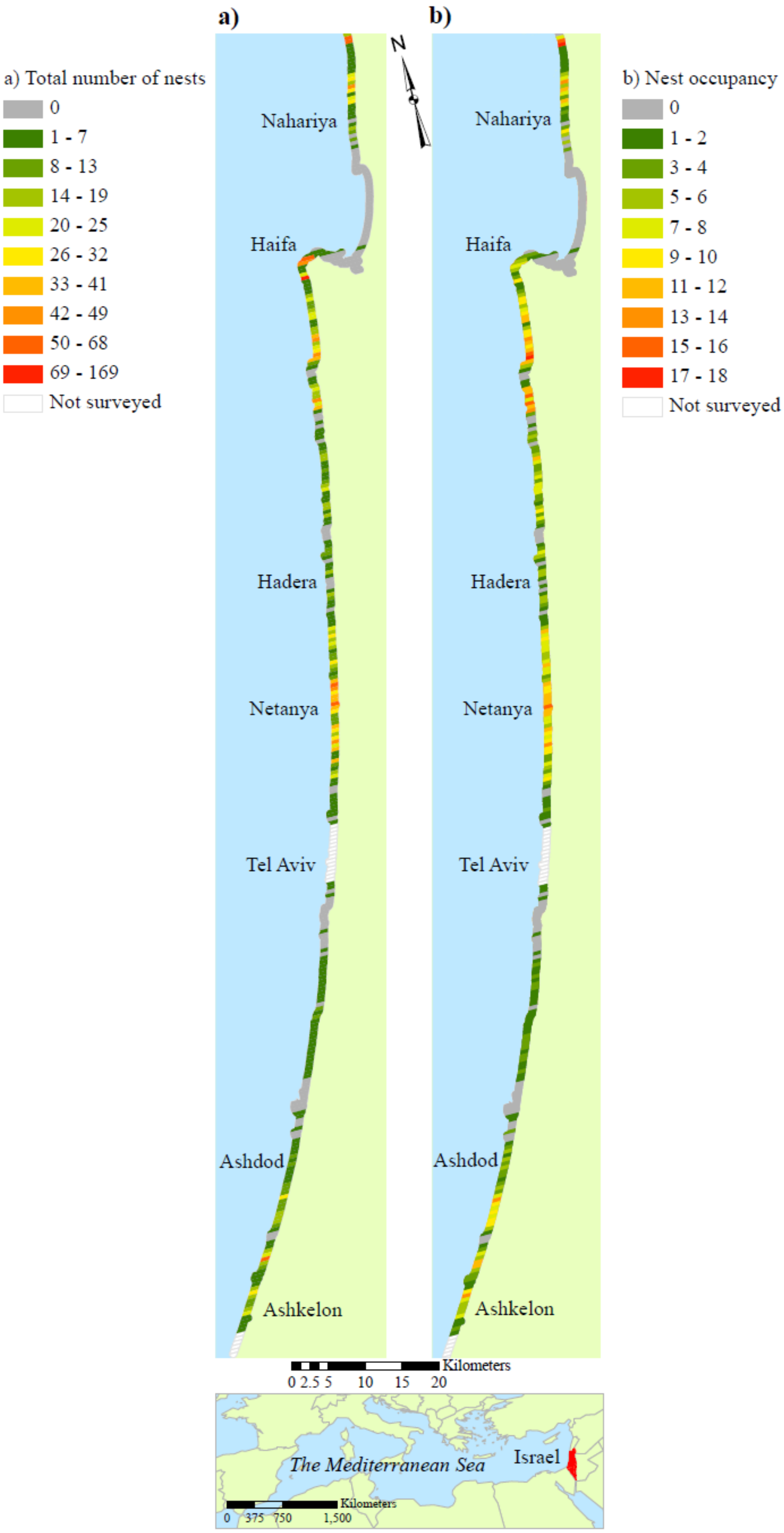


Figure 5.1. Map showing the study area along the Mediterranean coast of Israel, using the Israel Transverse Mercator Grid. (a) Total number of sea turtle nests summed from 1993 to 2011 within each spatial unit (1 x 0.5 km) along the coast of Israel; (b) sea turtle nest occupancy (presence/absence) was summed from 1993 to 2011 within each spatial unit. Israel's location within the Mediterranean basin is displayed at the bottom. The map was created with ESRI (2011) ArcGIS, Coastline: Survey of Israel, Turtle data: Israel Nature and Parks Authority.

5.3.3. Night light data

Two satellite images of the Israel coastline were used for this study, SAC-C (2007; 300 m) and ISS (2003; 60 m). We used a 2007 satellite image from Argentine's Space Agency (CONAE 2007) acquired by the High Sensitivity Technological Camera (HSTC) onboard the SAC-C satellite launched in 2000 (Fig. 5.2a). This image showed night lights at a spatial resolution of 300 m (Colomb et al. 2003) for the entire Israeli coastline. The SAC-C image underwent an inverse Fourier transformation to remove striping effects, using Idrisi Taiga (Clark Labs 2010; Levin & Duke 2012). Our second image, ISS, was from astronaut photography onboard the International Space Station (ISS mission 6). Imagery was obtained via Kodad DSC 760 camera at a resolution of 60 m in 2003 (Image Science and Analysis Laboratory 2003). The spatial extent of this image did not cover the entire Israeli coastline (missing data beyond Haifa) but was included due to the difficulty of obtaining high spatial resolution satellite images which covers the entire coastline of Israel. Night light data for 286 of the 336 spatial units were covered by the ISS image (Fig. 5.2b). For both satellite images we determined an average pixel brightness value for each spatial unit with ArcGIS tools (ESRI 2011).

5.3.4 Other explanatory variables

In addition to testing the importance of night lights at predicting turtle nesting patterns, we examined the effect of 21 additional variables that were hypothesized to affect sea turtle nesting and which were available for the full study region. These variables were divided into two groups; anthropogenic and environmental (see Table 5.1 for the full list of variables tested).

5.3.5 Statistical analysis

Our statistical analysis was designed to address our two major research questions;

5.3.5.1 Satellite night lights and sea turtle nests

We tested the ability of the two night light images to explain turtle nest distribution along the coast of Israel. Spearman's rank correlation coefficients were used to test for associations between turtle nest distribution and the average pixel values derived from the two night light images. To test our hypothesis that turtles prefer nesting in darker areas, we split our data into three night light intensity groups based on pixel values (high, moderate and low – each group with an equal number of spatial units) from both satellite images. The three groups were compared via the non-parametric Kruskal–Wallis one-way analysis of variance conducted in R software (R Development Core Team 2011). Quantile regression was used to further explore the relationship between sea turtle nests and night lights along the entire Israel coastline using the SAC-C image. Quantile regression was performed using the R quantreg package (Koenker 2007) with an exponential fit and bootstrapping for residuals.

5.3.5.2 The importance of satellite night lights

Here we examined the importance of night lights when considering other variables which may influence sea turtle nest distribution. We also aimed to construct models that predict: (1) the total number of nests per spatial unit and (2) turtle nest persistence, for the entire Israeli coastline with night lights (using the SAC-C image) and 21 broad scale explanatory variables (Table 5.1). We used generalized linear modeling (GLM) in R. GLMs simultaneously explore which variables and/or their interactions explain the highest amount of variability in turtle nest distribution. Prior to beginning the modeling procedure we tested for collinearity among the explanatory variables using Spearman rank correlations coefficient and Variance Inflation Factors (VIFs).We used a cut-off value of 3 for removing collinearity from the resulting VIFs (Zuur et al., 2007), and ±0.5 for Spearman's rank correlations coefficients between pairs of variables (Booth et al. 1994). For this analysis we used GLMs with a Poisson distribution, detected over dispersion and corrected the standard errors using quasi-GLMs (Zuur et al. 2009). Due to deviations in the coastline, the area of each spatial unit was not constant and therefore we performed our models with an offset variable for area (Zuur et al. 2009). Model simplification was conducted by dropping each explanatory variable in turn and removing the term that led to the smallest non-significant change in deviance according to F-tests (using the drop1 command in R; Zuur et al. 2009). Model validation was conducted using the deviance residuals plotted against the fitted residuals, explanatory variables and spatial coordinates. We also tested our raw data and models residuals for spatial auto-correlation using spline correlograms with 95% pointwise bootstrap confidence intervals and a maximum lag distance of 10 km (Bjørnstad & Falck 2001; Zuur et al. 2009).

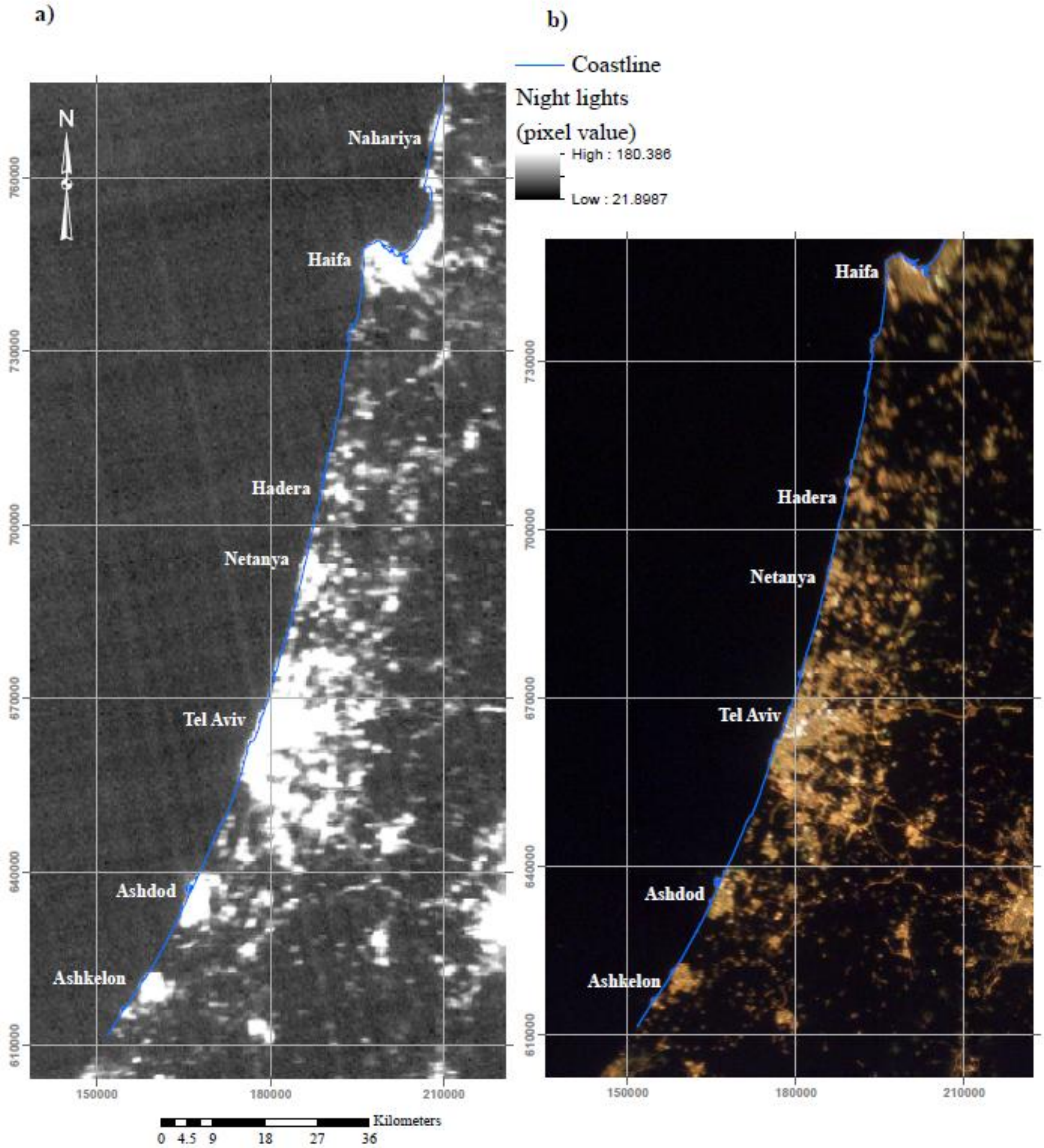


Figure 5.2. The satellite images used in this study for calculating night lights along the coast of Israel. Major cities are displayed. (a) SAC-C satellite from Argentine's Space Agency (CONAE 2007), pixel resolution is 300 m and covers the full extent of the Israeli coastline (full coverage). (b) Image from International Space Station astronaut photography, pixel resolution is 60 m (Image Science and Analysis Laboratory 2003), where northern coverage only extends to Haifa (partial coverage). The map was created with ESRI (2011) ArcGIS.

Table 5.1. Table displaying 21 variables used in this study (in GLM). Four anthropogenic based and 17 environmental variables were used that were suspected to be related to turtle nesting patterns (* = categorical variable).

Variables	**Data origin**
Anthropogenic based	
Human population density	Population density data were obtained as of 2007 for statistical units as defined by Israel Central Bureau of Statistics (CBS 2007). As a proxy for estimating the population residing near the beach, each spatial unit was given the population density of the closest municipality division alongside the coast.
Built-up areas (m)	Data for built up areas were available from the Israeli Ministry for Environmental Protection (Kaplan et al. 2006), within each spatial unit (CBS 2007). Built-up areas were calculated by the distance from the coastline (middle of spatial unit) to the closest built up area (m).
Infrastructure (m)	To determine the land-use type of the beach we used GIS data supplied by the Society for the Protection of Nature in Israel (SPNI) Open Landscape Institute (OLI). The distance (m) from the center of each spatial unit to beaches clear of national infrastructure (e.g., ports, roads, electrical grids, military areas) was measured.
Reserves	The current areas protected within nature reserves and national parks of Israel were provided by Israel's Nature and Parks Authority. The percentage of each rectangular unit that is protected by a reserve which is either officially declared or approved was calculated using ArcGIS (ESRI 2011). Reserves that are currently awaiting approval or recently proposed were not taken into consideration.
Environmental variables	
Beach area	We digitized the area of beach (sand area) from Google Earth (2011) satellite imagery, performed at the rectangular unit scale (500 m) in ArcGIS (ESRI 2011). We calculated the percentage of the spatial unit's area which was covered by beach.
Cliffs *	We included the presence and absence of cliffs bordering the shoreline of beaches as a categorical variable (1=cliffs, 0=no cliff). These data were provided by the Society for the Protection of Nature in Israel (SPNI) Open Landscape Institute (OLI).
Geomorphologic features	We used GIS data from a Geological Survey of Israel for the Ministry of Environment (Zilberman et al. 2006). Fifteen geomorphologic classes (Table S5.3) were considered in our analysis. We calculated the percentage of each geomorphologic feature within every rectangular unit.

5.4. Results

5.4.1 Satellite night lights and sea turtle nests

Night lights from the SAC-C image were negatively correlated with the total number of sea turtle nests (Spearman's rho = -0.31, p = 4.07e-09; Fig. 5.3a) and nest persistence (Spearman's rho = -0.34, p = 8.12e-11; Fig. 5.3b) across the Israel coastline. Comparison of the two satellite images when related to sea turtle nests indicated that the ISS image with the higher resolution gave only slightly more significant results compared to the SAC-C image (Table 5.2). We found that the total number of sea turtle nests (Kruskal– Wallis test, SAC-C p = 4.7e-0, ISS p = 1.01e-06; Fig. 5.4) and nest persistence (Kruskal–Wallis test, SAC-C p = 3.24e-08, ISS p = 1.28e-07; Fig. 5.5) within our spatial units were significantly different for the three groups of night light intensity. The mean rank of turtle nest numbers was highest in the low pixel group (mean SAC-C = 202.46; ISS = 173.82), which refers to darker sites, compared to the mean of the moderate (mean SAC-C = 169.91; ISS = 147.08) and high (mean SAC-C = 133.13; ISS = 111.42) groups for both satellite images. Similarly, for both satellite images the mean rank of turtle nest persistence was highest in the low pixel group (mean SAC-C = 206.50; ISS = 175.28), compared to moderate (mean SAC-C = 167.87; ISS = 148.40) and high (mean SACC = 131.13; ISS = 108.65) groups. Quantile regression showed that the 0.5 (median) and 0.75 quantiles were statistically significant for the relationship between night lights and sea turtle nests along the entire coastline of Israel (see Table S5.1).

Table 5.2. Spearman rank correlation coefficient of night lights (pixel values) from two satellite images with sea turtle nest persistence and the total number of sea turtle nests (summed over 19 year period within 336 spatial units) along the coast of Israel.

	Total number of sea turtle nests		**Sea turtle nest persistence**	
Satellite night light image	**Spearman's rank correlation coefficient**	***p***	**Spearman's rank correlation coefficient**	***p***
SAC-C (Entire Israel Mediterranean coast)	-0.31	4.07e-09	-0.34	8.12e-11
ISS (Partial coast)	-0.37	7.71e-11	-0.39	6.44e-12
SAC-C (Partial coast as used in ISS image)	-0.35	1.11e-09	-0.38	3.20e-11

5.4.2 The importance of satellite night lights

Night lights were found to be a significant explanatory variable for explaining the sea turtle nesting activity in both of our resulting GLMs (Table 5.3). Our resulting models were able to predict 18% (pseudo r^2) of the total number of sea turtle nests and 32% of sea turtle nest persistence within the spatial units along the entire coast of Israel. Of the 22 (including night lights) explanatory variable used in the modeling process, five variables were considered important for explaining the total number of sea turtle nests within our spatial units: night lights (F = 7.60, p = 0.01), cliffs (F = 26.22, p = 5.19e-07), the interaction between human population density and infrastructure (F = 10.22, p = 1.53e-03) and red sandy clay loam (F = 5.63, p = 0.02). Similar variables were considered significant for explaining sea turtle nest persistence, three two-way interactions made up our final model: the interaction between beach area and human population density (F = 4.91, p = 0.03), night lights and cliffs (F = 4.62, p = 0.03) and human population density and infrastructure (F = 5.57, p = 0.02; Table 5.3). The only explanatory variable showing signs of collinearity with night lights was built up areas along the coast (Spearman's rho = -0.61) however this variable was not significant in our models. We also found that the only interaction with night lights was the presence of cliffs in our model that explains sea turtle nest persistence. No spatial autocorrelation or collinearity (VIFs all below 3; Table S5.2) among our explanatory variables was found and our models met the validation requirements (Fig. S5.2; Fig. S5.3).

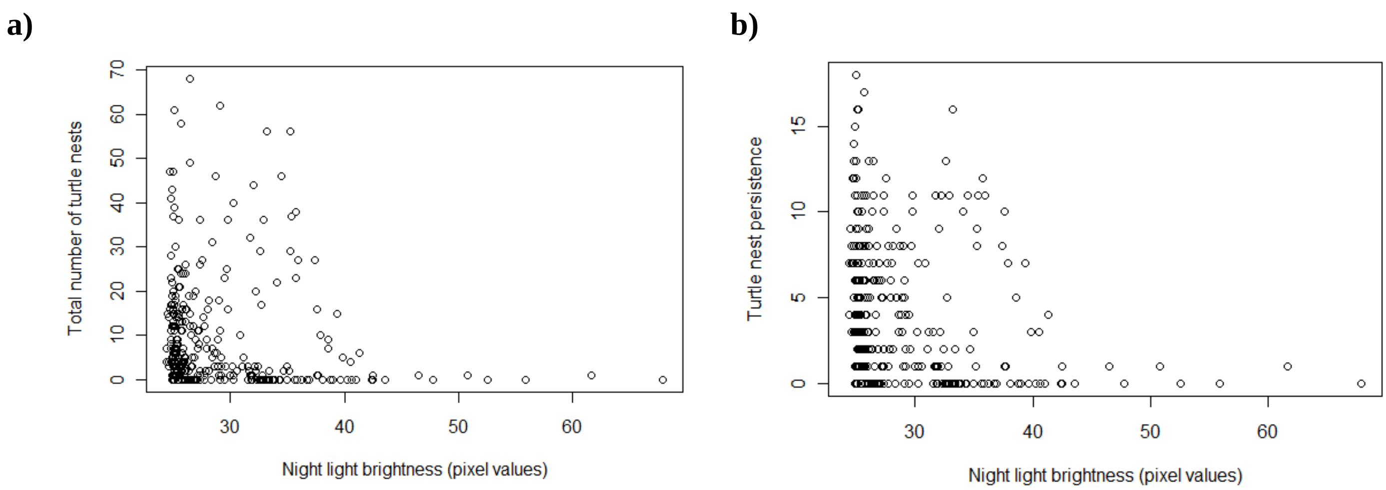


Figure 5.3. Scatter plot using spatial units (1 x 0.5 km) along the coast of Israel to show relationships between sea turtle nesting activity over a 19 years period (1993–2011) and night light intensity derived from a satellite image (SAC-C; CONAE 2007). One outlier was removed from the plot for visualization purposes. (a) Total number of sea turtle nests summed per spatial unit (1 x 0.5 km). (b) Sea turtle nesting persistence (presence/absences) summed over time period for each spatial unit.

a) b)

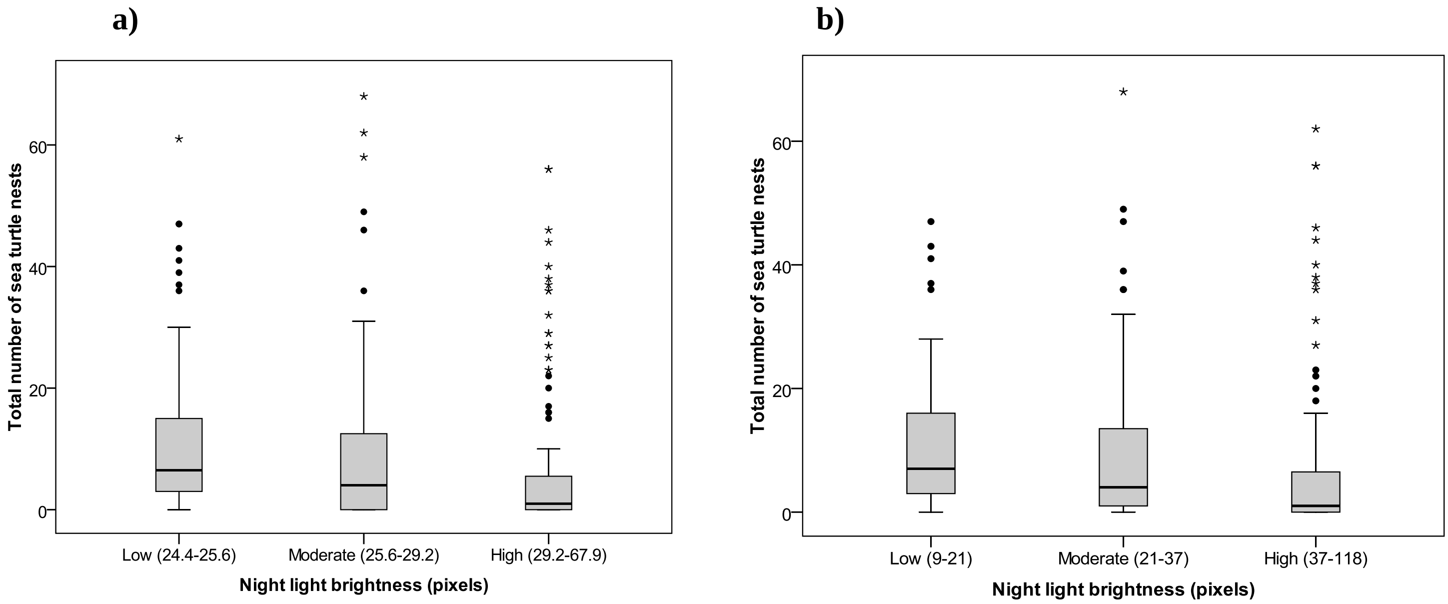


Figure 5.4. Box plots of Kruskal–Wallis one-way analysis of variance of three groups of night light intensity; high (well-lit areas), moderate, and low (dark areas) related to the total number of sea turtle nests occupancy (summed for years 1993–2011) along the coast of Israel. Pixel values of the three groups are in bracket. One outlier was removed from the plot for visualization purposes. (a) SAC-C satellite image (CONAE 2007), (b) ISS satellite image (Image Science and Analysis Laboratory 2003).

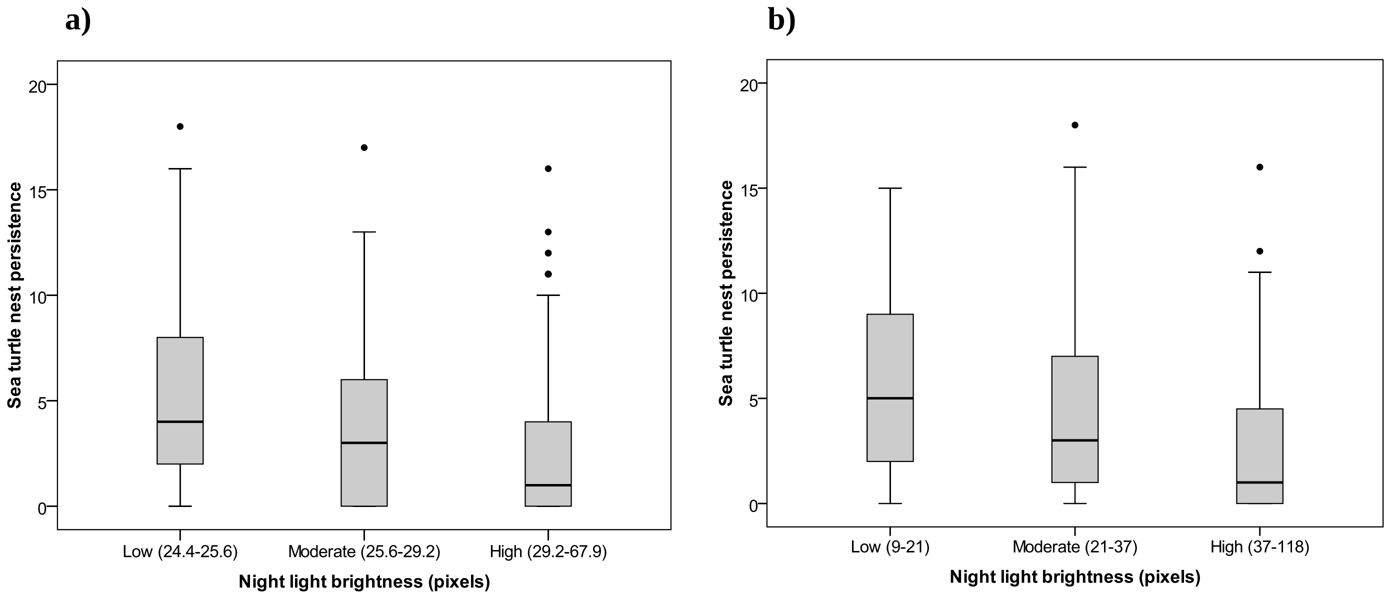


Figure 5.5. Box plots of Kruskal–Wallis one-way analysis of variance of three groups of night light intensity; high (well-lit areas), moderate, and low (dark areas) related to sea turtle nest occupancy (presences/absence) frequency (summed for the years 1993–2011) along the coast of Israel. Pixel values of the three groups are in brackets. One outlier was removed from the plot for visualization purposes. (a) SAC-C satellite image (CONAE 2007), (b) ISS satellite image (Image Science and Analysis Laboratory 2003).

Table 5.3. Minimum adequate quasi-Poisson GLM to explain sea turtle nest persistence and the total number of sea turtle nests (between 1993-2011) within spatial units along the entire coastline of Israel. See Table 5.1 for details regarding explanatory variables. Interactions between explanatory variables are marked with a cross. Rows with no values signify explanatory variables that were eliminated within the modelling process and did not contribute to the final model.

	Total number of nests							**Nest persistence**						
Explanatory variable	**Coefficient**	**SE**	***t***	***p***	***df***	***F***	***p***	**Coefficient**	**SE**	***t***	***p***	***df***	***F***	***p***
Night lights (SAC-C image) – negative exponential	3.34e+10	1.79e+10	1.87	0.06	1	7.60	0.01 **	6.39e+10	9.60e+09	6.66	1.18e-10 ***			
Cliffs	8.16e-01	2.30e-01	3.54	4.56e-04***	1	26.22	5.19e-07 ***	1.09e+00	1.67e-01	6.52	2.64e-10 ***			
Infrastructure	-2.44e-04	1.31e-04	-1.87	0.06				-3.88e-04	9.03e-05	-4.30	2.30e-05 ***			
Human population density	-4.06e-05	3.63e-05	-1.12	0.26				-9.10e-05	3.70e-05	-2.46	0.01 *			
Beach area								1.70e-02	7.81e-03	2.17	0.03 *			
Beach area x Human population density								1.62e-05	7.57e-06	2.14	0.03*	1	4.91	0.03 *
Night lights (neg exp) x Cliffs								-5.73e+10	2.85e+10	-2.01	0.04 *	1	4.62	0.03 *
Human population density x Infrastructure	-5.47e-07	4.96e-07	-1.10	0.27	1	10.22	1.53e-03 **	-2.80e-07	1.81e-07	-1.54	0.12	1	5.57	0.02 *
Red sandy clay loam (Geo_2)	-1.8e-02	1.28e-02	-1.46	0.15	1	5.63	0.02 *							

Statistical Significance: * - 0.05, ** - 0.01, *** 0.001

5.5 Discussion

This study demonstrates a novel application of satellite night light imagery to help predict nesting activity of Endangered sea turtles. While the impact of artificial night lights on biodiversity is often overlooked, we found that the intensity of coastal night lights derived from satellite-imagery is a significant determinant of sea turtle nest distribution. Results from our GLMs indicated that night light intensity remained an important predictor of sea turtle nest distribution when other anthropogenic and environmental factors were considered. For threatened species with large scale spatial movement such as sea turtles, where factors that influence their selection of nesting sites are largely unknown, improving our ability to determine their nesting patterns can enable us to better direct and target our conservation efforts.

This is one of the first studies to explore the relationship between nesting sea turtles and night lights at a regional spatial scale. Our results indicate that the intensity of artificial night lights along the Mediterranean coastline of Israel affects sea turtle nesting patterns, where well lit beaches have lower occurrences of nesting turtles. These large scale findings are supported by localscale studies that show nesting is influenced by night light intensity (Margaritoulis 1985; Witherington 1992). Thus, our broad scale study provides support for the hypothesis that sea turtles prefer darker beach sites for nesting. By utilizing information derived from satellite night light imagery we can explore broader spatial patterns between species and the night environment which were previously spatially restrictive. Our results suggest that night lights derived from satellite-based images provide a useful tool for assessing broad-scale spatial patterns of sea turtle nest sites.

In addition to artificial night lights, we identified other new and significant variables and their interactions that help predict sea turtle nesting activity at a broad spatial scale. The significant predictors found in both our GLMs, besides night lights, were the presence of cliffs (positive effect), human population density (negative effect) and infrastructure (negative effect). Although we were limited with the inclusion of explanatory variables due to data availability at this broad scale, we found new and unexplored explanatory variables that influence sea turtle nesting. This is the first study to find that the presence of coastal cliffs have an important positive influence on sea turtle nests. Findings by Kikukawa et al. (1999) indicated that beach height is an important variable, and Salmon et al. (1995a) found a positive correlation with tall objects along the shoreline, however to our knowledge, no studies have explicitly explored the effect of cliffs. While cliffs were a positive effect on sea turtle nests in our study, we suggest that there may be negative effects in some countries with large tidal ranges or areas where sea levels are beginning to rise (Fish et al. 2005). In such areas the presences of cliffs may cause a barrier for nesting turtles, where the landward movements of nesting turtles are restricted, thus a potential cause of nest destruction by sea water inundation (Fish et al. 2005). We recommend further investigation of other beaches with cliffs around the Mediterranean to better understand the effect that coastal cliffs have on sea turtle nests and its further application for conservation. Hence, at this broad scale we were able to identify variables that influence sea turtle nesting, which is particularly important to consider in conservation management when very little is known about their spatial distribution.

Night lights and cliffs as individual components have an important effect on sea turtle nests and combined have an important positive interaction effect (Table 5.3). This is exemplified by the case of Netanya (Fig. 5.2), a coastal city in Israel where beaches have a high number of sea turtle nests, shoreline cliffs and bright night lights. This interaction should be further explored in small-scale field studies to understand the nature of this relationship and the impact that cliffs near coastal cities exhibit on nesting sea turtles. Beach areas with bright night lights and beach cliffs may be prime areas to focus conservation efforts for the recovery of nesting sea turtle populations.

Anthropogenic based variables may be useful for predicting species distribution and activity within highly modified environments such as the coastal zone. In previous studies at local scales, environmental variables have been predominantly used for determining sea turtle nesting activity (Wood & Bjorndal 2000; Karavas et al. 2005; Mazaris et al. 2006). However, findings from our study suggest that human based variables were important. Other studies which have included human based variables have also found that sea turtle nests were negatively influenced by such factors. For example, Weishampel et al. (2003) found that nests of green and loggerhead sea turtles increased as the density of human development was lower along beaches in east Florida. A multiple regression approach by Kikukawa et al. (1999) also found that loggerhead sea turtle nests in Okinawajima, Japan, significantly increased with distance from human settlements. We suggest that today with the increasing number of anthropogenic threats on the coastal environment that inclusion of human based factors may serve as helpful predictors of sea turtle nesting patterns or other coastal species.

Artificial night lights may pose a greater threat to sea turtle nests compared with other anthropogenic threats. Our GLM results showed that night lights were more significant at explaining sea turtle nests distribution then other anthropogenic threats such as the human population density, infrastructure and built up areas. Unlike these other variables, night lights account for the presence of most human night time activity, including beach side restaurants, shopping districts, ports and residential areas. Interestingly, we also found that higher resolution satellite night light imagery, comparison between the ISS and SCC-C images, was better related to sea turtle nesting patterns (Table 5.2). Thus, the threat of night lights on sea turtle nesting, while evident from laboratory and small-scale field experiments (Witherington 1992; Salmon et al. 1995b) can also be explored with the use of high resolution satellite imagery.

To date, very few explanatory variables and models have been identified which can aid our understanding of nesting patterns of threatened sea turtle species (Garcon et al. 2009). Clearly there are additional unknown factors which affect sea turtle nest distribution. Our resulting models were able to explain 18% and 32% of turtle nest variance. These values suggest that there are other factors which contribute to predicting sea turtle nest distribution. Other contributing factors could be related to the hypothesis that sea turtles use multiple environmental factors/cues with thresholds to reach before choosing a nesting site (Wood & Bjorndal 2000; Mazaris et al. 2006). Alternatively, these factors could be due to recently explored climatic factors, predation, other anthropogenic threats, interactions among variables (Leighton et al. 2011; Rizkalla & Savage 2011; Van Houtan & Halley 2011) or small scale environmental conditions that are not found at this large scale (Wood & Bjorndal 2000). Thus, with the little knowledge we have on sea turtle nesting patterns, combined with their threatened status, we propose that satellite night light imagery may be a useful tool for the prediction of sea turtle nest distribution at a broad spatial scale and recommend its incorporation into future studies.

5.5.1 Conservation implications

The advancements in spatial analysis and applications (Sen et al. 2006) continually allow us to consider new techniques and methods to explore and predict species assemblages and patterns at broader spatial scales with higher resolution (Kerr & Ostrovsky 2003; Turner et al. 2003). In recent years studies have been quantifying biodiversity with remote sensing tools and satellite imagery (Levin et al. 2007; Lahoz-Monfort et al. 2010; Rocchini et al. 2010; Bradter et al. 2011). While such tools and methods cannot replace field work at smaller scales, they can serve as useful tools for exploring larger spatial-scales. In particular circumstances where field work locations are inaccessible or spatial extents are too large, remote sensing can provide us with the best knowledge at hand. Further research therefore, should be conducted with these tools at broader spatial scales and regional levels in order to advance our understanding of species habitat selection, movement and threats. Predicting species habitats, movements and identifying their threats can greatly aid conservation decisions, which are often made with relatively sparse information (Pressey 2004). While this study examines nesting sea turtles, the same methodology can be applied to other species that are disturbed by artificial night lights. For such species, we propose that satellite night light imagery can be incorporated into conservation planning in order to mitigate the threat of night lights when selecting priority conservation areas or reserves. This approach is especially relevant for rare and threatened species such as sea turtles, for which there is a limited time to act in the face of increasing human-pressures and where action is needed at broad scales.

5.6 Supplementary material

Table S5.1. Quantile regression of night lights (SAC-C image) and sea turtle nest activity from 1993-2011.

Total number of nests				
Quantile	value	SE	*t*	*p*
95	-0.01	0.03	-0.39	0.79
90	3.35e-3	0.03	0.1	0.91
75	-0.07	0.04	-2.09	0.04*
50	-0.37	0.15	-2.42	0.02*
Nest persistence				
Quantile	value	SE	*t*	*p*
95	-0.02	0.01	-1.68	0.09
90	-0.02	0.02	-1.50	0.13
75	-0.06	0.03	-2.36	0.02*
50	-0.31	0.14	-2.29	0.02*

Table S5.2. Variance inflation factors calculated in R to test collinearity amongst twenty-two explanatory variables within GLMs (Zuur et al. 2009).

Explanatory Variable	**Variance inflation factors (VIFs)**
Night lights (pixel value from SAC-C image)	1.40
Human population density	1.17
Beach area	1.77
Built-up areas	2.09
Infrastructure	1.72
Cliff presence	1.25
Reserves	1.23
Geo_1	1.22
Geo_2	1.18
Geo_3	1.08
Geo_4	1.23
Geo_5	1.11
Geo_6	1.26
Geo_7	1.33
Geo_8	1.06
Geo_9	1.20
Geo_10	1.20
Geo_11	1.26
Geo_12	1.15
Geo_13	1.33
Geo_14	1.06
Geo_15	1.05

Table S5.3. Table of 15 geological classes defined in the geomorphologic map of Israel's beaches. GIS data were collected from the Geological Survey of Israel for the Ministry of Environmental (Zilberman et al. 2006).

Substrate code	**Description**
Geo_1	Aeolionite (kurkar)
Geo_2	Red sandy clay loam (Hamra soil)
Geo_3	Holocene aeolionite (kurkar)
Geo_4	Stabilised brown sand mixed with archeological remnants.
Geo_5	Stabilized dunes
Geo_6	Stabilized inter-dune sand
Geo_7	Active sand dune חול מנושב
Geo_8	Archeological sites
Geo_9	Tidal beach area (swash area)
Geo_10	Beach rocks
Geo_11	Aeolionite (kurkar) tables (near the water surface)
Geo_12	Rivers and drainage canals
Geo_13	Clay soils
Geo_14	Alluvial soils
Geo_15	Construction and industrial waste

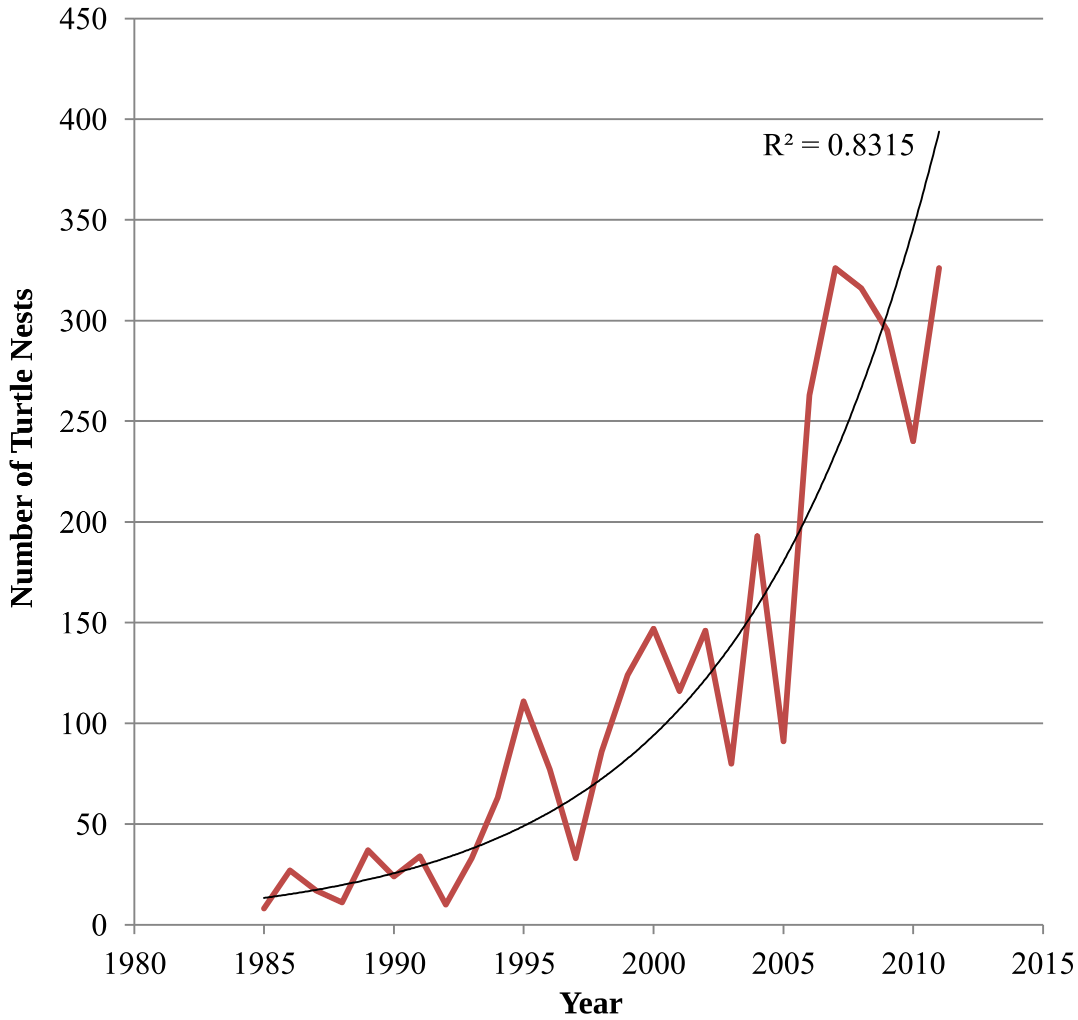


Figure S5.1. Number of yearly sea turtle nests along the coast of Israel from 1993 to 2011 (Levy 2011).

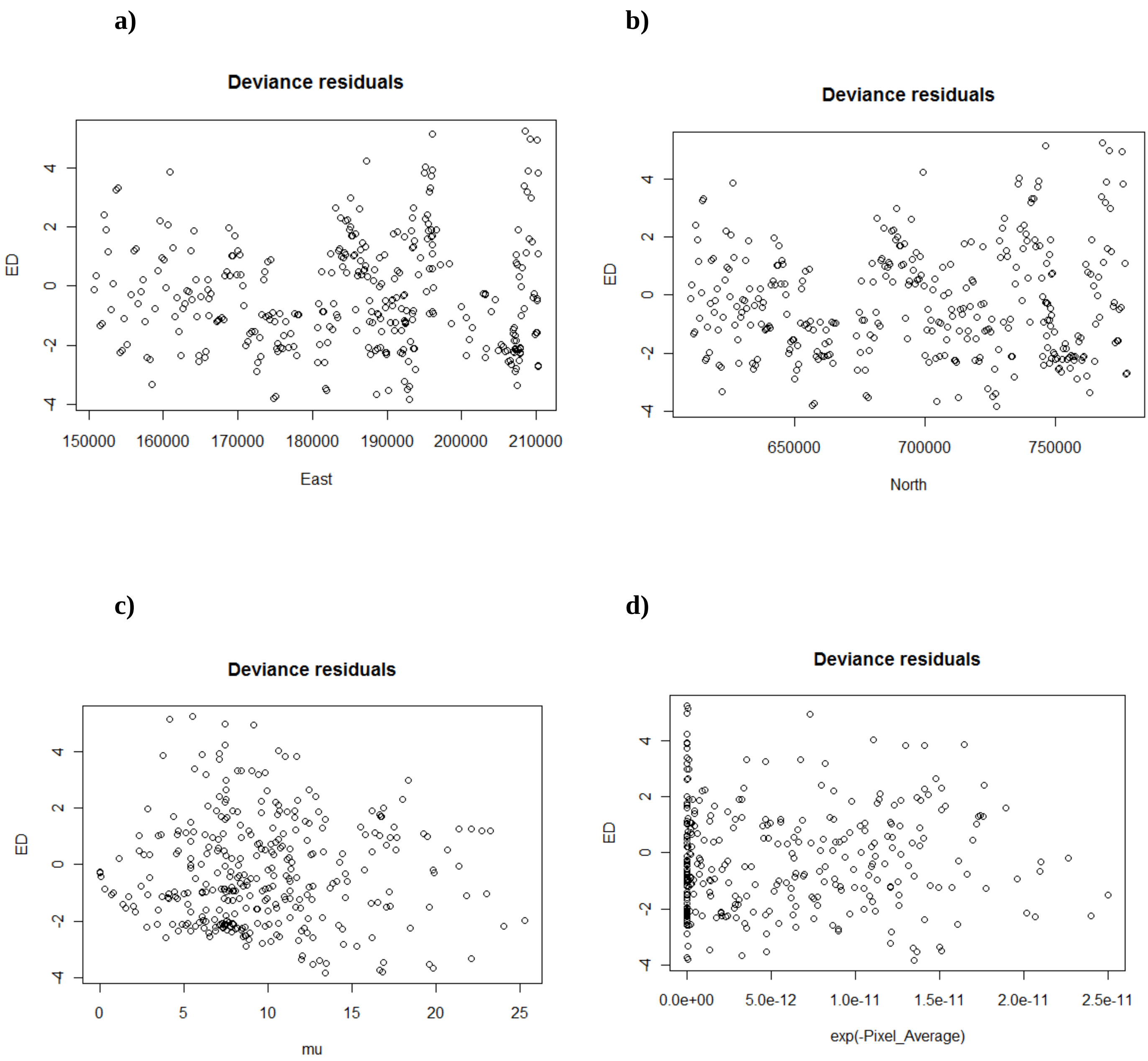


Figure S5.2. Model validation for GLM explaining total sea turtle nest numbers. a) deviance residuals against eastings of each spatial unit b) deviance residuals against northings of each spatial unit c) deviance residuals applied on optimal quasi-Poisson model d) deviance residuals against night lights (pixel value).

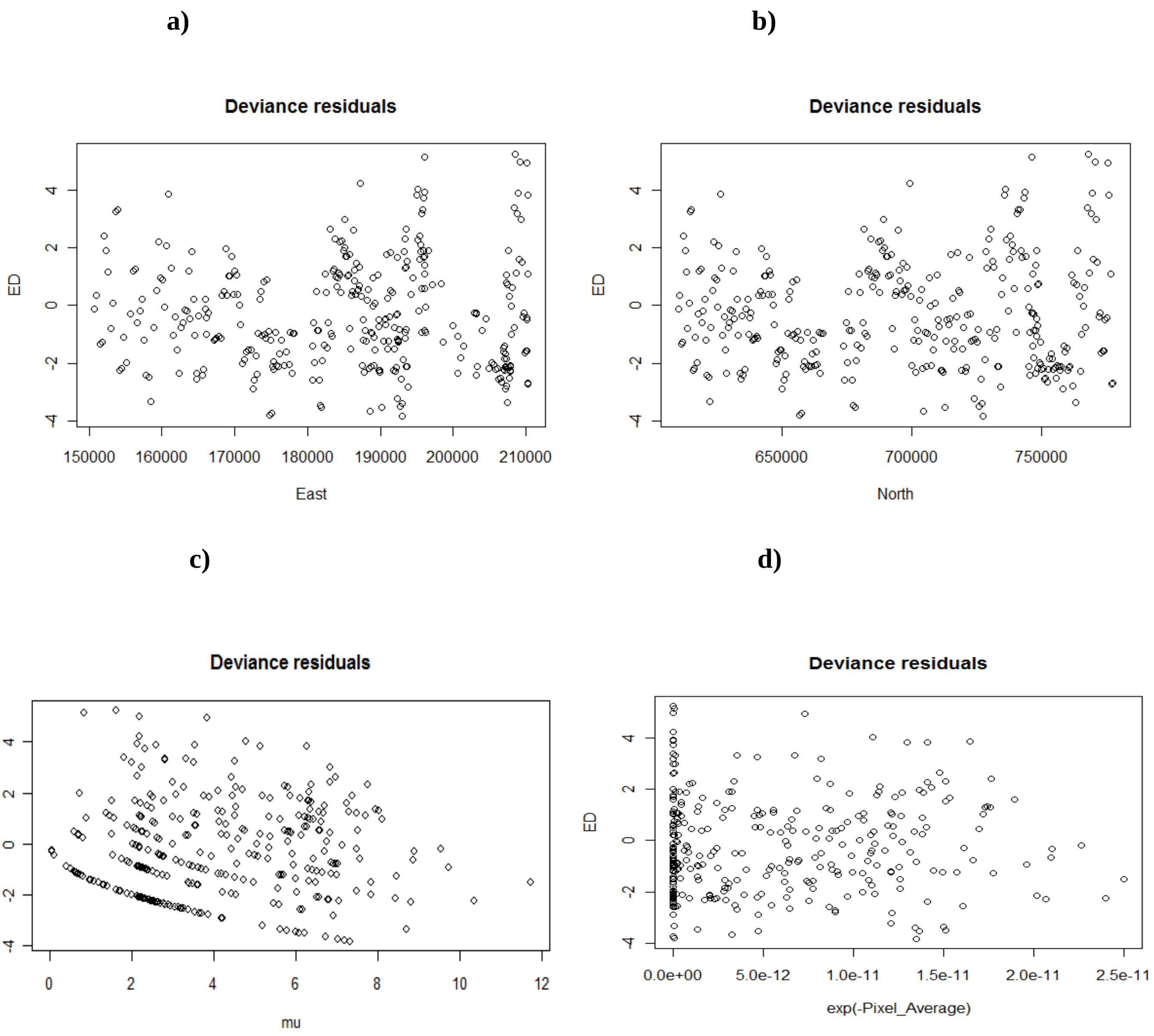


Figure S5.3. Model validation for GLM explaining sea turtle nest persistence a) deviance residuals against eastings of each spatial unit b) deviance residuals against northings of each spatial unit c) deviance residuals applied on optimal quasi-Poisson model d) deviance residuals against night lights (pixel value).

Chapter 6

The Crowded Sea: Incorporating multiple marine activities in conservation plans can significantly alter spatial priorities

Tamar gas field, Israel. Photo credit: Getty images

PLoS ONE (2014) **9**, e104489.

6.1 Abstract

Successful implementation of marine conservation plans is largely inhibited by inadequate consideration of the broader social and economic context within which conservation operates. Marine waters and their biodiversity are shared by a host of stakeholders, such as commercial fishers, recreational users and offshore developers. Hence, to improve implementation success of conservation plans, we must incorporate other marine activities while explicitly examining trade-offs that may be required. In this study, we test how the inclusion of multiple marine activities can shape conservation plans. We used the entire Mediterranean territorial waters of Israel as a case study to compare four planning scenarios with increasing levels of complexity, where additional zones, threats and activities were added (e.g., commercial fisheries, hydrocarbon exploration interests, aquaculture, and shipping lanes). We applied the marine zoning decision support tool Marxan to each planning scenario and tested a) the ability of each scenario to reach biodiversity targets, b) the change in opportunity cost and c) the alteration of spatial conservation priorities. We found that by including increasing numbers of marine activities and zones in the planning process, greater compromises are required to reach conservation objectives. Complex plans with more activities incurred greater opportunity cost and did not reach biodiversity targets as easily as simplified plans with less marine activities. We discovered that including hydrocarbon data in the planning process significantly alters spatial priorities. For the territorial waters of Israel we found that in order to protect at least 10% of the distribution range of 166 marine biodiversity features there would be a loss of ~15% of annual commercial fishery revenue and ~5% of prospective hydrocarbon revenue. This case study follows an illustrated framework for adopting a transparent systematic process to balance biodiversity goals and economic considerations within a country's territorial waters.

6.2 Introduction

Implementing marine conservation plans is a major challenge. Plans that determine priority areas for conservation are often based solely on biological and ecological information (Knight & Cowling 2007). One of the main factors inhibiting the uptake of marine conservation plans by decision makers is inadequate consideration of the broader social and economic context within which conservation operates (Knight et al. 2008; Weeks et al. 2010a; Biggs et al. 2011). Marine waters and their biodiversity are shared by a host of stakeholders and interest groups, such as commercial fishers, recreational users and offshore developers (Douvere 2008). Inclusion of the activities of these multiple marine users within conservation plans is critical for achieving plans which are realistic and achievable in the real world, thereby moving from paper to action (Knight et al. 2008).

Conservation planners must try to explicitly consider other marine activities within conservation plans, to ensure no time is wasted over trying to conserve areas essential for other uses (Naidoo et al. 2006). Competition for ocean space is becoming increasing intensified as resource extraction and developments are expanding to include the marine realm (Norse 2008). Offshore activities such as commercial fishing, aquaculture facilities, sand mining, desalination plants, offshore wind farms and offshore power plants, provide countries with substantial economic gains (Douvere 2008). Currently, hydrocarbon operations are one of the largest economic stakeholders in the sea (Butt et al. 2013), and provide countries with huge potential and realized monetary benefits, and are expected to increase economic and political independence (Shaffer 2011; Tagliapietra 2013). However, incorporation of such economic activities is often absent from marine conservation planning literature. Despite the little willingness for countries to protect marine areas that are deemed economically important (Douvere & Ehler 2009), excluding other marine activities in conservation planning means we may not be able to design a marine reserve network that is representative or economically viable (Barr & Possingham 2013).

Disregarding other marine activities in marine conservation planning may also mean that anthropogenic threats to biodiversity are being ignored. When planning marine reserves that aim to reap sustainable long-term benefits it is important to examine the threats to biodiversity of the system that could impair this goal. However, reserve planning should not be solely based upon threat data (Pressey & Bottrill 2008). Examples of threats to biodiversity for consideration in reserve planning include: shipping lanes which pose a collision risk to marine mammals (Redfern et al. 2013), trawlers and demersal longliners which are damaging to benthic environments and responsible for the majority of annual sea turtles deaths via by-catch (Casale 2011), and marine energy installations which have been linked to habitat loss, noise pollution and invasive species (Inger et al. 2009). In some cases marine users have made changes or modifications, such as altering the path of shipping lanes for cetaceans (NOAA 2012). However, in cases where compromises cannot be met, conservation planners must be able to incorporate the potential threats to biodiversity into the planning process.

A common misconception is that marine zoning itself is a conservation planning tool. Marine zoning is the allocation of particular activities to specified marine areas (Douvere 2008; Agardy 2010). This practice can help reduce user conflict by separating incompatible activities (Norse 2008; Ehler & Douvere 2009; Agardy 2010). Several countries have stepped up to implement zoning strategies for their waters, the largest and perhaps most successful example of marine zoning is the Great Barrier Reef Marine Park off the coast of Queensland, Australia (Day 2002; Fernandes et al. 2005). More recent zoning efforts occurring around the globe include the United Kingdom Irish Sea Pilot (Boyes et al. 2007), the Belgian Exclusive Economic Zone (Douvere & Ehler 2009), the waters of Norway (Agardy 2010), Australia's entire commonwealth waters (DSEWPC 2013) and the zoning of China's territorial sea (Cao & Wong 2007). However, key elements are often missing from some zoning plans to ensure biodiversity goals are met. For marine zoning to be used as an appropriate method or tool for protecting marine biodiversity it must enable an explicit consideration of the trade-off between biodiversity and socio-economic objectives (Klein et al. 2009). Furthermore, zoning plans need to ensure that the zoning system provides protection that is representative of as many biodiversity features as possible (Klein et al. 2009; Barr & Possingham 2013).

The concept of including other activities within marine conservation planning is slowly emerging. Unlike marine spatial planning (MSP) which aims to plan water spaces to meet objectives of multiple marine users and stakeholders, (Ehler & Douvere 2009; Foley et al. 2010), marine conservation planning (MCP) is centred on one primary goal - achieving biodiversity protection (Agardy 2010). Recently, several systematic conservation plans in the marine realm have focused on a hybrid approach; reaching conservation objectives while also minimizing the opportunity cost to fishery stakeholders (Klein et al. 2009; Watts et al. 2009; Grantham et al. 2013). However, only some of these plans have been expanded to other social and economic contexts (e.g., Agostini et al. 2010; Weeks et al. 2010a). Facilitating the inclusion of other activities into marine conservation planning is the emerging development of zoning software that enables multiple objectives to be considered (e.g., Marxan with Zones; Watts et al. 2009). Up to now there has been little application of these new tools to address the complexity of marine conservation planning at regional scales or an entire country scale. As many countries around the globe aim to implement conservation measures by zoning their waters (Agardy 2010), it is important to develop an explicit zoning process which integrates the current spatial occupancy of other marine activities and where possible their economic objectives. The inclusion of other marine uses in marine conservation planning means that we need to carefully consider the trade-offs that underpin the resulting conservation plans and ensure that biodiversity goals are adequately achieved.

In this study we follow a framework (Fig. 6.1) using a systematic approach for zoning territorial waters to achieve the protection of marine biodiversity in the face of multiple anthropogenic threats and economic activities. Within this context, we aim to test how increased complexity (by the inclusion of zones, multiple activities and economic factors) in marine conservation planning alters: a) the ability to reach biodiversity targets, b) the opportunity cost, and c) the spatial conservation priorities. Furthermore, we aim to examine the explicit incorporation of prospective hydrocarbon extraction into marine conservation planning (Shaffer 2011).

6.3 Methods

Here our methods follow the steps outlined in Figure 6.1.

6.3.1 Spatial setting and study area

As a case study, we examined Israel's complete Mediterranean territorial waters. Israel is located in the eastern Mediterranean Sea and has relatively small territorial waters (~4200 km^2) compared with other coastal countries around the world. Currently, it faces rapid exploitation of its marine resources and aims to expand its protection of marine biodiversity (European Commission 2011). Israel's Mediterranean Sea territorial waters are defined by the National Planning Authority of Israel and are used by The Israel Nature and Parks Authority (NPA) for marine reserve planning. The territorial waters of Israel's Mediterranean Sea spreads along a coastline ~190 km long, and extends outwards for 12 nautical miles from the coast to a depth of ~1000 m, covering an area of ~4200 km^2 (European Commission 2011). For our analyses, we divided this study area into 1 x 1 km planning units, resulting in a total of 4,205 planning units.

6.3.2 Compiling biodiversity features

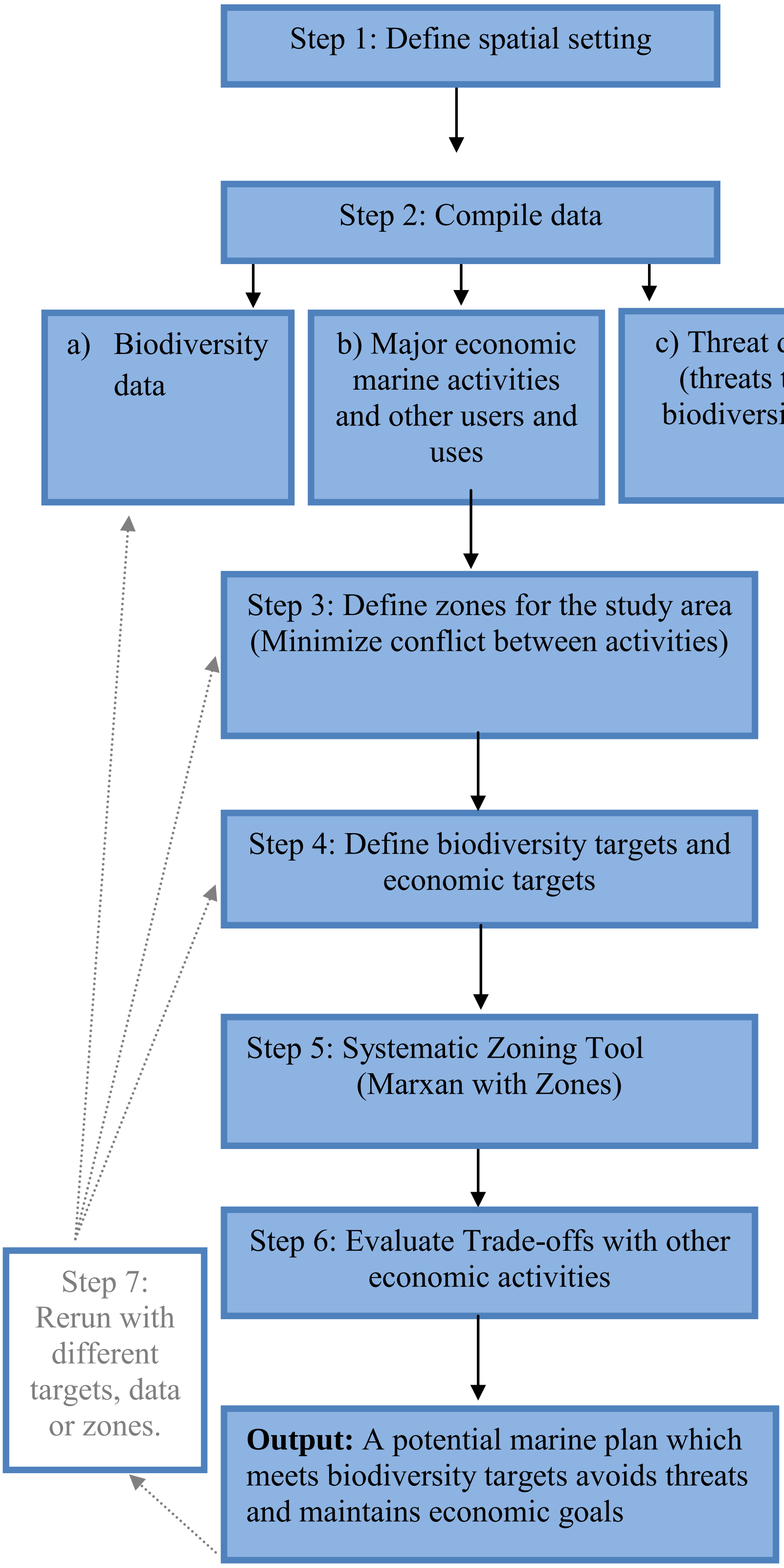
In order to select marine areas which will fulfil a representative reserve network where all types of biodiversity are protected we compiled available distribution data of Israel's Mediterranean territorial waters of biotic and abiotic features. These included 166 biodiversity features, comprising of vertebrate marine species (153 fishes, 2 turtles, 1 cetacean), and 10 geomorphologic features (Fig. 6.2a; see Table S6.1 for a list of species and features included in this study).

6.3.2.1 Marine species distribution data

We compiled data from currently available published studies on native cartilaginous and bony fishes whose distribution lies within Israel's Mediterranean waters (Diamant et al. 1986; Spanier et al. 1989; Goren & Galil 2001; Golani et al. 2007; Stern 2010; Levit 2012; Lipsky 2012; Edelist 2013). All native (non-alien) fish species (153 species) present in these publications were included in our study. We digitized the documented depth ranges of these native fish species using ArcGIS (ESRI 2008; Fig. S6.1; Table S6.2) and sea floor bathymetry (Amante & Eakins 2009), following methods in Tognelli et al. (2005) and Clark and Tittensor (2010). We derived the distributions via a number of sources; locations and depth ranges from the above eight studies, data from the Hebrew University of Jerusalem Fish Collection (accessed 2012), ranges as documented in Golani et al. (2006), and by expert opinion (for further details see Appendix S6.2).

The distribution of sea turtles within Israel's marine waters has not been well documented and their preferred feeding, foraging and mating areas are currently poorly known. Therefore, we used the locations of established nesting sites (within Mazor et al. 2013a) in Israel for both the green (*Chelonia mydas*) and loggerhead (*Caretta caretta*) sea turtle species. The targeted nesting habitats for protection in this study were chosen as planning units adjacent to nesting beaches with over 20 nest counts (from 1993-2011) and a persistence of more than five years of nesting at a particular site, in accordance with expert opinion from rangers and scientists at Israel's Nature and Parks Authority and Sea Turtle Rescue Centre.

We included the distribution of the common bottlenose dolphin (*Tursiops truncatus*), the most common cetacean species in Israel's territorial waters. Other cetacean species exist in Israel's waters but not enough observational data exists to determine priority habitats for these species. The common bottlenose dolphin has been sighted throughout Israel's territorial waters, therefore to better direct our conservation efforts we have considered important habitat areas as the species distribution. Scheinin (2010) identifies three core areas for feeding and foraging, an area at a depth of 40-50 m near Ashkelon, an area at a depth of 30-60 m between Ashdod and Palmachim beaches and another area off the coast of Netanya at a depth of 90-120 m. These three core habitat areas cover in total 213.64 km^2 (for additional information see Appendix S6.2).



Case Study:

Territorial waters of Israel's Mediterranean Sea

a) Total of 166 species (marine fishes, sea turtles and marine mammals; Fig. 6.2a).
b) Major: Offshore oil and gas exploration and commercial fishing (4 types of fisheries) Other activities: diving, military areas, desalination plants, aquaculture
c) Shipping lanes, pipelines, desalination plants, aquaculture, fishing, oil and gas exploration *some of these overlap with step 3

Four zones (see Table 6.1).

Biodiversity target set according to IUCN status & CBD target. No economic targets were set.

Freely available software (http://www.uq.edu.au/marxan/)

The trade-offs between fishery grounds, hydrocarbon exploration and biodiversity targets were explored.

Four planning scenarios were tested by including different marine uses and threats (Table 6.2).

Figure 6.1. Proposed framework for incorporating multiple activities and threats into marine conservation planning. These show the steps followed in the case study presented in this paper that encompasses Israel's entire Mediterranean territorial waters.

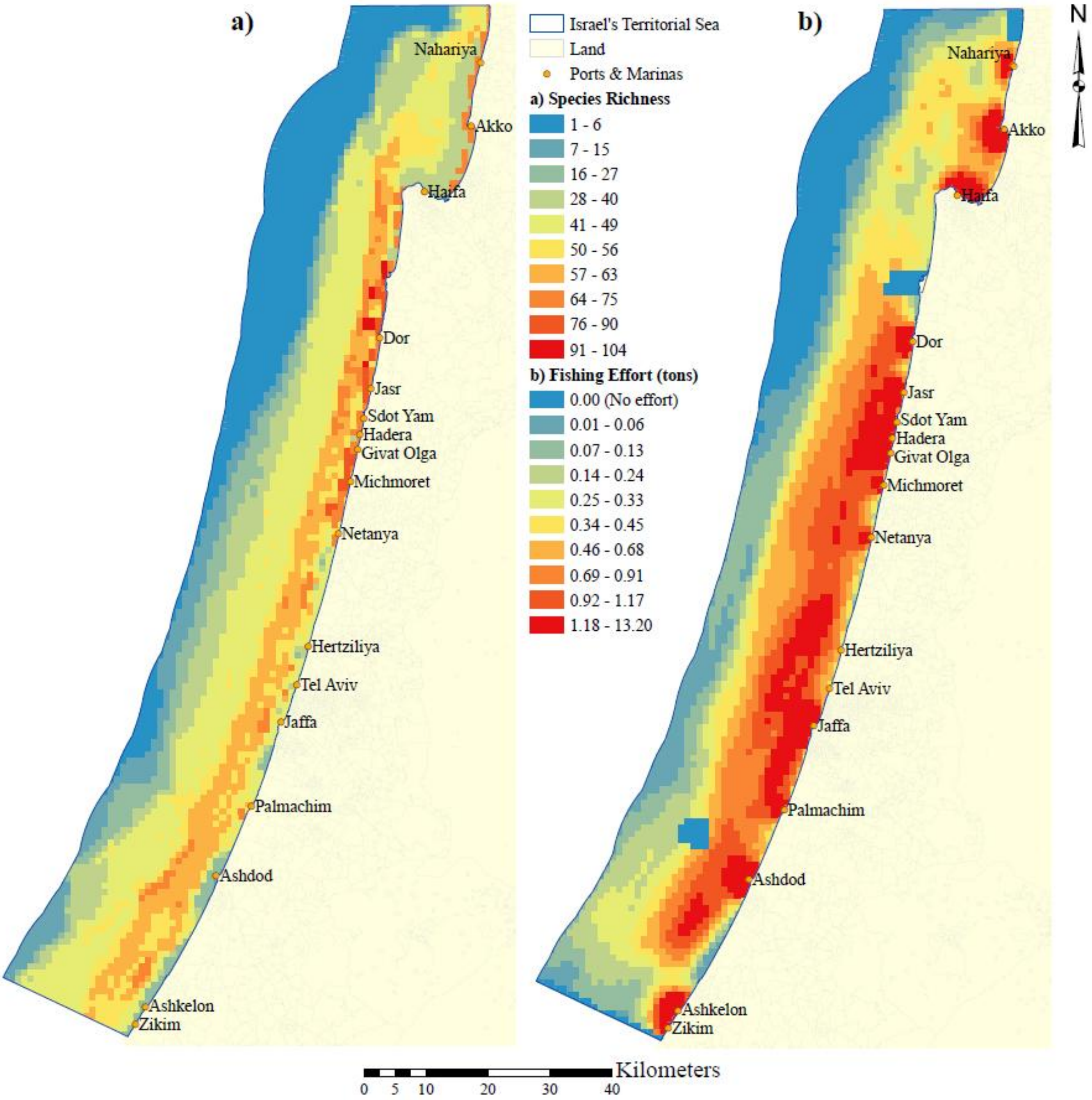


Figure 6.2. Biodiversity features and fishing effort in Israel's Mediterranean Sea territorial waters; a) species richness of 166 biodiversity features (species and geomorphologic features), b) combined fishing effort (entangling nets, longliners, purse seiners and trawlers), where the blue areas (no effort) are restricted fishing areas; marine reserves, military areas and aquaculture.

6.3.2.2 Geomorphological features

In order to represent different types of marine habitats we included geomorphologic features to serve as surrogate "biodiversity features". We used ten geomorphologic features within Israel territorial waters that were mapped (in 2008) and provided by The Israel Nature and Parks Authority. These features include: shallow rocks, kurkar (calcareous aeolianite) ridges, kurkar bustan, deep kurkar ridges, continental shelf silt, continental shelf sand, continental ridges, large canyons, continental slope and canyons, deep sea (Israel Nature and Parks Authority 2012).

6.3.2.3 Setting biodiversity targets

Biodiversity targets were set to protect a percentage of the species distribution according to its level of global threat based on the IUCN red list criteria (The Israel Nature and Parks Authority 2012; Table S6.1) and current range size. We set a 10% target for species that were listed "Least Concern" by the IUCN and all other fish species that have not been evaluated by IUCN. This target was increased to 15% for species listed "Vulnerable" by the IUCN (2013a) and to 20% for species listed "Endangered" by the by the IUCN. Species listed "Endangered" that had a distribution of less than 1% of the study area were given a target of 50%. For the geomorphological features, we set a target to protect 5% of all features and those that are represented by an area less than 1% of Israel's territorial waters were given a 10% target. We also set a constraint that at least 5% of the distribution of all species and features must be placed within the no-take zone (Conservation Zone), meaning that the rest of the biodiversity target could be fulfilled in other zones. While our target setting approach does not consider whether the target is adequate at conserving the species or maintaining population viability, it aims to address the IUCN criteria that define the risk of species extinction as applied in Kark et al. (2009) and Lieberknecht et al (2010). To test the sensitivity of our results we also used a 10% target for each species and a 5% target for each geomorphologic feature.

6.3.3 Incorporating economic activities in the sea

We included the two major economic activities in the Mediterranean waters of Israel (commercial fishing and hydrocarbon operations) in the conservation planning exercise. While there are other localized marine activities and features (addressed below; Table 6.1) commercial fishing and hydrocarbon operations are activities that span across Israel's territorial waters and rely on resource extraction. Thus, we focused on these activities which are likely to be the main source of opportunity cost incurred when implementing marine protected areas and zones. We translated these activities into opportunity cost layers for use within Marxan. Opportunity cost in this study was defined as the value of forgone economic activities (commercial fishing and hydrocarbon operations) when a particular area (planning unit) is made into a protected area that excludes these economic activities. As spatial opportunity cost data were unavailable for these activities, we developed surrogates to represent the annual revenue (approximation of annual opportunity cost) of each economic activity within our 1 km^2 planning units. Here we used annual values to reflect the relative opportunity cost differences across the territorial waters of Israel. We used the most current available data for Israel's territorial waters for each of these activities, specifically the year 2009 for commercial fisheries and year 2012 for hydrocarbon operations. The minimal fluctuation of Israel's annual commercial fishing catch and value over the last few years suggests that the available data of these activities is relatively comparable (Edelist et al. 2013).

6.3.3.1 Opportunity cost of commercial fisheries

We developed surrogate opportunity cost layers of commercial fishing by spatially mapping fishing effort for the four major commercial fishing gears used in Israel; entangling nets, longliners, purse seiners and trawlers (see Fig. S6.2; S6.3; see Appendix S6.3 for detailed methods). We derived effort maps by equations which assume effort is proportional to the number of fishing vessels at each port for each gear type and effort decreases exponentially with distance from port (methods described in Mazor et al. 2013b). For each gear type we used expert opinion (total of 25 experts) to refine our effort layers. We did this by constraining our effort layer by the maximum depth that each fishing gear is used and incorporating weightings over habitats and areas that are targeted by particular gear types. For entangling nets we constrained our effort layer by a depth of 50 m (maximum depth that entangling nets are used in Israel's as confirmed by 15 entangling net fishers in Israel). Longliners fishing effort was weighted by both distance from port and rocky habitat (targeted fishing areas) and confined to 50 m depth (confirmed by 6 longline fishers in Israel; Fig. S6.3). For purse seiners effort was weighted across two distinct areas in the north and south at a depth between 10 – 50 m as determined by expert opinion (6 purse seine fishers; Fig. S6.2). Trawling effort was based on data collected from on-board GPS devices by Edelist (2013) between the years 2009 – 2011 and trawling data from Israel's Department of Fisheries and Aquaculture (2012) (Fig. S6.2). Using these effort maps we created surrogate opportunity cost layers by overlaying the annual revenue (year 2009) reported by Edelist et al. (2013) for each fishing gear type, thereby, assigning monetary values to each planning unit for each fishing gear type (Fig. 6.2b).

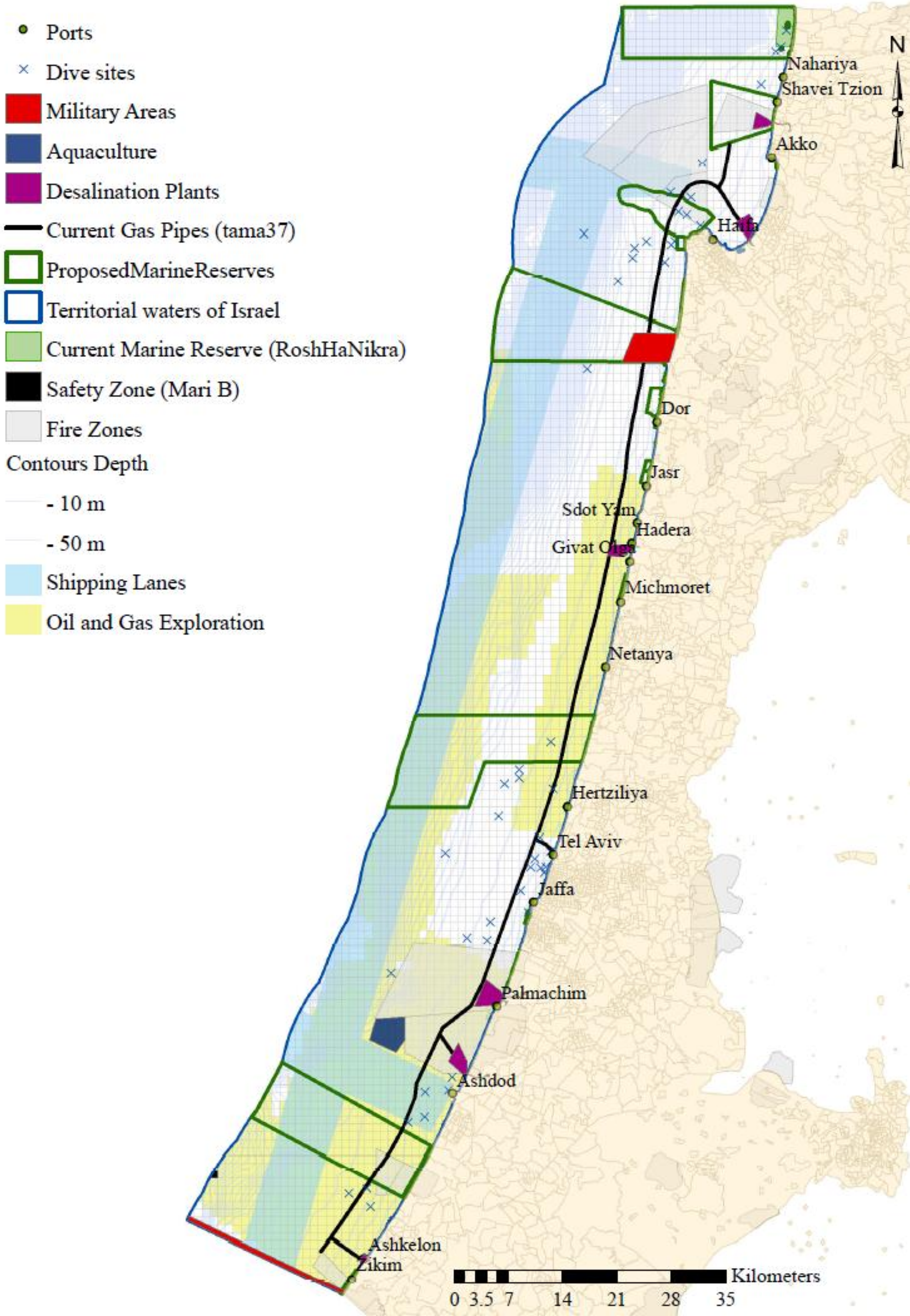


Figure 6.3. A map of the activities of Israel's Mediterranean territorial waters included in this study.

6.3.3.2 Opportunity cost of hydrocarbon operations

Spatial data identifying offshore oil and gas operations and leased and licensed marine extraction areas was provided by Israel's Ministry of Interior from the National Master Plan of Israel (Tama 34b). Areas of Israel's Mediterranean waters are licensed to several oil and gas companies (e.g., Noble Energy, Shemen, Delek) for hydrocarbon exploration for a period of seven years (resources 2012). If economically viable resources are found within these licensed areas they can then be leased by energy companies with a fifty year production permit. Unexplored "blank" areas will be temporarily left aside as Israel is trying to limit exploration into these new areas. The licensed areas that were not explored will be recycled if there is no exploration in them.

As no reliable data sources were available for a total estimation of the value of Israel's offshore oil and gas reserves we performed calculations using data from Israel's Ministry of Energy and Water Resources (Israel Department of Fisheries 2012; Table S6.3) and converted these estimated reserve quantities into monetary values. We multiplied the annual average international market price of oil (NIS per barrel = 404.52 in 2012; World Bank http://www.worldbank.org/) and natural gas (NIS per thousands of cubic meters =399.33; International Monetary Fund http://www.imf.org/external/index.htm) with Israel's estimated reserve volumes. These calculations resulted in a static estimate (year 2012 values) of the value of Israel's oil and gas reserves (not including extraction cost), but we realize that prices will fluctuate annually and are expected to reach higher values in the future, thus our calculated values are expected to be an under estimate (unless estimated reservoirs will be smaller than predicted). We have estimated the value of Israel's offshore oil and gas reserves at ~US$324 billion (~1250 billion NIS; Table S6.3), with 15% of this amount retrieved from the territorial waters (US$50 billion). Our resulting equation gives a greater weighting to the opportunity cost of leased areas (known sources of oil and gas; $\alpha = 1$) compared to licensed areas (half weighting $\alpha = 0.5$):

$$Cost\ of\ one\ exploration\ unit\ (EU)\ = \frac{\alpha(Area\ of\ EU)}{\sum Area\ of\ all\ EU} * Value\ of\ oil\ and\ gas\ (US\$),$$

$$where, \quad leased\ areas\ \alpha\ = 1\ and\ licensed\ areas\ \alpha = 0.5\ ,$$

$$and,\ the\ Cost\ of\ each\ planning\ unit\ (PU)\ = \frac{Area\ PU}{Area\ EU} * Value\ of\ EU\ unit\ (US\$).$$

6.3.3.3 Considering additional marine activities in conservation planning

There are many features to consider when planning marine conservation within territorial waters. Israel has a relatively small territorial water area with a large number of marine activities (Fig. 6.3). In addition to the fishing and hydrocarbon operations (included as opportunity cost) we included eight additional marine activities. These include: aquaculture, desalination plants, dive sites, current protected areas, exploration safety zone (500 m buffer around hydrocarbon exploration sites), military areas (fire zones), shipping lanes and pipelines (Table 6.1; see Appendix S6.4 for a full description of these activities and their data sources).We included these other activities by assigning their usage to specific zones (see Table 6.1).

Table 6.1. Four zones for Israel's territorial waters that restrict and permit different activities. Additional threats and marine activities (listed below) in Israel's territorial waters have been locked to particular zones as per the four scenarios. A "✓" in the column means that this activity was permitted in this zone, where an "x" it is prohibited.

	Zones			
Activities	**Conservation Zone "No Take"**	**Benthic Protection Zone**	**Exploration Zone**	**Economic Zone "General use"**
Trawling	X	X	X	✓
Purse Seiners	X	✓	X	✓
Gillnetting	X	✓	X	✓
Long liners	X	✓	X	✓
Oil and Gas Exploration	X	X	✓	✓
Additional threats and marine activities				
Aquaculture	X	X	X	✓
Current protected areas[7]	✓	X	X	X
Desalination plants	X	X	X	✓
Diving	✓	✓	✓	✓
Military areas	✓	✓	✓	✓
Pipelines	X	X	✓	✓
Safety area[8]	X	X	X	X
Shipping lanes	X	✓	X	✓

[7] Rosh HaNikra
[8] Mari B Platform

6.3.4 Systematic planning tools and planning scenarios

Marxan with Zones is a conservation decision-support tool that enables the user to prioritize places for different zones to achieve multiple objectives (Klein et al. 2009; Watts et al. 2009). This tool is an extension of Marxan (Ball et al. 2009), a globally used conservation planning tool for marine and terrestrial realms (Watts et al. 2009). Marxan works by minimizing one variable (e.g., the opportunity cost of commercial fishing), creating a system that is separated into areas which are protected or non-protected (Klein et al. 2009). In comparison, Marxan with Zones aims to minimize the sum of costs (i.e., incorporating more than two opportunity cost layers) and enables the user to develop a more complex system of zones that provide varying degrees of protection and have zone specific actions, objectives and restrictions (Watts et al. 2009).

We applied Marxan (Ball et al. 2009) and Marxan with Zones (Watts et al. 2009) to compare four planning scenarios for Israel's Mediterranean territorial waters (see Table 6.2). For each planning scenario we aimed to meet the same biodiversity targets while minimizing the opportunity cost incurred by other marine activities, as described below. The four scenarios increase in complexity with the inclusion of human activities (threats) and economic objectives; Simple Planning, Basic Zoning, Intermediate Zoning and Complex Zoning (Table 6.2). We define the term "activities" as any other activity within Israel's marine waters that is not biodiversity protection as proposed in this study. For the first scenario, Simple Planning, we used Marxan (without zoning) and tested two sub-scenarios; Simple Planning A with six activities and commercial fishing opportunity cost, and Simple Planning B with seven activities and combined commercial fishing and hydrocarbon extraction opportunity cost. Our second scenario, Basic Zoning, used Marxan with Zones and included three zones and six other activities. The third scenario, Intermediate Zoning, used Marxan with Zones and included four zones and seven activities (three sub-scenarios A, B and C for protection effectiveness of the Exploration Zone; see Appendix S6.1 for full explanation). In the fourth scenario, Complex Zoning, we used Marxan with Zones with four zones (for descriptions of each zone see Table 6.1) and ten other activities. For a detailed description of each scenario see Supplementary material Appendix S6.1.

Table 6.2. Four planning scenarios that were examined using Marxan and Marxan with Zones for Israel's territorial Mediterranean waters. The inclusion of features and data in each scenario is represented by a plus sign (+). Planning scenarios increase (from the Simple Planning to Complex Zoning scenario) in complexity by the planning tool, zones and number of activities included. For more detailed information on each of the zones see Table 6.1.

		Planning Scenarios						
		1. Simple Planning		**2. Basic Zoning**	**3. Intermediate Zoning**			**4. Complex Zoning**
		A	**B**		**A**	**B**	**C**	
Planning Tool:	Marxan	+	+					
	Marxan with Zones			+	+	+	+	+
Zones	Conservation Zone (No-Take)			+	+	+	+	+
	Economic Zone (General Use)			+	+	+	+	+
	Benthic Protection Zone			+	+	+	+	+
	Exploration Zone (% of effectiveness at protecting species[9])				+ (25%)	+ (50%)	+ (75%)	+ (50%)
Marine activities *(Included as opportunity cost[10])*	Commercial Fisheries: Trawlers, Purse Seiners, Gill Nets, Long liners	+	+	+	+	+	+	+
	Hydrocarbon Operations		+		+	+	+	+
(Assigned to specific zones) [11]	Aquaculture	+	+	+	+	+	+	+
	Current Protected Areas	+	+	+	+	+	+	+
	Diving areas	+	+	+	+	+	+	+
	Military areas	+	+	+	+	+	+	+
	Safety platform	+	+	+	+	+	+	+
	Shipping lanes							+
	Desalination plants							+
	Pipelines							+
Number of Zones		**0**		**3**	**4**			**4**
Total number of marine activities included in the analysis		**6**	**7**	**6**	**7**			**10**

[9] We assigned possible percentages of biodiversity protection that may be achieved by the Exploration Zone. The Conservation Zone assumes 100% protection of .biodiversity but due to the unknown impacts of hydrocarbon exploration we tested different values (25%, 50%, and 75%). See File S2. for further details.

[10] The opportunity cost layers are the variables where are minimized in Marxan software: In Marxan this is treated as one minimized cost layer, and in Marxan with Zones these opportunity cost layers are combined (summed) and then minimized.

[11] See Table 1 for the zones that each marine activity is permitted or restricted within and File S2. for detailed explanation of each activity and their data references.

6.3.5 Comparing planning scenarios

Four planning scenarios (Table 6.2) were compared. The Simple Planning scenario (without zoning) was run using Marxan and the other three scenarios (Basic Zoning, Intermediate Zoning and Complex Zoning) used Marxan with Zones, all scenarios with 1,000 runs each. Based on the results of the 1,000 runs we calculated the average opportunity cost and number of 1 km^2 planning units within each zone that were needed to meet our biodiversity targets. We tested the ability of planning scenarios to meet all biodiversity targets. In cases where targets were unable to be reached for a particular species, we eliminated the constraint for 5% of their protection to be met in the Conservation Zone. Thus, we re-ran our results with the same altered targets for all scenarios. We then mapped the selection frequency outputs (number of time a planning unit is selected in Marxan for a particular zone) for each planning scenario and each zone. To compare between zoning configurations and scenarios we also mapped the best solution that Marxan could find. To test the similarity between the selection frequency outputs for each scenario we used the Spearman Rank Correlation (ρ). Higher values indicate a more similar spatial pattern in selection frequencies, meaning that these plans will require similar conservation actions.

6.3.6 Evaluating trade-offs

We evaluated the trade-off between meeting biodiversity targets and maximizing annual fishery revenue for each of the four fisheries in Israel's Mediterranean Sea, following methods described in Klein et al. (2009). In this analysis we assume that lost area relates to lost revenue, without considering possible redistribution of fishing or hydrocarbon extraction efforts. These trade-offs can only be evaluated for scenarios using Marxan with Zones that enables multiple variables to be considered. We set fishery targets where we aimed to preserve an equal percentage of the total fishing revenue (from the fishing effort maps) for each of the four fishery gear types. These targets could only be met within zones that did not restrict that type of fishery (Table 6.1). Expanding this analysis, we tested the trade-off with areas that are leased and licensed for oil and gas (using the hydrocarbon opportunity cost layer described above). We therefore included a hydrocarbon target (preserving hydrocarbon industry revenue) as well as both biodiversity and fishery targets. Fishery targets were extracted from the previous trade-off analysis; the highest target where all biodiversity targets were met.

6.4 Results

6.4.1 Comparing planning scenarios for territorial waters

Here we compared our four planning scenarios (Table 6.2) by: a) the ability to reach biodiversity targets, b) the change in opportunity cost and c) the alteration of spatial conservation priorities.

a) Biodiversity targets

We found that meeting the same biodiversity targets became more difficult as our planning scenarios included more marine activities. The Simple Planning (without zoning and six activities) and Basic Zoning (three zones and six activities) scenarios met all biodiversity targets, the Intermediate Zoning scenario (four zones and seven activities) met 98% of targets and the Complex Zoning scenario (four zones and ten activities) met 96% of targets (Table 6.3). Our constraint (5% target in the Conservation Zone – no-take area) was unable to be met in the Intermediate and Complex Zoning scenarios for nine species (Table S6.4) that had restricted distribution ranges that overlapped with prospective hydrocarbon exploration areas. For each of these nine species we eliminated constraint; however the overall biodiversity target for these nine species remained and was met within other zones. Targets were then able to be met for all planning scenarios.

b) Opportunity cost

We found that more complex planning scenarios incurred greater opportunity cost (Table 6.3). When comparing the two Simple Planning scenarios (Simple Planning A with six users and commercial fishing opportunity cost, and Simple Planning B with seven users and combined commercial fishing and potential hydrocarbon opportunity cost) we found that a reserve network that only included the opportunity cost of fishing had a substantially lower cost (Simple Planning A = US$2.05 million) compared to a plan that included the opportunity cost of hydrocarbon operations (Simple Planning B = US$595,132.38 million). Comparing our zoning scenarios (when targets are met 100% in each scenario) we found that the most expensive zoning scenario is the Intermediate Zoning scenario A that assumes the Exploration Zone can provide a zone effectiveness measure of twenty-five percent. This opportunity cost decreased as the Exploration Zone's ability to protect biodiversity (zone effectiveness) was increased to fifty percent (Intermediate Zoning scenario B 10.5% cost decrease) and seventy-five percent (Intermediate Zoning scenario C 14.9% cost decrease); allowing targets to be met more easily within the Exploration Zone. The Intermediate Zoning scenario increased opportunity cost by 27.8% from the Basic Zoning scenario (three zones and six activities) as we introduced the opportunity cost of prospective oil and gas reserves as well as a fourth zone (Exploration Zone). The Complex Zoning scenario also increased the opportunity cost of the Basic Zoning plan by 35.7% and Intermediate Zoning B plan by 6.2%.

Table 6.3. Results showing average opportunity cost for 1000 Marxan runs for each planning scenario. Targets were set according to IUCN criteria and the size of a species distribution range (as described in the methods section). The constraint/target that 5% of the distribution of all features needs to be within the Conservation Zone (no-take zone) was unable to be reached for nine species. This constraint was removed for these species so targets could all be met. This table shows the opportunity cost of each planning scenario, the percentage of biodiversity targets met in the scenario and the percentage of "no-take area" surface coverage of the entire reserve system. For a description of planning scenarios see Table 6.2.

Planning Scenario	**Opportunity cost (US$ million)**	**Percent of targets met**	**Percent of conservation zone in entire reserve design (no-take areas)**
Simple Planning A	2.05	100	22
Simple Planning B	595,132.38	100	21
Marxan with Zones			
Basic Zoning	4.09	100	22
Intermediate Zoning B 50%	333,004.37	98	17
Complex Zoning	4.20	96	14
Marxan with Zones (minus nine species for the 5% Conservation Zone target)			
Basic Zoning	3.92	100	22
Intermediate Zoning A 25%	5.59	100	18
Intermediate Zoning B 50%	5.01	100	17
Intermediate Zoning C 75%	4.76	100	17
Complex Zoning	5.32	100	14

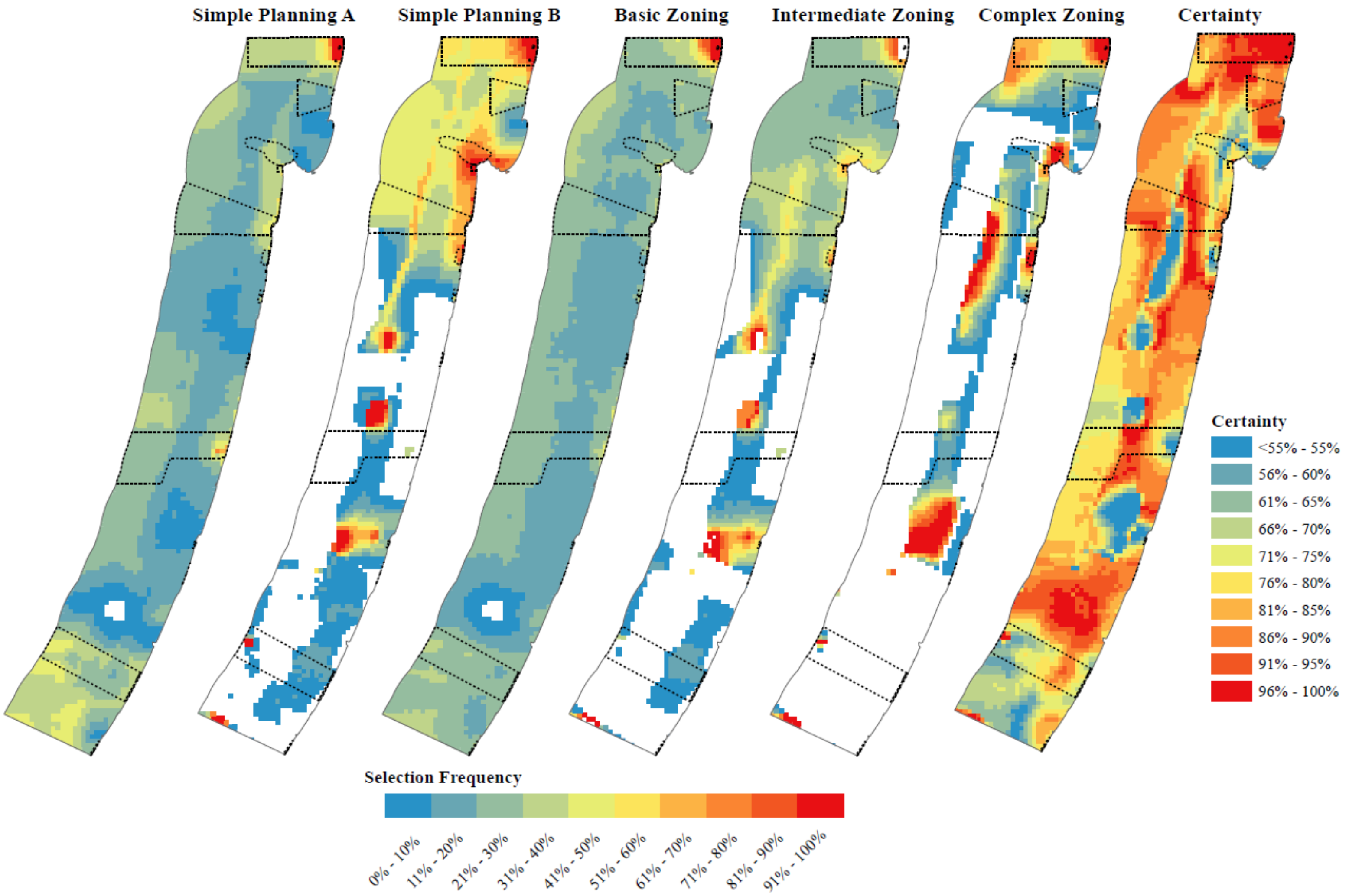


Figure 6.4. Selection frequency output maps (shows the percentage of times a planning unit was selected when run in Marxan 1000 times) from Marxan with Zones for each Zone and each zoning scenario. All scenarios met biodiversity targets. The dashed black lines represent the proposed marine reserve system by Israel's Nature and Parks Authority. White areas on the maps indicate no selection (0% selection frequency) in these areas. The certainty map expresses the level of certainty/agreement of planning units selected (either highly selected for no-take areas or low selection) across all planning scenarios. Therefore, the higher the percentage of certainty means there is more agreement between scenarios.

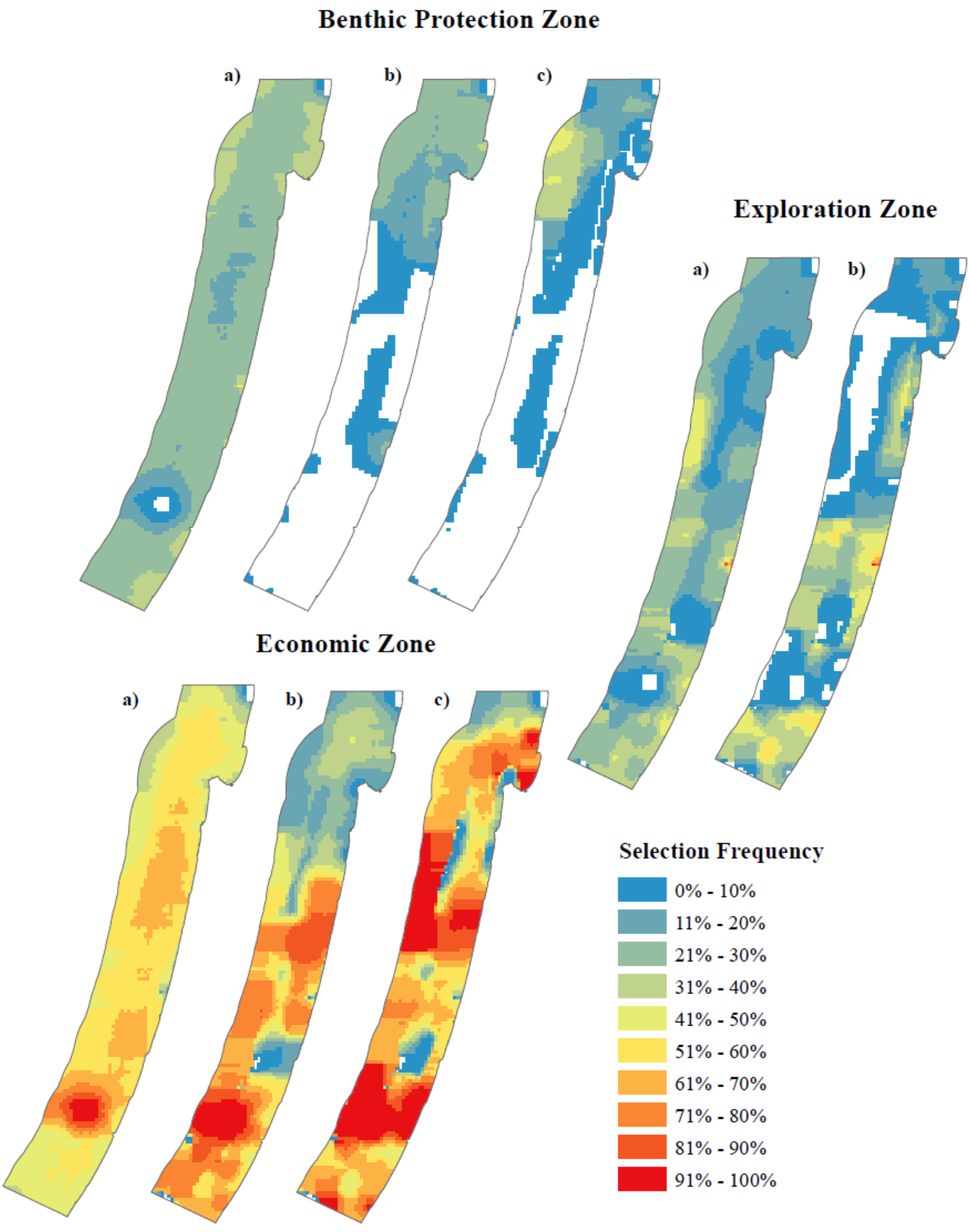


Figure 6.5. Selection frequency output maps (shows the percentage of times a planning unit was selected when run in Marxan 1000 times) from Marxan with Zones for each Zone and each zoning scenario. For the Benthic Protection Zone and Economic Zone the three scenarios are a) Basic Zoning, b) Intermediate Zoning, c) Complex Zoning. For the Exploration Zone the two scenarios are a) Intermediate Zoning and b) Complex Zoning.

c) Conservation priorities

Selection frequency outputs from our analysis indicated that spatial configurations are substantially altered by the inclusion of hydrocarbon opportunity cost (Fig. 6.4; Fig. 6.5). The scenarios Simple Planning A (without zoning and six activities) and Basic Zoning (three zones and six users), which did not include hydrocarbon opportunity cost had a high Spearman's rank correlation of $\rho = 0.84$ ($p < 0.001$; Table 6.4).

We found that priority areas for no-take reserves (Conservation Zone) were mainly concentrated in the north and south of Israel's territorial waters. From the best solution outputs (Fig. 6.6) no-take areas moved from areas in the south to areas in the north with the inclusion of potential hydrocarbon extraction data. Similarly, the three scenarios that included hydrocarbon opportunity cost (Simple Planning B, Intermediate Zoning and Complex Zoning) had selection frequency outputs that were significantly correlated (Table 6.4). The most similar spatial outputs were between Simple Planning B and Intermediate Zoning B ($\rho = 0.86$, $p < 0.001$; Table 6.4). In these three scenarios, we discovered that spatial priorities were much more restricted (higher spectrum of selection frequency; see Fig. 6.4) than the Simple Planning A and Basic Zoning scenarios. Areas with high selection frequency for placing no-take reserves were off the coast of Jaffa, in coastal waters between Dor and Haifa Bay and along the northern border with Lebanon (Fig. 6.4).

Priority areas for each zone become more pronounced as planning scenarios became more complex and restricted by the inclusion of other marine activities. Conservation priorities for the Benthic Zone were most similar between the Intermediate Zoning B and Complex Zoning ($\rho = 0.82$, $p < 0.001$; Table 6.4). In all scenarios we find that Benthic Protection Zone has higher selection frequency in the northern part of the Sea. In the best solution outputs we also notice how benthic protection becomes confined to the north with the inclusion of the Exploration Zone (Fig. 6.6). The Economic Zone has highest selection frequency in the south for the Basic Zoning scenario where high fishing pressure is evident. In the Intermediate Zoning B scenario the high selection frequency of this zone extends over the south and central region where hydrocarbon is included. Further expansion of this zone's high selection frequency extends to the north as shipping lanes and pipelines are included in the Complex Zoning scenario. The Exploration Zone's priority areas were dissimilar between the Intermediate and Complex Planning scenarios, ($\rho = 0.42$, $p < 0.001$; Table 6.4). The inclusion of other marine activities affected the available area for the Exploration Zone.

6.4.2 Evaluating trade-offs between conservation and economic objectives

In the Basic Zoning scenario (three zones and six users) all biodiversity targets were met with a loss of 7% of commercial fishing revenue (Fig. 6.7a). By increasing the complexity of our planning scenarios (including more marine activities) we found that our biodiversity targets could only be met by decreasing the area of fishery grounds, consequently decreasing the revenue. Hence, the resulting fishing revenue loss was 12% for the Intermediate Zoning scenario (zoning network that includes four zones) and 15% for the Complex Zoning scenario.

In comparison, by including the economic objectives of hydrocarbon operations while meeting biodiversity and fishery targets (all four fishing gear types targeted 88% (Intermediate Zoning) and 85% (Complex Zoning) of revenue; values obtained from Fig. 7a), a small revenue loss was incurred (Fig. 6.7b). For the Intermediate Zoning scenario 5% of hydrocarbon revenue was lost. Similarly, the Complex Zoning scenario kept biodiversity and fishery targets with revenue losses of 6%. Therefore, for a loss of ~5% of hydrocarbon revenue, fishery and biodiversity targets could be fully met. Interestingly, we found that the drop-off rate of not meeting biodiversity and fishery targets was very minimal for the hydrocarbon industry in comparison with the rate at which biodiversity and fishery targets were traded off. Moreover, if hydrocarbon revenue was not traded-off (100% revenue was maintained), biodiversity and fishery targets could reach ~98% (Fig. 6.7b). However, if fishery revenue was not traded off (100% revenue was maintained), biodiversity targets could only reach between 93-85% (Fig 6.7a).

Table 6.4. Spearman rank correlation (ρ) of the similarity between the selection frequency outputs of each planning scenario. High values (closer to 1) indicate a more similar spatial pattern in selection frequencies, meaning that these plans will require similar conservation actions. All scenarios show significant correlations (p<0.001).

Zone	**Planning Scenario**	**Simple Planning A**	**Simple Planning B**	**Basic Zoning**	**Intermediate Zoning B**	**Complex Zoning**
Conservation Zone	Simple Planning A		0.69	0.84	-0.09	-0.09
	Simple Planning B	0.69		0.11	0.86	0.67
	Basic Zoning	0.84	0.11		-0.04	-0.06
	Intermediate Zoning B	-0.09	0.86	-0.04		0.73
	Complex Zoning	-0.09	0.67	-0.06	0.73	
Benthic Zone	Basic Zoning				0.45	0.33
	Intermediate Zoning B			0.45		0.82
	Complex Zoning			0.33	0.82	
Economic Zone	Basic Zoning				0.41	0.29
	Intermediate Zoning B			0.41		0.58
	Complex Zoning			0.29	0.58	
Exploration Zone	Intermediate Zoning B					0.42
	Complex Zoning				0.42	

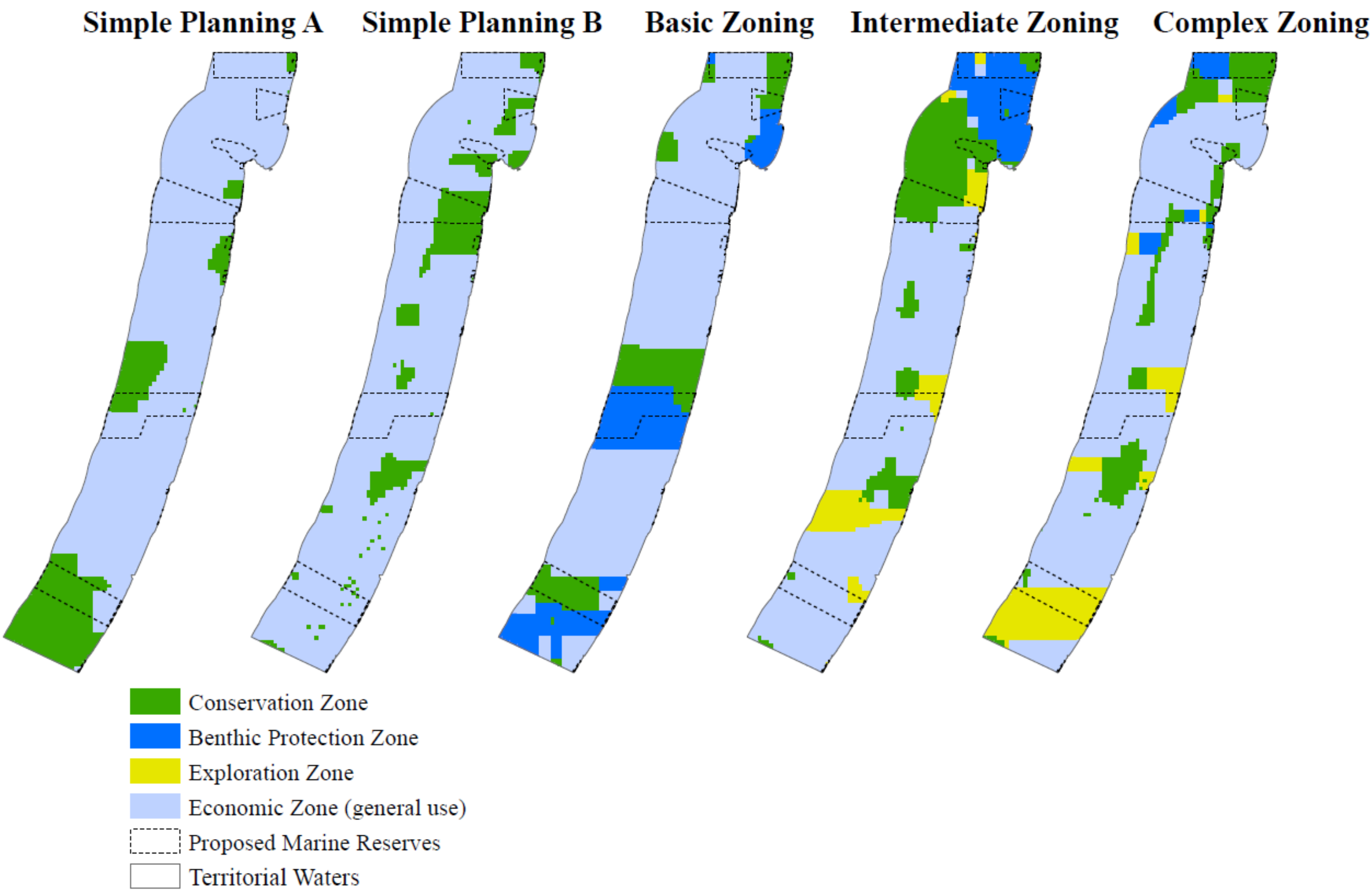


Figure 6.6. Marxan best solution outputs (the reserve configuration that best reduces opportunity cost and meets biodiversity targets from 1000 Marxan runs) for each planning scenario. The four colours designate the four types of zones (see Table 6.1).

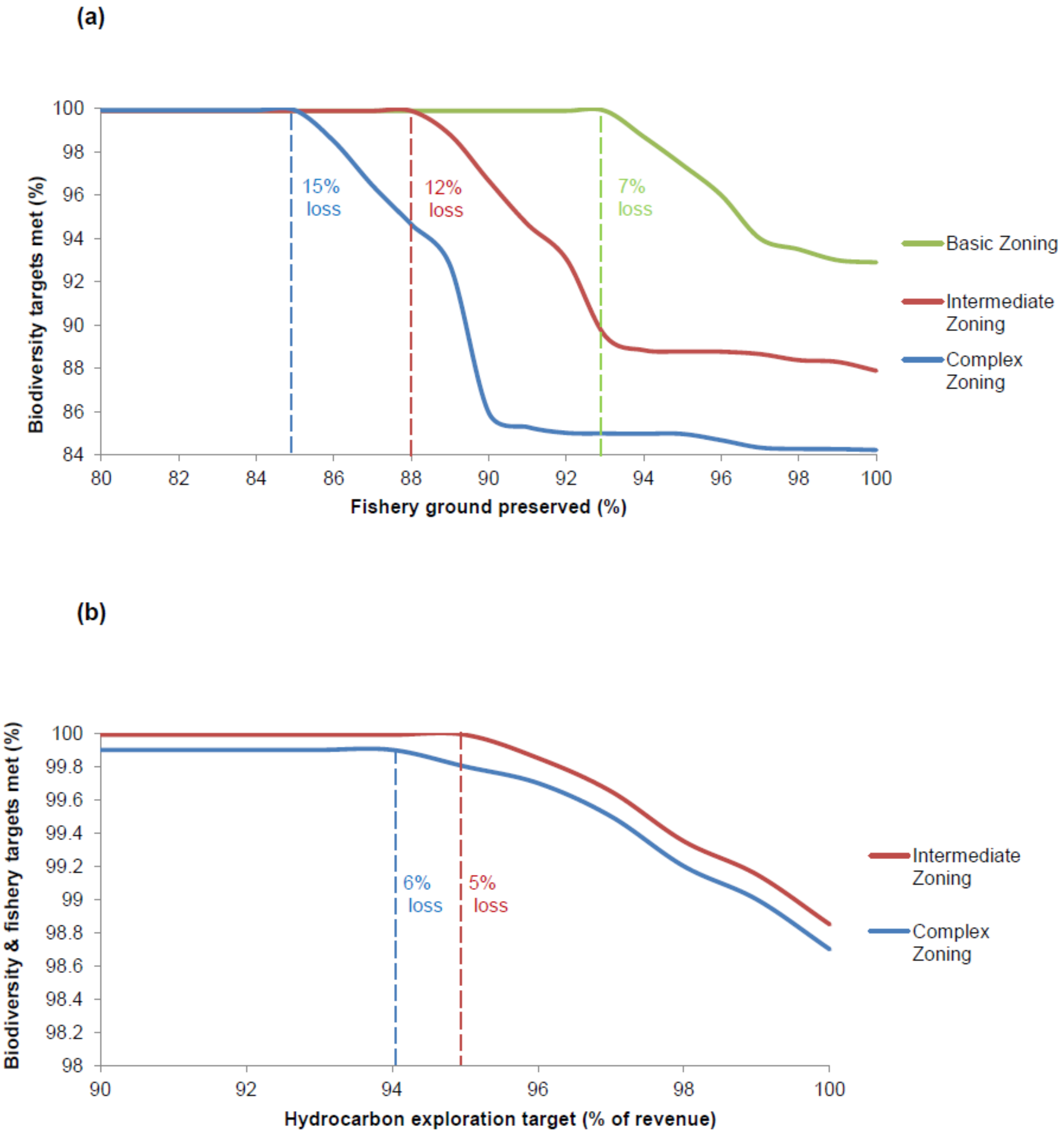


Figure 6.7. The trade-off between meeting biodiversity targets and maintaining economic objectives for each zoning scenario: (a) biodiversity targets are met when the fishery targets (percentage of annual fishery revenue) are less than 93% (7% revenue loss) in the Basic Zoning scenario (three zones and six activities), less than 88% (12% revenue loss) in the Intermediate Zoning B scenario (four zones and seven activities), and less than 85% (15% revenue loss) in the Complex Zoning scenario (four zones and ten activities), (b) biodiversity targets are met when hydrocarbon operations (leased and licensed expected revenue) are less than ≤ 95% (5% revenue loss) in the Intermediate Zoning scenario and less than 94% (6% revenue loss) in the Complex Zoning scenario.

6.5 Discussion

This study demonstrates how conservation objectives can be achieved while considering economic objectives where there are multiple marine activities. We found that the inclusion of many activities in marine conservation plans can significantly alter spatial priorities (Table 6.4; Fig. 6.4; Fig. 6.5). Economic goals are more compromised (in this case for the fisheries and hydrocarbon industries) (Fig. 6.7) to achieve biodiversity targets when there are more marine activities in the planning process. Moreover, complex plans with more activities involved greater opportunity cost and did not reach biodiversity targets as easily as more simplified plans with less marine activities. Given that a complex plan is working with a more constrained problem, this result is expected (McDonald 2009; Weeks et al. 2010a). Despite the increased opportunity cost and lack of spatial flexibility to achieve biodiversity goals with more complex conservation plans, planning that incorporates other activities can steer us towards areas which are feasible (greater potential for implementation success), minimize conflict with other users and reduce threats to biodiversity.

Conservation planning and zoning with multiple activities is challenging. Our case study shows that decisions made by conservation planners such as the number of zones or number of marine activities included in the planning process can substantially shape the resulting zone and reserve configuration. Therefore, it is important to first identify the impact that each activity and feature could have on marine biodiversity in the study system and the appropriate conservation action to take (Pressey et al. 2007). Here we follow a framework (Fig. 6.1) to help conservation planners address offshore activities and their potential threat in the marine realm. This framework outlines the steps needed to comprehensively zone for biodiversity protection while maintaining economic goals and can be a useful guide for countries currently striving to zone their waters (Agardy 2010). One of the most important steps is testing the sensitivity of the results to user decisions (e.g., the inclusion of data, number of zones, the targets, see Step 7 Fig. 6.1; Warman et al. 2004). Other challenges that need to be accounted for when zoning include: the lack of shared information between stakeholders (Levin et al. 2014), the unknown expansion and objectives of industries (Sivas & Caldwell 2008), the unknown value of economic industries (Douvere 2008), and unforseen threats or disasters (Agardy 2010). Given that some of these challenges can be overcome, in reality, conservation planning is largely shaped by stakeholder perceptions and the willingness to trade-off economic and conservation objectives. Moreover, there is no one correct solution to planning within a complex system (Game et al. 2013).

Trade-off analysis is an important step to include in conservation planning (Hirsch et al. 2011; Halpern et al. 2013). It enables us to determine how much of a commercial activity may be forgone in order to achieve biodiversity targets. It also helps to address the implementation gap (the gap between conservation planning and real-world action) inherent in many conservation plans (Knight et al. 2008). However, Hirsch et al. (2011) cautions that not every problem can be solved by compromise. For example, we assume in our study that a portion of the hydrocarbon leased and licensed areas and commercial fishing grounds are available for trade-off, whereas stakeholders may disagree and reject any compromise. In marine conservation, fishing trade-offs have been the focus of several studies (Klein et al. 2009; Weeks et al. 2010a; Grantham et al. 2013). In this study we have incorporated economic trade-offs for both the commercial fishing and hydrocarbon industries, indicating the necessary compromises that are needed to meet our biodiversity targets in each planning scenario. Specifically, we have triaged targets for nine species that were unable to be met within no-take zones (of the Intermediate Zoning and Complex Zoning scenarios) and enabled them to be met within other zones. We suggest that future work should expand this type of analysis to examine the trade-offs with other social, economic and cultural activities where appropriate.

Marine features and activities which are confined in space may be difficult, or impossible to trade-off. In this study we introduced a range of features into marine conservation planning in addition to the more traditionally used fishing such as pipelines, shipping lanes, desalination plants and aquaculture. In comparison to the full coverage of commercial fishing practises and the wide cover of hydrocarbon exploration across the study area, other features are restricted to a specific area (Fig. 6.3). Such restricted features are difficult to plan around as they often cannot be traded-off. Fishing effort for example can be redispersed to other spatial areas when an area is declared a marine reserve (Halpern et al. 2004; Roberts 2005), whereas aquaculture farms are more difficult to relocate. We also found that linear-shaped features such as pipelines and shipping lanes influence the shape of marine zones, causing thin elongated zones (Fig. 6.6). Therefore, conservation planners must decide whether such features are planned around, planned with, or ignored. Performing a cost-benefit analysis of altering some of these features (e.g., rerouting shipping lanes, planning reserves over pipelines or moving planned aquaculture cages) within various planning scenarios could be a way to examine their potential flexibility within the reserve system. We suggest that future research explores the way that such features are included in conservation planning as they can have an influence on the selection of conservation priorities.

We found that the incorporation of oil and gas exploration can substantially alter spatial priorities and the opportunity cost of conservation. This is the first time that offshore hydrocarbon operations are explicitly incorporated in a Marxan analysis. Possible reasons for its absence in previous conservation plans are because a) the economic gains that are at stake are so large that these areas are "off-limits" to all other marine activities (e.g., Australian commonwealth zoning plan; Barr & Possingham 2013), b) uncertainty as to how to incorporate hydrocarbon information and c) the uncertain future of the industry that is dependent on new discoveries and may quickly demand large marine space (e.g., new discoveries in the Mediterranean Sea, EIA 2013). If we incorporate hydrocarbon information by assuming such areas cannot be protected, we may not be able to achieve a representative reserve network. One of the problems we encountered with including prospective hydrocarbon exploration is that sometimes biodiversity targets could not be achieved because a few species substantially overlapped with hydrocarbon interests. Thus, we must carefully assess our targets and understand the compromises or actions that need to be taken in order to ensure conservation-worthy species are maintained in the face of hydrocarbon operations. We suggest that conservation plans endeavour to incorporate mining and fossil fuel data where possible to avoid costly conservation mistakes.

The ability of hydrocarbon exploration areas to provide some level of protection for biodiversity is unknown. In this study we tested different levels of protection from the "Exploration Zone" (see Intermediate Zoning Table 6.3; "zone effectiveness" see Makino et al. (2013b)) and found that opportunity cost is reduced if hydrocarbon areas are able to contribute to biodiversity protection. This is a novel conservation planning example that incorporates the notion of hydrocarbon areas providing some conservation benefit. The impacts of oil spills and gas leaks on marine biodiversity are severe and are well documented (Gomez et al. 2003; Lee & Lin 2013; Rooker et al. 2013). Likewise, there is some understanding of the impacts of offshore construction and extraction e.g., drilling impacts that are damaging to benthic structures (Davies et al. 2007). However we have little understanding of the impacts posed by the ongoing maintenance of a drilling site, or one that is dormant (leased or licensed without current activity). We suggest that future research focuses on better understanding the impacts that hydrocarbon operations pose on marine biodiversity and further develop ways to include hydrocarbon information into marine conservation plans.

Our study has interesting implications for Israel. We found that for Israel's territorial waters we can meet all our biodiversity targets (but not all no-take zone targets) for a loss of ~15% of annual commercial fishery revenue and ~5% of potential hydrocarbon revenue. A reduction of 7% of fishery revenue was needed to meet our biodiversity targets if hydrocarbon exploration is ignored. Our planning scenarios indicate that a surface area of 14-22% (Table 6.3) of Israel's territorial waters needs to be protected to meet biodiversity targets. The marine area reserved in Israel is currently less than 1% of the territorial waters (Yahel & Angert 2012), although none of these are considered no-take areas. Efforts are currently being undertaken to expand Israel's reserve network. The proposed network has been planned using species gradients and the representation of geomorphological features, without the use of Marxan or other similar tools (Yahel & Angert 2012). We found that there is some overlap between the proposed marine reserves and the high priority areas found in our study (Fig. 6.4). The primary overlapping areas include: the proposed reserve in the north (Rosh Hanikra), the Haifa headland, the proposed reserve near Atlit and the smaller sized reserve near Dor. While different methods have been used for these two plans, some results are overlapping and we recommend that these areas that overlap should be targeted as initial reserve priorities for Israel as they are robust to the kind of process used to define priorities. However, it should be cautioned, that while overlapping priorities could be a good starting point, they will not likely provide a representative network that meets biodiversity targets.

Marine conservation planning often lacks good quality spatial data and must therefore rely on surrogate measures (Naidoo et al. 2006; Weeks et al. 2010b; Levin et al. 2014). In this study, the surrogate fishing effort layers were generated with large involvement and input from experts. In comparison, our opportunity cost layer for hydrocarbon operations, although based on available government data, may less accurately reflect unpredictable shifts in future opportunity cost due to the fluctuating price of fossil fuels. Here we also set relatively low biodiversity targets because very minimal marine protection exists in this area, thus, these targets are potentially achievable (20% of Israel's Mediterranean Sea needs to be protected to meet our biodiversity targets (Table 6.3), corresponding with Israel's proposed target by the Israel Nature and Parks Authority (Yahel & Angert 2012)). These targets do not guarantee species persistence, but increasing these targets may mean that other targets become unachievable, particularly within the Intermediate Zoning and Complex Zoning scenarios. We have included several novel features in our planning (e.g., aquaculture farms, desalination plants, shipping lanes and pipelines), yet there are other features that could be incorporated in future work, for example sand mining, offshore power plants, tourism and recreational fishing. Similarly, management, monitoring and hydrocarbon production costs can also be included in future studies (Naidoo et al. 2006). The aim of this study was to evaluate the impact of including multiple features and activities into marine conservation, however we do intend for these results to serve as useful baseline plans for the territorial waters of Israel. To improve the selection of conservation priorities in Israel's Mediterranean waters future work should attempt to build upon these scenarios, including additional species data for which data is currently limited, incorporate additional marine activities and create more robust cost layers with the availability of new data.

This case study can serve as an example for many other countries around the world, which are faced with the need to carefully balance economic considerations while protecting marine biodiversity. It is particularly relevant for countries surrounding the Mediterranean Sea that share common challenges and arising threats from developing offshore hydrocarbon exploration to biodiversity and ecosystems (EIA 2013). Our results suggest that planning with more complexity (e.g., multiple economic objectives, multiple threats and multiple zones) will be slightly more costly, have higher trade-offs with other marine activities and will require more input data. Despite these inefficiencies, a complex plan considers the objectives of more stakeholders (marine activities) and is more likely to result in successful implementation of conservation outcomes (Knight et al. 2008) and better compliance than a plan which ignores other activities. In the Mediterranean region with its many marine users, this is particularly important where compliance is often a major limiting factor in reserve design and implementation success (Fenberg et al. 2012). We propose that countries aiming to protect marine biodiversity in their territorial waters should move from a single objective approach to one that links to the broader socioeconomic context incorporating multiple activities. A way forward may be the incorporation of lessons from marine spatial planning (Douvere & Ehler 2009; Ehler & Douvere 2009) into marine conservation planning, while aiming to maintain biodiversity goals and examine trade-offs. Explicitly quantifying trade-offs can provide an initial starting point for discussion between stakeholders (Hirsch et al. 2011) and ultimately enable successful conservation outcomes which other marine users are willing to comply with.

6.6 Supplementary material

Appendix S6.1. Description of each planning scenario

The first scenario “Simple Planning” uses Marxan which aims to maximise conservation objectives for a minimal cost. We defined the cost in this study as the opportunity cost of the commercial fishing sector if an area is turned into a marine reserve. Thus, our aim is to minimise the impact to commercial fishermen and at the same time reach our biodiversity targets (described above). We further extend this scenario “Simple Planning A”, to also incorporate the opportunity cost of offshore oil and gas extraction as quantified above, which we refer to as “Simple Planning B”.

For three planning scenarios we used a zoning tool, Marxan with Zones. This tool is an extension of Marxan, where multiple objectives are minimised to produce a compact system of marine zones. We ran Marxan with Zones using defined zones (see Table 6.1). Marxan with Zones enable us to meet our biodiversity targets within multiple zones. We set our “Conservation Zone” (no take zone) to include at least 10% of the total species distribution. Our second scenario is “Basic Zoning” which only uses three zones; Conservation Zone; Benthic Protection Zone and Economic Zone.

The third scenario, Intermediate Zoning scenario, uses all four zones (Table 6.1) and tests three possible values for the zone effectiveness of the “Exploration Zone”. Zone effectiveness is the percentage of protection a given zone provides for each conservation feature (Watts et al. 2009). These values are difficult to obtain as quantifying the impact of different activities on numerous species and habitats is very difficult. Thus, previous studies (Mills et al. 2011) have used expert opinion to obtain zone effectiveness measurements for conservation features. Due to the difficulty of quantifying the impact of hydrocarbon exploration on the multiple species in the study and the absence of data available on this topic we apply protection levels of A) 25%, B) 50% and C) 75% for the Exploration Zone. These percentages try to represent different hypotheses, one that oil and gas exploration infrastructure yields benefits e.g., artificial reef structures that can increase biodiversity and can protect species, and another that suggests there are many threats e.g., oil spills, sound pollution, light pollution and destruction of benthic structures and organisms when platform installation occurs. For the “Economic Zone” (general usage zone) we assume it has no protection and thus a zone effectiveness of 0% for each feature, the “Conservation Zone” an effectiveness of 100% protection for each features, and for the “Benthic Protection Zone” we used species vulnerability values from expert surveys extracted from published literature (http://www.fishbase.org/; Cheung et al. 2005; Donlan et al. 2010; Coll et al. 2010), and for geomorphologic structures, we set a value of 50% due to the unknown zone effectiveness on these formations.

The fourth scenario, Complex Zoning, assumes that all leased and licensed hydrocarbon areas in the territorial waters should be avoided when planning a marine reserve to prevent future incompatible objectives with this industry. Therefore we assume that this prospective hydrocarbon area can only be allocated to either the Exploration Zone or the Economic Zone. In this scenario we also incorporate shipping lanes, desalination plants and pipelines (see Table 6.1 for a list of activities and their placement in specific zones).

Appendix S6.2. Method of deriving biodiversity features data

Fishes: We removed fish species that had less than two sightings in the past ten years on the basis of expert opinion that these are species are rare and their distribution is unknown. Removed species included: *Eutrigla gurnardus, Microlipophrys nigriceps, Remora remora, Rhinobatus cemiculus, Scomber scombrus, Sprattus* sprattus, thus resulting in a total of 153 fish species. It is important to note that none of the eight studies we derived our information from have been conducted at depths greater than 200m. Thus, we have little understanding of the number of species present in deeper territorial waters. However, it is expected that species richness and abundance decline as the continental shelf ends and slopes towards the depths of the ocean floor (Morantal et al. 1998; Kallianiotis et al. 2000; D'Onghia et al. 2004; Tecchio et al. 2011). Experts [12]that were used in this study to verify distribution and ranges included; Dr. Golani, Dr. Goren, Dr. Rilov, Dr. Edelist and Dr. Brokovich.

Cetaceans: We only included the Common Bottlenose Dolphin (*Tursiops truncatus*) in this study. While other regular mammal species that visit Israel's waters include the: Striped Dolphin, Common Dolphin, Risso's Dolphin, Rough-toothed Dolphin and Cuvier's Beaked Whale (Kerem et al. 2010) not enough observational data exists to determine priority habitats for these species. Surveys for the Common Bottlenose Dolphin were recorded in Scheinin (2010) along with extra sightings from Israel Marine Mammal Research & Assistance Centre (IMMRAC) between the years 2003 - 2011. From these sources sightings of the Common Bottlenose Dolphin have been recorded throughout Israel's territorial waters. A sampling effort analysis on the available data provided by A. Scheinin was performed by Israel's Nature and Parks Authority and the three core feeding and foraging areas were identified from this analysis (Kerem et al. 2014).

[12] Dr. Golani – The Hebrew University of Jerusalem; Dr. Goren – TelAviv University; Dr. Rilov; Israel Oceanographic and Limnological Research institute; Dr. Edelist – University of Haifa, Dr. Brokovich – The Hebrew University of Jerusalem.

Appendix S6.3. Methods for calculating opportunity cost of commercial fishers in Israel.

Opportunity cost of commercial fishers

We developed spatial fishing effort maps for all four fishing gears used in Israel; entangling nets, longliners, purse seiners and trawlers (see Fig. S6.2; S6.3). Effort maps were compiled by expert opinion, which involved seventeen experienced fishers, two fisheries rangers, six marine scientists including experts from the department of Fisheries and Agriculture. Using these effort maps we further created surrogate cost layers by overlying the 2009 annual revenue reported by Edelist et al. (2013) for each fishing gear type. For the combined fishing effort for Israel's Mediterranean territorial waters (see Fig. 6.2b).

Entangling nets:

The maximum depth that entangling nets are used in Israel's waters is ~50 m (confirmed by 15 entangling net fishers in Israel). Using bathymetry data (Amante & Eakins 2009) we cropped our study area from the coast line to the 50m contour line. We distributed the 2009 entangling net captured biomass (615.6 ton) which is valued at US$3.53 million (Edelist et al. 2013) across the designated fishing area. Specifically, we weighted our cost layer by the number of entangling net fishing boats at each port along the coastline of Israel (data provided by Department of Fisheries & Aquaculture 2010) and assume that effort decreases exponentially with distance from port (see Mazor et al. 2013b). Thus, each planning unit (1 km^2) *pu* represents the total annual revenue that is extracted from this area using entangling nets and can be defined by:

$$pu\ revenue = \frac{effort * pu\ area}{\sum of\ all\ pu\ (effort * pu\ area)} * revenue\ (US\$)$$

$$effort\ = \sum (number\ of\ fishing\ boats\ at\ port\ *\ exp\ (-0.01\ *\ distance\ to\ port))$$

Longliners:

The majority (~90% of fishermen) of longliners are used in Israel's waters at depths less than ~50 m (confirmed by 6 long line fishers in Israel). While there are a few long liners that operate beyond this depth, for this study we have set a maximum depth of 50 m. Using bathymetry data (Amante & Eakins 2009) we cropped our study area from the coast line to the 50 m contour line, remaining with 1,784 planning units. We used the 2009 biomass (130 ton) which is valued at US$1.56 million (Edelist et al. 2013). As a surrogate measure to spatially represent the revenue of long liners within Israel's waters we weighted each planning unit with revenue values that decrease exponentially

with distance from ports (weighted by the number of longliner fishing vessels at each port; see Equation 1 & 2)) and by distance to nearest rocky habitat (kukar ridges; The Israel Nature and Parks Authority 2012) which are targeted in longline fishing. As longliner fishing efforts are concentrated mostly on rocky habitats we set a weighting so the revenue in each planning unit was:
*pu revenue = (effort on rocky habitats * 0.75) + (effort from ports * 0.25)*

Purse Seiners:
There are two distinct areas for purse seiners in Israel. One is concentrated in the north in the Haifa Bay and the second area is in the south between Ashdod to Tel-Aviv at a depth of ~10 – 50m (concentration of effort around a pipeline that is at a depth of ~30m). Here we weighted planning units by the % of effort as derived from expert opinions (6 purse seine fishers) and spread the annual revenue of purse seiners (US$1.38 million; Edelist et al. 2013) over these weighted planning units.

Trawlers:
Trawling lines recorded by on-boat GPS devices were obtained by Edelist (2013) between the years 2009 – 2011. We combined this trawling data with extra trawling data mapped by the Ministry of fisheries (Israel Department of Fisheries 2012). Using this combined data we used the Kernel Density tool in ArcGIS (ESRI 2010) to calculate a magnitude per planning unit from the trawl line (polylines) features. The annual 2009 revenue value for trawling (US$6.67 million; Edelist et al. 2013) was overlaid and weighted to reflect this effort distribution.

Appendix S6.4. Description of economic activities and threats in Israel's Mediterranean waters.

Aquaculture:

There is currently only one approved aquaculture farm in Israel's Mediterranean waters off the coast of Ashdod. Currently the only species farmed is Sea bream *Sparus aurata.* The existing aquaculture facilities comprise an area of ~14 km^2 (Department of Fisheries & Aquaculture 2013). These data was provided by the Ministry of Fisheries and Aquaculture, State of Israel Ministry of Agriculture & Rural Development (Department of Fisheries & Aquaculture 2013). These areas are locked into the "Economic Zone".

Current protected areas: There is currently one marine protected area at the northern border of Israel "Rosh Hanikra" which is controlled and monitored by The Israel Nature and Parks Authority. This is an area of 11.4 km^2 and was locked in as a reserve into all conservation plans. These spatial GIS data were provided by The Israel Nature and Parks Authority (2012).

Desalination plants:

Spatial data were provided by the Israel Ministry of Interior from the National Master plan of Israel (Tama 34b).

Diving:

There are 47 dive sites in Israel's Mediterranean Sea territorial waters. Of these, 34 are shipwreck sites and 13 are natural dive sites. Data were provided by Feder (2012).

Exploration safety zones:

Active hydrocarbon drilling platforms are required to implement a 500 m radius around the platform as a safety area following the International Maritime Organization Safety Zone Resolution A.671 (1989; IMO 2013). This area prevents access of these waters to all other activities such as diving, fishing and scientific research. In the territorial waters of Israel only one active drilling area around the Mari B platform has these restrictions. We have locked out this whole planning unit (1 km^2) from our analysis as no activity is permitted inside this area.

Military areas (Fire Zones):

There are several military areas or divisions within the Mediterranean Sea of Israel (see "Fire Zones" in Fig. 6.3). Most of these allow access to fishermen however there are two areas (see "Military areas" in Fig. 6.3) that restrict entry 1) area near Atilt, and 2) a buffer zone adjacent to the

Gaza border (~0.5 km). These areas perhaps act as de-facto marine reserve areas where little exploitation of resources is occurring. A comparative study was done (Sonin 2008) and it was found that these military controlled areas harbour fish species with greater biomass and diversity. Spatial GIS data for military areas were provided by The Israel Nature and Parks Authority (2012).

Shipping lanes: Here we geo-referenced and digitised shipping lanes using ArcGIS 10 (ESRI 2010) from a map provided by the Society for the Protection of Nature in Israel (SPNI) Open Landscape Institute (OLI).

Pipelines

Data of offshore oil and gas pipe lines were provided by the Israel Ministry of Interior from the National Master plan of Israel (Tama 37). These data included; 1) existing pipelines and 2) planned pipelines for the transmission of natural gas.

Table S6.1. Conservation targets for each 166 conservation feature (153 fish species, 2 sea turtle species, one cetacean and 10 geomorphologic features) were set using the IUCN red listings for species; 10% target was set for all species, 15% target was set for IUCN “Vulnerable” listed species, 20% target for IUCN “Endangered” species and for “Endangered” that have <1% of the area of the territorial waters a target of 50% was set. A 5% target was set for all geomorphologic features and features with <1% of area were given a 10% target. Zones: Economic Zone (general) effectiveness = 0%; No-Take Zone effectiveness = 100% Benthic Protection zone effectiveness = Fish base values for fishes, Turtles (Donlan et al. 2010), Marine mammals (Coll et al. 2010), geomorphologic structures (we set 50% due to unknown zone effectiveness) Exploration Zone effectiveness = 3 scenarios 25%, 50% and 75%.

Conservation features	**Vulnerability**	**Benthic Protection Zone effectiveness**	**Overall Targets (IUCN)**
Fish Species			
Aidablennius sphynx	Low vulnerability (16 of 100; FishBase)	84%	10%
Alectis alexandrinus	Moderate to high vulnerability (50 of 100; FishBase)	50%	10%
Anthias anthias	Moderate vulnerability (38 of 100; FishBase)	62%	10%
Apogon imberbis	Low vulnerability (15 of 100; FishBase)	85%	10%
Argentina sphyraena	Moderate vulnerability (36 of 100; FishBase)	64%	10%
Argyrosomus regius	Very high vulnerability (79 of 100; FishBase)	21%	10%
Ariosoma balearicum	Low to moderate vulnerability (31 of 100; FishBase)	69%	10%
Arnoglossus kessleri	Low vulnerability (21 of 100; FishBase)	79%	10%
Arnoglossus laterna	Moderate vulnerability (36 of 100; FishBase)	64%	10%
Atherina boyeri	Moderate vulnerability (43 of 100; FishBase)	57%	10%
Auxis rochei	Low to moderate vulnerability (34 of 100; FishBase).	66%	10%
Balistes capriscus (old name Balistes carolinensis)	Low to moderate vulnerability (32 of 100; FishBase).	38%	10%
Blennius ocellaris (ocelatus)	Low to moderate vulnerability (30 of 100; FishBase).	70%	10%
Boops boops	Moderate vulnerability (41 of 100; FishBase).	59%	10%
Bothus podas	Moderate to high vulnerability (51 of 100;	49%	10%

	FishBase).		
Belone belone	Moderate vulnerability (39 of 100; FishBase).	61%	10%
Callionymus risso	Low vulnerability (16 of 100; FishBase)	84%	10%
Capros aper	Low vulnerability (16 of 100; FishBase).	84%	10%
Caranx crysos	Low to moderate vulnerability (34 of 100; FishBase).	66%	10%
Caranx rhonchus	Moderate vulnerability (36 of 100; FishBase).	64%	10%
Carcharhinus obscurus (catgalligious)	Vulnerable – IUCN	12%	15%
Cepola macrophthalma (was Cepola rubescens)	Moderate to high vulnerability (50 of 100; FishBase)	50%	10%
Chelidonichthys lucernus (was lucerna and Triga lucerna)	High vulnerability (58 of 100; FishBase)	42%	10%
Chelon labrosus	Least Concern – IUCN	37%	10%
Chlorophthalmus agassizii	Low to moderate vulnerability (32 of 100; FishBase)	68%	10%
Chromis chromis	Moderate vulnerability (35 of 100; FishBase).	65%	10%
Chromogobius quadrivittatus	Low vulnerability (20 of 100) Least Concern – IUCN (Endemic)	80%	10%
Chromogobius zebratus	Least Concern – IUCN (Endemic) Low vulnerability (15 of 100)	85%	10%
Citharus linguatula	Moderate vulnerability (36 of 100; FishBase).	64%	10%
Clinitrachus argentatus	Low vulnerability (10 of 100; FishBase).	90%	10%
Coelorhynchus coelorhynchus	High vulnerability (62 of 100; FishBase).	38%	10%
Conger conger	Very high vulnerability (86 of 100; FishBase).	14%	10%
Coris julis	High vulnerability (60 of 100) Least Concern - IUCN	40%	10%
Coryphoblennius galerita	Low vulnerability (20 of 100; FishBase).	80%	10%
Dactylopterus volitans	Low to moderate vulnerability (31 of 100;	69%	10%

	FishBase).		
Dasyatis chrysonota (catgalligious)	Least Concern – IUCN High to very high vulnerability (70 of 100)	30%	10%
Dasyatis pastinaca (catgalligious)	Very high vulnerability (82 of 100; FishBase).	18%	10%
Deltentosteus quadrimaculatus	Low vulnerability (20 of 100; FishBase).	80%	10%
Dentex gibbosus	High vulnerability (60 of 100; FishBase).	40%	10%
Dentex macrophthalmus	Moderate to high vulnerability (51 of 100; FishBase).	49%	10%
Dentex maroccanus	Moderate to high vulnerability (47 of 100; FishBase).	53%	10%
Dicentrarchus labrax	Least Concern – IUCN; High vulnerability (57 of 100)	43%	10%
Diplodus annularis	Moderate vulnerability (42 of 100; FishBase).	58%	10%
Diplodus cervinus	High to very high vulnerability (69 of 100; FishBase).	31%	10%
Diplodus sargus	High vulnerability (63 of 100; FishBase).	37%	10%
Diplodus vulgaris	Low to moderate vulnerability (33 of 100; FishBase).	67%	10%
Echelus myrus	Moderate to high vulnerability (49 of 100; FishBase).	51%	10%
Echeneis naucrates	Moderate to high vulnerability (54 of 100; FishBase).	46%	10%
Echiodon dentatus	Least Concern – IUCN; Low vulnerability (10 of 100)	90%	10%
Enchelycore anatina	High vulnerability (59 of 100; FishBase).	41%	10%
Engraulis encrasicolus	Low vulnerability (14 of 100; FishBase).	86%	10%
Epinephelus aeneus	Near threatened – IUCN; Moderate to high vulnerability (52 of 100)	48%	10%
Epinephelus costae	High to very high vulnerability (66 of 100; FishBase).	34%	10%
Epinephelus marginatus	Endangered – IUCN; High to very high vulnerability (72 of 100)	28%	20%

Euthynnus alletteratus	Least Concern – IUCN; High vulnerability (57 of 100)	43%	10%
Gnathophis mystax	Moderate vulnerability (44 of 100; FishBase).	56%	10%
Gobius bucchichi	Least Concern – IUCN Low vulnerability (15 of 100)	85%	10%
Gobius cobitis	Moderate vulnerability (39 of 100; FishBase).	61%	10%
Gobius cruentatus	Low to moderate vulnerability (31 of 100; FishBase).	69	10%
Gobius fallax	Least Concern – IUCN Low vulnerability (22 of 100)	78%	10%
Gobius niger	Moderate vulnerability (38 of 100; FishBase).	62%	10%
Gobius pagenllus	Low vulnerability (19 of 100; FishBase).	81%	10%
Gouania willdenowi	Low to moderate vulnerability (27 of 100; FishBase).	73%	10%
Gymnothorax unicolor	Moderate to high vulnerability (50 of 100; FishBase).	50%	10%
Helicolenus dactylopterus	Moderate to high vulnerability (52 of 100; FishBase).	48%	10%
Hippocampus guttulatus	Low vulnerability (24 of 100; FishBase).	76%	10%
Hypleurochilus bananensis	Least concern – IUCN *Endemic Low to moderate vulnerability (26 of 100)	74%	10%
Lepadogaster candollii	Moderate vulnerability (38 of 100; FishBase).	62%	10%
Lepadogaster lepadogaster	Least concern – IUCN *Endemic	64% Moderate vulnerability (36 of 100)	10%
Lepidopus caudatus	Moderate to high vulnerability (54 of 100; FishBase).	66%	10%
Lepidotrigla cavillone	Low vulnerability (25 of 100; FishBase).	75%	10%
Lesuerigobius suerii	Low vulnerability (12 of 100; FishBase).	88%	10%
Lipophrys canevae	Low vulnerability (15 of	85%	10%

	100; FishBase).		
Lipophrys pavo changed to Salaria pavo	Low to moderate vulnerability (25 of 100; FishBase).	75%	10%
Lipophrys trigloides	Low vulnerability (24 of 100; FishBase).	76%	10%
Lithognathus mormyrus	Moderate vulnerability (40 of 100; FishBase).	60%	10%
Liza aurata	Least concern – IUCN Moderate to high vulnerability (50 of 100)	50%	10%
Liza ramada	Least concern – IUCN Moderate vulnerability (37 of 100)	63%	10%
Macrorhamphosus scolopax	Least concern – IUCN Low to moderate vulnerability (27 of 100)	73%	10%
Merluccius merluccius	High vulnerability (65 of 100; FishBase).	35%	10%
Microchirus ocellatus	Low vulnerability (25 of 100; FishBase).	75%	10%
Mugil cephalus	Least concern – IUCN Moderate vulnerability (42 of 100)	58%	10%
Mullus barbatus	Moderate vulnerability (36 of 100; FishBase).	64%	10%
Mullus surmuletus	Moderate vulnerability (37 of 100; FishBase).	63%	10%
Muraena helena	High to very high vulnerability (74 of 100; FishBase).	26%	10%
Mycetroperca rubra	Least concern – IUCN Very high vulnerability (81 of 100)	19%	10%
Oblada melanura	Low to moderate vulnerability (34 of 100; FishBase).	66%	10%
Oedalechilus labeo	Moderate vulnerability (40 of 100; FishBase).	60%	10%
Ophiodon barbatum	Low to moderate vulnerability (32 of 100; FishBase).	68%	10%
Pagellus acarne	Moderate vulnerability (43 of 100; FishBase).	57%	10%
Pagellus bogaraveo	High to very high vulnerability (70 of 100; FishBase).	30%	10%
Pagellus erythrinus	Moderate to high vulnerability (54 of 100; FishBase).	46%	10%
Pagrus	Moderate to high	53%	10%

coeruleostictus	vulnerability (47 of 100; FishBase).		
Pagrus pagrus	Endangered - IUCN High to very high vulnerability (66 of 100)	34%	20%
Parablennius gattorugine	Least concern – IUCN Low vulnerability (21 of 100)	79%	10%
Parablennius incognitus	Low vulnerability (14 of 100; FishBase).	86%	10%
Parablennius rouxi	Least concern – IUCN Low vulnerability (16 of 100)	84%	10%
Parablennius saguinolentus	Low to moderate vulnerability (30 of 100; FishBase).	70%	10%
Parablennius tentacularis	Low to moderate vulnerability (26 of 100; FishBase).	74%	10%
Parablennius zvonimiri	Low vulnerability (14 of 100; ; FishBase) *Endemic	86%	10%
Phycis blennoides	High vulnerability (58 of 100; FishBase).	42%	10%
Pomadasys incisus	Least concern – IUCN Moderate vulnerability (40 of 100)	60%	10%
Raja clavata (catgalligious)	Near threatened – IUCN Very high vulnerability (76 of 100)	24%	10%
Raja miraletus (catgalligious)	Least concern – IUCN Moderate to high vulnerability (51 of 100)	49%	10%
Raja montagui (catgalligious)	Least concern – IUCN High vulnerability (59 of 100)	41%	10%
Rhinobatos rhinobatos (catgalligious)	Endangered – IUCN High to very high vulnerability (68 of 100)	32%	20%
Sardina pilchardus	Moderate vulnerability (36 of 100; FishBase).	64%	10%
Sardinella aurita	Moderate vulnerability (36 of 100; FishBase).	64%	10%
Sardinella maderensis	Low to moderate vulnerability (33 of 100; FishBase).	67%	10%
Sargocentron rubrum	Low vulnerability (24 of 100; FishBase).	76%	10%
Sarpa salpa	Moderate vulnerability (41 of 100; FishBase).	59%	10%
Scartella cristata	Low vulnerability (23 of 100; FishBase).	77%	10%
Sciaena umbra	High vulnerability (64 of 100; FishBase).	36%	10%

Scomber japonicus	Least concern – IUCN Moderate to high vulnerability (46 of 100)	54%	10%
Scorpaena elongata	High to very high vulnerability (67 of 100; FishBase).	23%	10%
Scorpaena maderensis	Moderate vulnerability (36 of 100; FishBase).	64%	10%
Scorpaena notata	Moderate vulnerability (42 of 100; FishBase).	58%	10%
Scorpaena porcus	Moderate to high vulnerability (49 of 100; FishBase).	51%	10%
Scorpaena scrofa	High to very high vulnerability (68 of 100; FishBase).	32%	10%
Seriola dumerili	Moderate to high vulnerability (54 of 100; FishBase).	46%	10%
Serranus cabrilla	Moderate vulnerability (36 of 100; FishBase).	64%	10%
Serranus hepatus	Low to moderate vulnerability (31 of 100; FishBase).	69%	10%
Serranus scriba	Moderate vulnerability (38 of 100; FishBase).	63%	10%
Solea solea	Low to moderate vulnerability (35 of 100; FishBase).	65%	10%
Sparisoma cretense	Least concern – IUCN Moderate vulnerability (36 of 100)	64%	10%
Sparus aurata	Low to moderate vulnerability (35 of 100; FishBase).	65%	10%
Sphoeroides pachygaster	Vulnerable – IUCN Moderate to high vulnerability (47 of 100)	53%	15%
Sphyraena sphyraena	Moderate to high vulnerability (49 of 100; FishBase).	51%	10%
Spicara maena	Low to moderate vulnerability (33 of 100; FishBase).	67%	10%
Spicara smaris	Moderate vulnerability (39 of 100; FishBase).	61%	10%
Symphodus mediterraneus	Least concern – IUCN Low vulnerability (23 of 100)	77%	10%
Symphodus ocellatus	Least concern- IUCN *Endemic Low vulnerability (14 of	86%	10%

	100)		
Symphodus roissali	Least concern – IUCN Low to moderate vulnerability (31 of 100)	69%	10%
Symphodus tinca	Least concern – IUCN Moderate vulnerability (37 of 100)	63%	10%
Synodus saurus	Least concern – IUCN Low to moderate vulnerability (31 of 100)	69%	10%
Thalassoma pavo	Least concern – IUCN Moderate vulnerability (40 of 100)	60%	10%
Torpedo marmorata (catgalligious)	High to very high vulnerability (69 of 100; FishBase).	31%	10%
Torpedo torpedo (catgalligious)	High to very high vulnerability (65 of 100; FishBase).	35%	10%
Trachinotus ovatus (catgalligious)	Moderate vulnerability (38 of 100; FishBase).	62%	10%
Trachinus araneus (catgalligious)	Moderate vulnerability (42 of 100; FishBase).	58%	10%
Trachinus draco (catgalligious)	Moderate vulnerability (42 of 100; FishBase).	58%	10%
Trachurus mediterraneus	Moderate to high vulnerability (46 of 100; FishBase).	54%	10%
Trachurus trachurus	High vulnerability (56 of 100; FishBase).	44%	10%
Trichiurus lepturus	High vulnerability (57 of 100; FishBase).	43%	10%
Trigloporus lastoviza	Low to moderate vulnerability (32 of 100; FishBase).	68%	10%
Tripterygion delaisi	Low vulnerability (14 of 100; FishBase).	86%	10%
Tripterygion melanurus	Least concern – IUCN *Endemic Low vulnerability (10 of 100)	90%	10%
Tripterygion tripteronotus	Least concern – IUCN *Endemic Low vulnerability (13 of 100)	87%	10%
Umbrina cirrosa	Moderate vulnerability (40 of 100; FishBase).	60%	10%
Uranoscopus scaber	Moderate vulnerability (44 of 100; FishBase).	56%	10%
Xyrichthys novacula	Least concern – IUCN Moderate vulnerability (36 of 100)	74%	10%

Zebrus zebrus	Least concern – IUCN *Endemic Low vulnerability (13 of 100)	87%	10%
Zeus faber	Moderate vulnerability (41 of 100; FishBase).	59%	10%
Sea turtle species			
Caretta caretta	Endangered – IUCN 74% (bycatch fishing threat score from expert based survey in Donlan et al. 2010)	26%	50%
Chelonia mydas	Endangered – IUCN 71% (bycatch fishing threat score from expert based survey in Donlan et al. 2010)	29%	50%
Cetaceans			
Tursiops truncatus (Mediterranean sea sub-population	Vulnerable – IUCN 60% (threat analysis by Coll et al. 2010)	60%	15%
Geomorphological features			
Shallow rocks (25.31 km^2)	>1% of territorial waters	50%	10%
Kukar ridges (245.14 km^2)		50%	5%
Kukar bustan (11.12 km^2)	>1% of territorial waters	50%	10%
Deep kukar ridges (188.64 km^2)		50%	5 %
Continental shelf silt (233.99 km^2)		50%	5 %
Continental shelf sand (2,040.98 km^2)		50%	5 %
Continental ridges (35.97 km^2)	>1% of territorial waters	50%	10%
Big canyons (31.80 km^2)	>1% of territorial waters	50%	10 %
Continental slope and canyons (585.23 km^2)		50%	5%
Deep Sea (561.07 km^2)		50%	5%

Table S6.2. A list of 159 native fish species (cartilaginous fishes and bony fishes) in Israel's territorial waters complied from eight publications with ranges and locations checked against the Hebrew University Collection. Six species were removed from this list and are marked by a * (see Appendix S6.2 for further information), therefore this study used a total of 153 species.

Native Fish species	**Golani et al. (2007)** Site 1	**Golani et al. (2007)** Site 2	**Golani et al. (2007)** Site 3	**Edelist (2013)**	**Diamant et al. (1986)**	**Goren & Galil (2001)**	**Levit (2012)**	**Spanier et al. (1989)**	**Lipsky (2012) South**	**Lipsky (2012) Center**	**Lipsky (2012) North**	**Stern (2010)**	**Range as documented in the Hebrew University Collection**
Aidablennius sphynx	X	X	X		X	X							RoshHaNikra - Palmachim
Alectis alexandrinus				X			X					X	Haifa - Ashqelon
Anthias anthias	X	X	X		X								Haifa - Yaffo
Apogon imberbis	X	X	X	X	X		X	X	X	X	X		RoshHaNikra - Hertziliya
Argentina sphyraena				X									Haifa-Hadera
Argyrosomus regius											X	X	Haifa-Yafo
Ariosoma balearicum				X			X					X	Haifa - Yaffo
Arnoglossus kessleri							X					X	Haifa, Yaffo
Arnoglossus laterna							X					X	Haifa - Ashdod
Atherina boyeri	X	X	X			X	X					X	Shikmona - Gaza
Auxis rochei												X	Haifa
Balistes capriscus (old name Balistes carolinensis)				X			X	X	X	X		X	Akko - Ashdod
Blennius ocellaris (ocelatus)				X			X						Haifa - Ashkelon
Boops boops	X	X	X	X	X	X	X	X				X	Haifa bay - Gaza
Bothus podas	X	X	X		X		X					X	Akko - Gaza
Belone belone												X	Gaza - Haifa bay
Callionymus risso					X		X						Yaffo
Capros aper				X									Haifa - Gaza
Caranx crysos	X	X	X	X	X	X	X	X				X	Akko - Gaza

Caranx rhonchus												X	Haifa - Gazza
Carcharhinus obscurus (catgalligious)				X									Yaffo - Ashdod
Cepola macrophthalma (was Cepola rubescens)							X						Haifa Bya
Chelidonichthys lucernus (was lucerna and Triga lucerna)							X					X	Haifa - Ashkelon
Chelon labrosus	X	X	X		X								RoshHaNikra - Tel Aviv
Chlorophthalmus agassizii				X			X						Haifa - Ashdod
Chromis chromis	X	X	X					X	X	X	X		Akko - Sdot Yam
Chromogobius quadrivittatus	X	X	X		X								RoshHaNikra - Mikhmoret
Chromogobius zebratus	X	X	X								X		RoshHaNikra - Neve Yam
Citharus linguatula				X			X					X	Haifa - Ashdod
Clinitrachus argentatus	X	X	X		X	X							Shiqmona - Mikhmoret
Coelorhynchus coelorhynchus				X									Haifa - Ashdod
Conger conger				X			X					X	Haifa - Ashdod
Coris julis	X	X	X		X	X		X	X				Full Coastline
Coryphoblennius galerita	X	X	X		X	X							Shiqmona - Michmoret
Dactylopterus volitans							X	X				X	Akko - Yaffo
Dasyatis chrysonota (catgalligious)							X						
Dasyatis pastinaca (catgalligious)				X			X	X	X	0	0	X	Haifa - Nakdiman
Deltentosteus quadrimaculatus							X					X	Ashkelon -Hadera
Dentex gibbosus							X					X	Haifa - Jaffo
Dentex macrophthalmus				X				X					Akko - Yaffo

Dentex maroccanus							X						Haifa - Ashkelon
Dicentrarchus labrax						X							Haifa - Zikim
Diplodus annularis				X			X					X	Haifa - Jaffo
Diplodus cervinus	X	X	X	X	X		X		X	X		X	Akko - Yaffo
Diplodus sargus	X	X	X	X	X	X	X	X	X	X	X	X	Full Coastline
Diplodus vulgaris	X	X	X	X	X		X	X	X	X	X	X	Akko - Palmachim
Echelus myrus				X			X					X	Nahariya - Ashdod
Echeneis naucrates				X			X					X	Akko - Gaza
Echiodon dentatus							X						Hadera - Ashdod
Enchelycore anatina										X			Yaffo
Engraulis encrasicolus				X			X		X			X	Haifa - Gaza
Epinephelus aeneus	X	X	X	X	X	X	X		X			X	Full Coastline
Epinephelus costae	X	X	X						X				Akko - Gaza
Epinephelus marginatus	X	X	X		X	X			X	X			Full Coastline
Euthynnus alletteratus								X				X	Akko - Yaffo
*Eutrigla gurnardus**												X	
Gnathophis mystax							X						Haifa - Ashdod
Gobius bucchichi	X	X	X		X	X				X			RoshHaNikra - Mikhmoret
Gobius cobitis	X	X	X		X	X							Full Coastline
Gobius cruentatus	X	X	X		X					X			Haifa bay - Ceasarea
Gobius fallax									X		X		
Gobius niger	X	X	X		X		X					X	Akko - Gaza
Gobius pagenllus	X	X	X		X	X					X		Full Coastline
Gouania willdenowi			X										RoshHaNikra
Gymnothorax unicolor	X	X	X		X								Haifa - Netanya
Helicolenus dactylopterus				X									Hertziliya - Ashdod
Hippocampus guttulatus							X						Haifa - Yaffo

Hypleurochilus bananensis	X	X	X										Akko - Michmoret
Lepadogaster candollii			X										RoshHaNikra
Lepadogaster lepadogaster			X										RoshHaNikra
Lepidopus caudatus				X									Nahariya - Ashdod
Lepidotrigla cavillone				X			X					X	Haifa bay - Ashkelon
Lesuerigobius suerii				X			X						Haifa - Yaffo
Lipophrys canevae	X	X	X		X	X							RoshHaNikra - Caesarea
Lipophrys pavo	X	X	X		X								Nahariya - Ashkelon
Lipophrys trigloides	X	X	X		X	X							RoshHaNikra - Bay Yam
Lithognathus mormyrus	X	X	X	X	X	X	X	X				X	Full Coastline
Liza aurata	X	X	X				X						Akko - Ashkelon
Liza ramada	X	X	X										Kishon - Gaza
Macrorhamphosus scolopax				X			X						Haifa - Gaza
Merluccius merluccius				X			X					X	Haifa - Ashkelon
Microchirus ocellatus				X			X						Haifa - Akhziv
*Microlipophrys nigriceps (none in Golani records) **					X								
Mugil cephalus	x	X	X		X								Akko - Hadera
Mullus barbatus				X			X					X	Akko - Ashdod
Mullus surmuletus	X	X	X	X	X	X	X	X			X	X	Haifa - Palmachim
Muraena helena	X	X	X					X					RoshHaNikra - Netanya
Mycetroperca rubra	X	X	X						X	X	X		RoshHaNikra - Tel Aviv
Oblada melanura	X	X	X	X	X			X	X		X		Full Coastline

Oedalechilus labeo	X	X	X		X	X							RoshHaNikra - Mikhmoret
Ophiodon barbatum				X			X						Haiffa - Yaffo
Pagellus acarne				X			X	X				X	Haifa - Tel Aviv
Pagellus bogaraveo				X									Hadera - Ashdod
Pagellus erythrinus				X			X	X				X	Haifa bay - Ashdod
Pagrus coeruleostictus				X			X	X		X		X	Akko - Yaffo
Pagrus pagrus							X	X					Haifa - Kishon
Parablennius gattorugine	X	X	X			X							Shikmona - Michmoret
Parablennius incognitus	X	X	X		X	X							Akko - Michmoret
Parablennius rouxi									X	X			
Parablennius saguinolentus	X	X	X		X	X							RoshHaNikra - Palmachim
Parablennius tentacularis										X			Rosh Hanikra
Parablennius zvonimiri	X	X	X		X	X			X	X			RoshHaNikra - Mikhmoret
Phycis blennoides							x						Haifa bay - Ashdod
Pomadasys incisus	X	X	X	X	X		X					X	Haifa bay - Gaza
Raja clavata (catgalligious)				X			X		X				Hadera - Ashdod
Raja miraletus (catgalligious)				X			X					X	Haifa - Palmachim
Raja montagui (catgalligious)												X	
Remora remora *								X					Yaffo
Rhinobatos rhinobatos (catgalligious)				X			X					X	Haifa - Tel Aviv
Rhinobatus cemiculus (catgalligious) *												X	Haifa - Ashkelon
Sardina pilchardus				X			X					X	Akko - Ashdod

Sardinella aurita				X			X	X				X	Akko - Askelon
Sardinella maderensis												X	Akko - Yaffo
Sargocentron rubrum	X	X	X	X			X	X	X	X	X	X	Full Coastline
Sarpa salpa	X	X	X		X							X	Haifa - Gaza
Scartella cristata	X	X	X		X	X							RoshHaNikra - Ashqelon
Sciaena umbra	X	X	X		X			X	X				Haifa - Gaza
Scomber japonicus				X			X					X	Full Coastline
Scomber scombrus *												X	
Scorpaena elongata				X			X						Akko - Netanya
Scorpaena maderensis	X	X	X		X	X	X		X	X	X		Shiqmona - Ashdod
Scorpaena notata				X			X						Akko - Yaffo
Scorpaena porcus	X	X	X		X	X		X					Full Coastline
Scorpaena scrofa						X		X					Akko - Ashkelon
Seriola dumerili	X	X	X	X	X		X		X			X	Akko - Yaffo
Serranus cabrilla				X			X	X		X	X		Akko - Ashkelon
Serranus hepatus				X			X					X	Haifa bay - Ashdod
Serranus scriba						X		X			X	X	Akko - Gaza
Solea solea	X	X	X				X					X	Haifa - Gaza
Sparisoma cretense	X	X	X		X			X	X	X			Akko - Asheklon
Sparus aurata				X			X	X				X	Haifa - Gaza
Sprattus sprattus *												X	
Sphoeroides pachygaster				X									Haifa - Ashdod
Sphyraena sphyraena				X			X	X				X	Haifa - Gaza
Spicara maena				X			X	X				X	Haifa - Ashdod
Spicara smaris				X			X	X				X	Haifa - Ashdod
Symphodus mediterraneus								X					Rosh - Haifa bay
Symphodus ocellatus	X	X	X		X								Haifa - Gaza

Symphodus roissali	X	X	X		X	X							Akko - Gaza
Symphodus tinca	X	X	X					X					Akko - Gaza
Synodus saurus				X			X	X			X	X	Haifa bay - Ashkelon
Thalassoma pavo	X	X	X		X	X		X	X	X	X		Akko - Gaza
Torpedo marmorata (catgalligious)							X					X	Akko - Ashdod
Torpedo torpedo (catgalligious)				X			X					X	Akko - Gaza
Trachinotus ovatus (catgalligious)	X	X	X										Akko - Gaza
Trachinus araneus (catgalligious)							X					X	Akko - Yaffo
Trachinus draco (catgalligious)	X	X	X		X		X					X	Akko - Gaza
Trachurus mediterraneus				X			X					X	Akko - Yaffo
Trachurus trachurus				X			X	X				X	Hadera - Yaffo
Trichiurus lepturus				X			X						Akko - Yaffo
Trigloporus lastoviza				X	X		X						Haifa - Ashkelon
Tripterygion delaisi	X	X	X			X							Shiqmona - Mikhmoret
Tripterygion melanurus	X	X	X										Akko - Ashdod
Tripterygion tripteronotus	X	X	X		X								RoshHaNikra - Ashdod
Umbrina cirrosa						X	X						Akko - Zikim
Uranoscopus scaber				X			X					X	Akko - Yaffo
Xyrichthys novacula				X			X					X	Nahariya - Yaffo
Zebrus zebrus	X	X	X			X							RoshHaNikra - Mikhmoret
Zeus faber				X			X						Akko - Ashdod

Table S6.3. Calculation of the value of Israel's oil and gas reserves using annual average prices from 2012. The crude oil (petroleum) annual average price for 2012 was 404.52 NIS per bbl (World Bank). The natural gas annual average price for 2012 was 399.33 NIS per thousand cubic metres (International Monetary Fund). The conversion rate was US$1 = 3.86 NIS (annual average conversion rate 2012; IMF)

	Reserves (Varshavsky 2012)	**Conversion into monetary values** **NIS billion (US$ million)**
Gas reserves	Proved = 278 Bcm	111.01 (28.76)
	Contingent = 522 Bcm	208.45 (54.00)
	Prospective = 680 Bcm	271.54 (70.35)
Total	1,480 Bcm	591.01 (153.11)
Oil reserves	Contingent = 230 MMbbl	93.04 (24.10)
	Prospective = 1,400 MMbbl	566.33 (146.72)
Total	1,630 MMbbl	659.37 (170.82)
Total		**1,250.28 (323.93)**

*MMbbl = one million barrels; Bcm = billion cubic meters

Table S6.4. Nine species that had the 5% conservation zone (no-take) constraint removed in order for the planning scenario to reach biodiversity targets. The spatial distribution of these species overlapped with the opportunity cost layer for hydrocarbon extraction.

Nine Species
Echiodon dentatus
Enchelycore anatina
Dasyatis chrysonota
Parablennius incognitus
Parablennius rouxi
Argyrosomus regius
Auxis rochei
Raja montagui
Tursiops truncatus

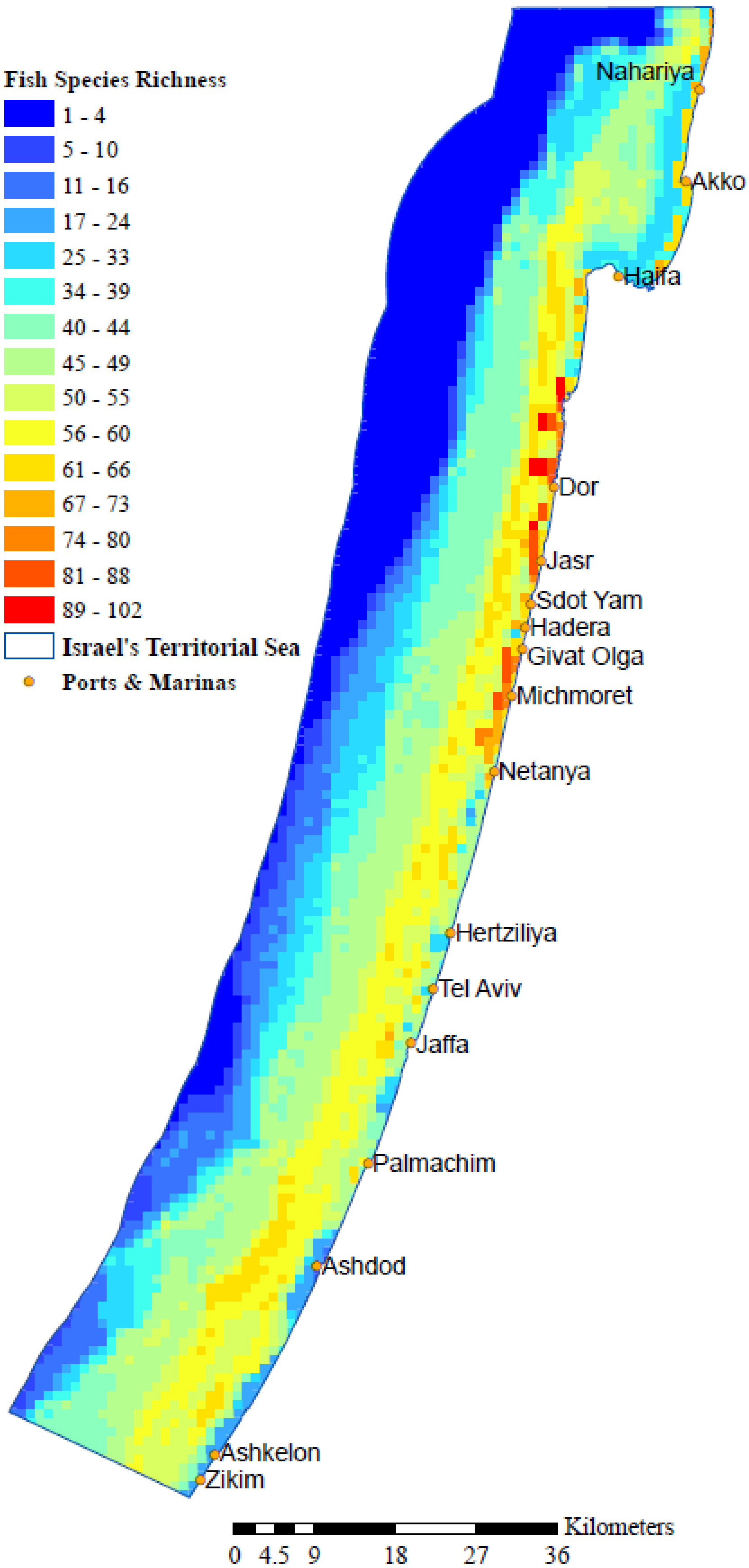


Figure S6.1. Species richness of 153 native fish species compiled from available studies and the Hebrew University of Jerusalem Fish Collection (for detailed information see Table S6.2)

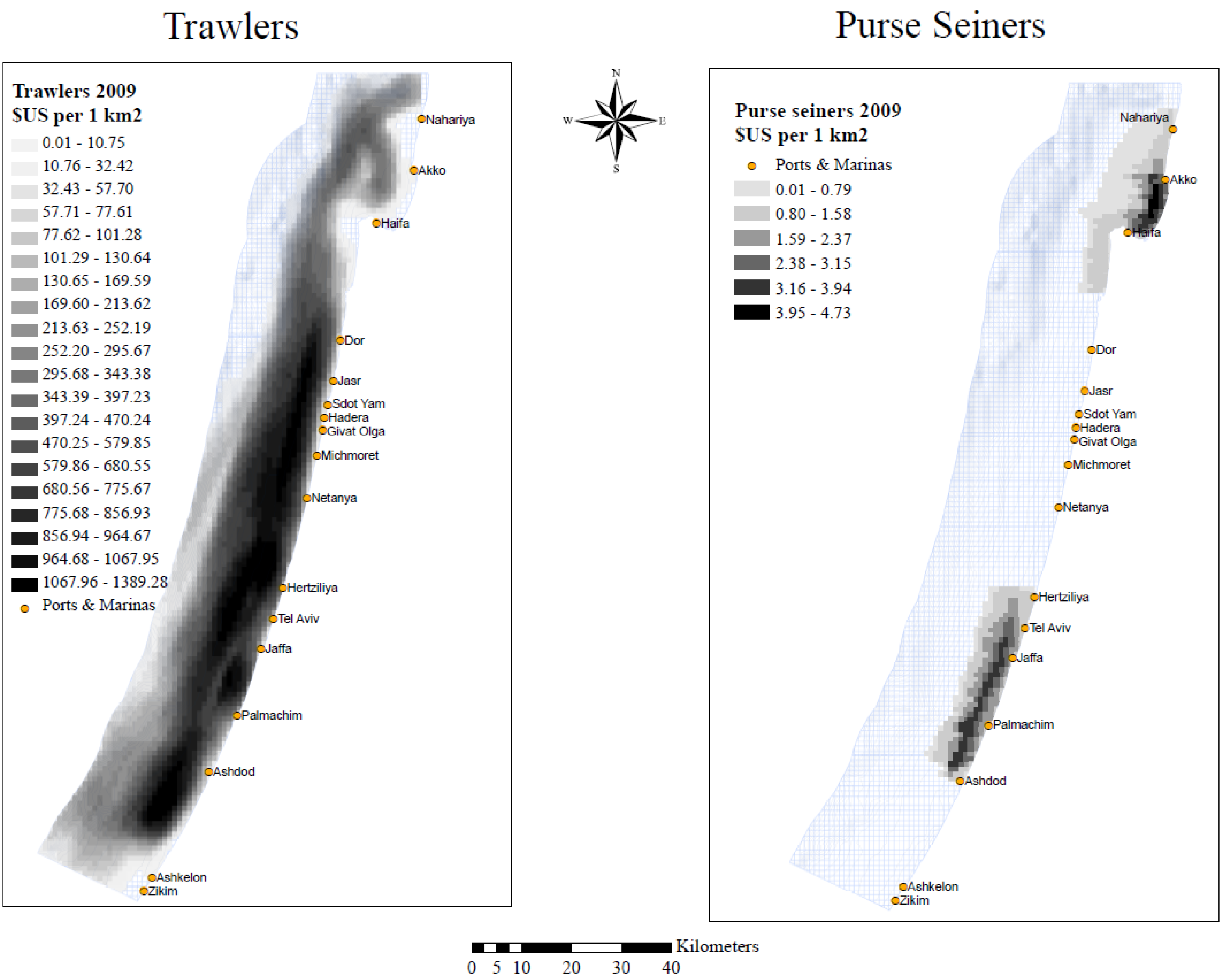


Figure S6.2. Fishing effort from trawlers and purse seiners in Israel's territorial waters of the Mediterranean Sea.

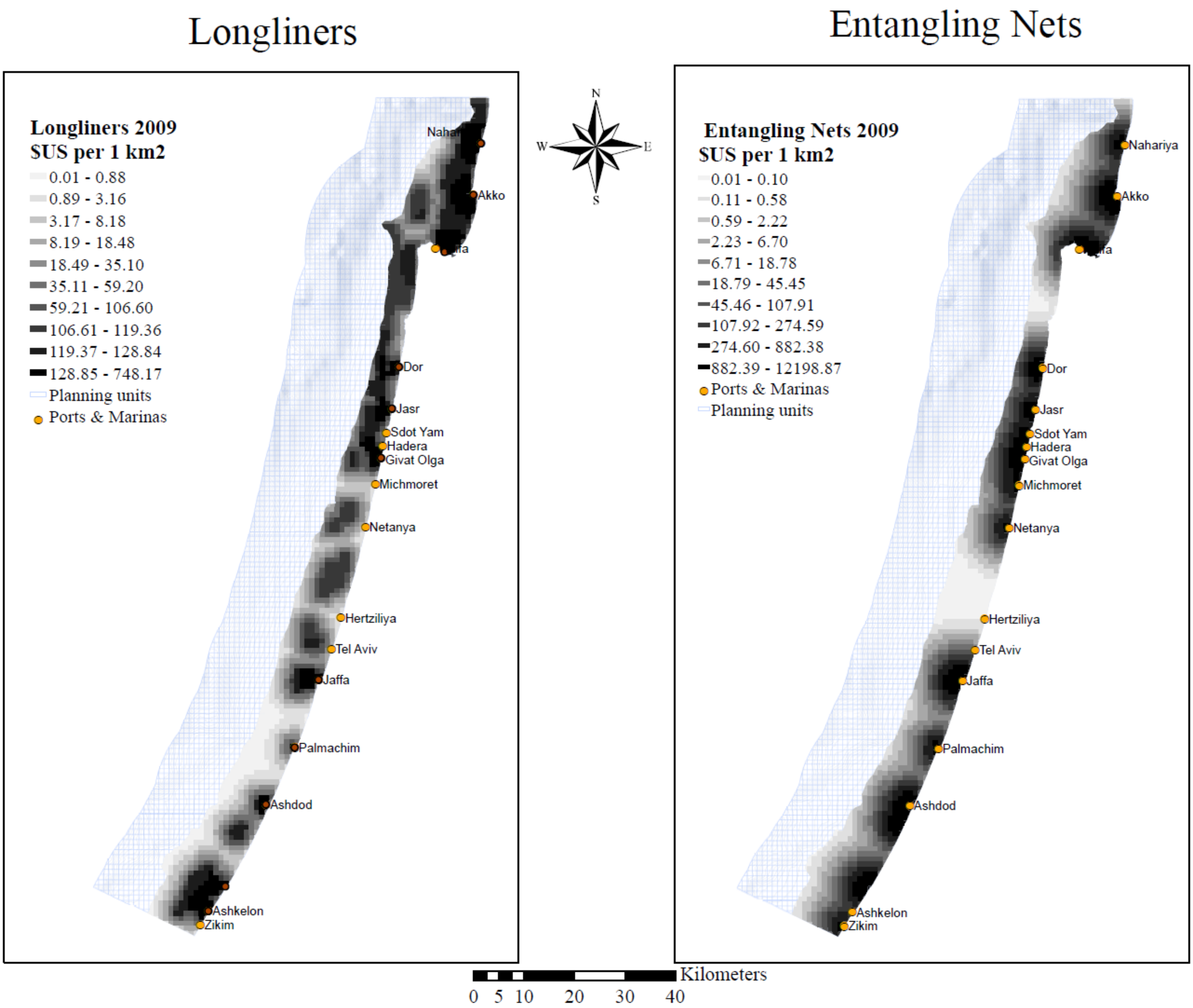


Figure S6.3. Fishing effort from long liners and entangling nets in Israel's territorial waters of the Mediterranean Sea.

Chapter 7

Conclusions

Dor Beach, Israel. Photo credit: S.Kark

7.1 Overview

This final chapter synthesises my thesis and explores the contributions of my findings to the theory of conservation prioritisation. I will discuss here how the previous five research chapters have the potential to influence the future of conservation decision making in the marine realm. Specifically, I highlight three key themes that arise from this thesis. I further address the practical contributions and advancements of this work for marine conservation planning in the Mediterranean Sea. Finally, I discuss the challenges of large-scale conservation planning and the future research directions resulting from my thesis.

7.2 Scientific advancements and conservation implications of this research

This thesis advances the theory of conservation prioritisation by building upon the framework of systematic conservation planning (Margules & Pressey 2000; Pressey & Bottrill 2009). Here, I propose seven additional steps for systematic conservation planning that emerge from the chapters of this thesis (Fig. 7.1). These steps address the need for conservation prioritisation strategies that better suit characteristics of the marine realm (Hockey & Branch 1994; Carr et al. 2003; Leslie 2005; Norse & Crowder 2005; Roberts 2005; Kearney et al. 2013). Specifically, I present steps that highlight three key themes of this thesis in which marine conservation prioritisation can be improved: *efficiency* (**Chapter 2** and **3**), *adequacy* **-** species protection (**Chapter 4** and **5**) and *implementation success* (**Chapter 6**). Below, I synthesise my research chapters and outline their contributions within each key theme.

7.2.1. Improving efficiency

This thesis has developed approaches that improve the efficiency of conservation plans. It is well known that conservation funds are largely inadequate for global biodiversity conservation (Waldron et al. 2013). Marine conservation is viewed as a lower priority than terrestrial conservation, and receives even less funding (Levin & Kochin 2004; Norse & Crowder 2005). Thus, with the limited resources that are available, it is important to develop efficient decisions. This thesis contributes two profitable strategies in **Chapter 2** and **Chapter 3** both based on collaboration. This research is particularly valuable for addressing the allocation of limited conservation resources in the marine realm.

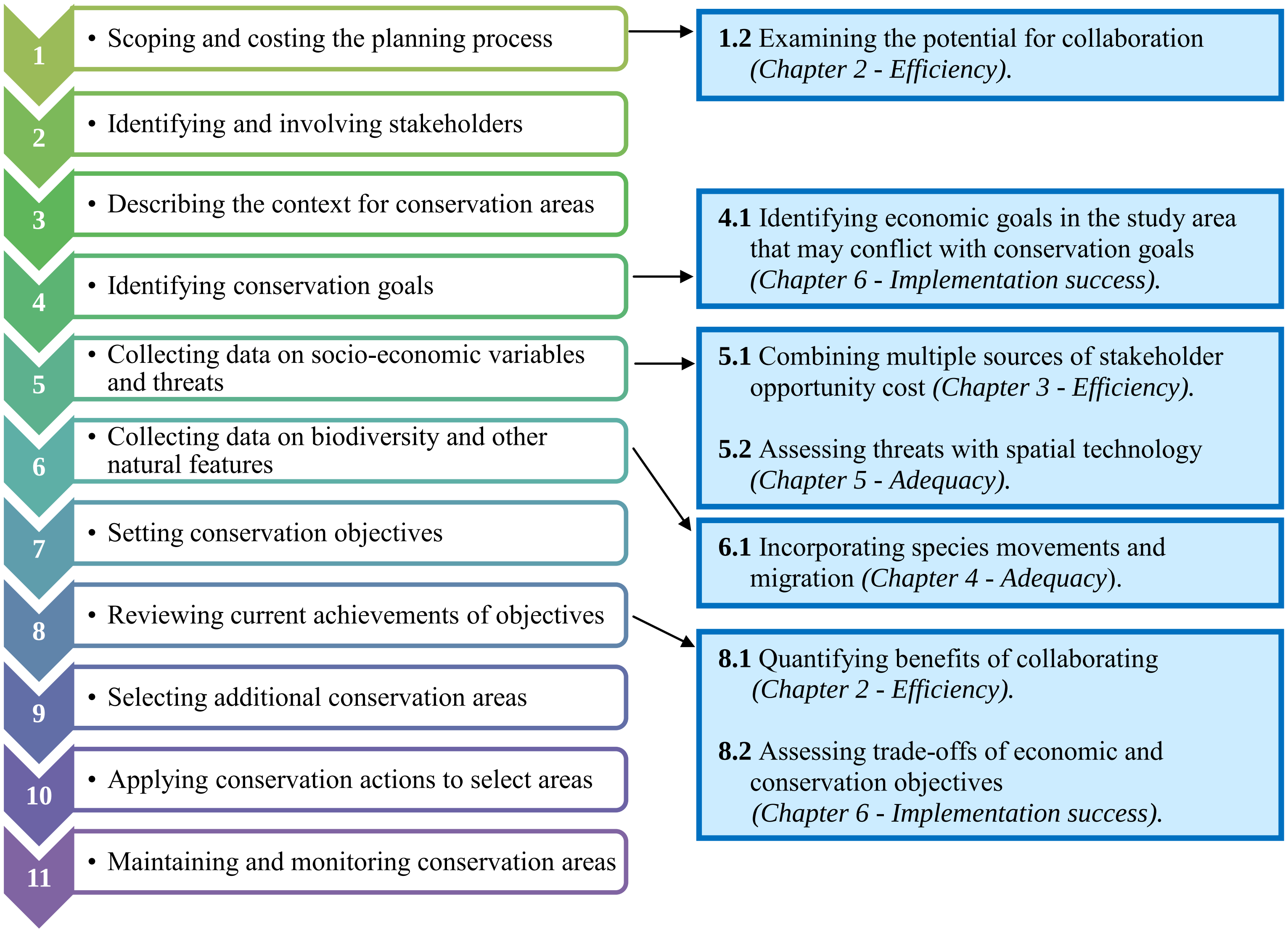


Figure 7.1. Framework for systematic conservation planning (Margules & Pressey 2000; Pressey & Bottrill 2009) with proposed steps (in blue) that emerge from the chapters in this thesis. Each additional step links to a chapter of this thesis and a key principle of systematic conservation planning as proposed by this work; efficiency, adequacy, and implementation success.

Chapter 2 explored how incorporating between-country collaboration can improve the efficiency of marine conservation planning. This study was the first to explicitly quantify the benefits of collaboration in the marine realm and shows that collaboration can have substantial monetary savings. The same conservation targets were achieved with reduced opportunity cost when countries of the Mediterranean Sea engaged in either full (all countries collaborated) or partial collaboration (countries from each continent collaborated). I discovered that the benefits of collaboration were unequal between countries. Some countries of the Mediterranean such as Spain, Tunisia and Italy incurred large cost savings, whereas other countries had losses such as France, Libya and Malta. This finding contributes to the very recent concept of equity in conservation planning (Halpern et al. 2013). Whereby I demonstrated the quantification of economic equity and highlight the importance of being aware of inequitable solutions and how this may relate to implementation challenges for conservation in the Mediterranean.

This approach has the potential to be replicated in other marine regions of the world such as the Caribbean, Coral Triangle, Eastern Africa, or Black Sea where marine waters are shared by multiple nations and conservation is complex. Furthermore, it could be applied within other marine settings such as collaboration between States around a lake (e.g., Lake Michigan) or river (e.g., Murray Darling River). I expect that my findings from the Mediterranean Sea that between-country collaboration will provide substantial monetary savings will be evident in most other marine settings. This is because terrestrial studies also indicate higher efficiency and advantages when collaborating (Strange et al. 2006, Bladt et al. 2009; Kark et al. 2009; Moilanen et al. 2013). However, my finding where some countries benefited from collaboration more than others may change, because the opportunity cost layer and species distribution used in the Mediterranean Sea analysis were highly heterogeneous. Other regions with a more homogenous spread of cost and species may provide different results where collaboration benefits are more equitable between the countries. For decision makers this could mean that collaboration may be easier to establish in homogenous regions as the cost burden will be more equally shared. It may also mean that collaboration benefits may not be as large as a heterogeneous plans, and perhaps some type of partial collaboration will be more beneficial.

The results of this research have significant implications for marine conservation planning as it demonstrates how collaboration can provide the same conservation outcome (meeting the same biodiversity targets) but for less cost. The quantification of these benefits may provide an incentive for countries to collaborate. Currently there are very few successful multinational collaborative marine conservation efforts around the globe (Katerere et al. 2001; Mackelworth 2012). Those that

do exist are usually in the form of transboundary-parks, and rarely involve collaboration between more than two countries (Mackelworth 2012). This research brings to attention the role of collaboration in marine conservation planning as a cost-effective method with the potential for further examination.
An important aspect to integrate into this type of analysis is feasibility. While collaboration may be theoretically beneficial and save cost it may be hampered by social, economic and political barriers (Kark et al. 2009). Combining my approach in **Chapter 2** with a post-assessment of potential collaboration via proxies such as trade, tourism, political history and shared legislation, as described in **Appendix 1** (Levin et al. 2013) could be a step forward for incorporating feasibility into marine conservation planning.

Another cost-effective concept formulated on collaboration is that between marine sectors. In **Chapter 3,** I developed an approach for including cost when planning large-scale MPA networks that span many countries. I found that conservation plans that aimed to minimise the opportunity cost of multiple marine sectors (commercial fisheries, recreation fisheries and aquaculture operations) were more efficient than plans that minimised the opportunity cost of one sector (e.g., commercial fishing). These plans were more efficient because the opportunity cost of each marine sector decreased and the spatial requirements of the MPA network were reduced. Besides quantifiable benefits, this approach can also help facilitate the selection of feasible spatial options for reservation because it reduces the possibility of future conflicting interests (between stakeholders and marine conservation planners) by explicitly minimising impacts on stakeholders.

Chapter 3 highlights the importance of incorporating cost in marine conservation planning. Without cost information, the efficiency of conservation plans is largely unknown (Naidoo et al. 2006). Although cost has been applied in earlier work, it is often excluded in marine plans at large spatial scales (Micheli et al. 2013b). I showed that cost within a large-scale multinational setting can greatly influence the selection of priority conservation areas. I also demonstrated that area is a poor cost surrogate in marine realms. The nature of activities within marine systems is different to that of terrestrial systems, where anthropogenic threats and pressures are built around the coastal interface. Subsequently, the cost of conservation is intensified around the coast and needs to be depicted this way in conservation plans as I demonstrated in **Chapter 3**. Thus, the inclusion of cost in marine conservation plans should not be ignored, but representation (i.e. type of cost) and uncertainty should be carefully considered when interpreting results.

7.2.2. Improving adequacy (species protection)

Protecting marine species is challenging given the natural flow and connectivity of the marine realm (Carr et al. 2003; Wilkinson et al. 2004; Gaines et al. 2010a). Marine species are often shared between countries, some travelling large distances across international borders (Norse & Crowder 2005). In this thesis, I explored two case studies of sea turtle conservation in the Mediterranean Sea. Sea turtles are exemplar species to investigate given their worldwide threatened status, their vast migrations across the sea and their use of land and sea environments for different life stages (Casale & Margaritoulis 2010). In **Chapter 4**, I presented a study that explicitly incorporates information for the entire migratory life cycle of sea turtles. In **Chapter 5**, I demonstrated an application of satellite night light imagery to help predict nesting habitat and determine threats of Endangered sea turtles. While these case studies can help improve the protection of sea turtles, these methods could be relevant for many other species. These chapters highlight the importance of improving the protection of marine species to ensure resulting conservation outcomes deliver adequate protection (Margules & Pressey 2000).

Currently, most marine conservation plans fail to incorporate the spatial connectivity that is needed to adequately protect migratory species (Martin et al. 2007, Runge et al. 2014). In **Chapter 4** I demonstrated the application of breeding, feeding and migration information to help inform sea turtle conservation priorities across the whole Mediterranean Sea. The methodology applied in this study helps overcome some of the challenges of planning for migratory species across all life stages. Two key findings arose from this research for advancing conservation planning of mobile species. One, I discovered that sea turtle conservation priorities were sensitive to the inclusion of movement information, and even a small number of telemetry tracks substantially helped capture migratory links within priority areas. Without access to telemetry information conservation plans lack the ability to adequately protect species such as sea turtles that move across vast spaces and between land and sea habitats. Two, that in order to convey connectivity between sea turtle habitats in conservation plans, efforts should focus on collecting a heterogeneous sample of tracking data over quantity. This later finding is particularly valuable for guiding the future of telemetry studies and improving its application towards conserving migratory species.

My findings on loggerhead sea turtles of the Mediterranean Sea are expected to provide one example of a broader application for the protection of migratory species. Applying this method to other species such as sharks (Eckert & Stewart 2001), seabirds (Péron & Grémillet 2013), and cetaceans (Bailey et al. 2009), that also have substantial tracking data can help further determine the value of telemetry data for conservation planning. Perhaps, with similar studies that demonstrate the importance of telemetry data it will provide incentives for future cooperation with greater access to telemetry data. Future efforts should aim to extract all available telemetry data where possible, perhaps using monetary incentives or intellectual safeguards, and compile regional or global databases for the incorporation of migration information into conservation plans. I suggest that already established collaborative frameworks such as the EU, or the IUCN, could be a potential starting point.

A fundamental step of systematic conservation planning is determining threats to species. Conservation of species requires mitigation (management) of such threats. Margules and Pressey (2000) state that a better understanding of the distribution patterns of threats is needed to aid the allocation of limited conservation resources. In **Chapter 5** I addressed this research need by examining the threat of artificial night lights on nesting sea turtles. This study found that sea turtles are negatively related to night light intensity, positively related to the presence of cliffs along the beach, and are concentrated in darker sections of the coast. Besides identifying the threat of artificial night lights for sea turtles, this work also demonstrated the novel application of satellite imagery as a tool for predicting sea turtle habitat at a broad scale.

This work also highlights the potential for satellite-imagery and remote sensing tools to inform us about species threats, distributions and patterns across broad spatial scales. The methodology of this study can be applied to other species that are disturbed by artificial night lights. Such tools can provide the best knowledge at hand when field work locations are inaccessible or analysis is needed across large spatial scales. The next step forward from this work is the incorporation of night light imagery into conservation planning tools such as Marxan. The inclusion of these data as a threat (or cost layer) for selecting priority conservation beaches for sea turtle conservation is important for determining places that ensure the persistence of sea turtles. Although not given much attention, in **Chapter 4** I included night lights in the cost layer as a proxy for determining the level of anthropogenic disturbance to sea turtle nesting beaches. I used night lights instead of human population data because important disturbances such as ports, shopping districts and sea-side entertainment would otherwise not be considered. Thus, I assumed that management effort would be greater (and more costly) in more disturbed areas. These chapters emphasise a starting point for future exploration and incorporation of such tools and methods into broad-scale conservation planning and prioritisation.

7.2.3. Improving implementation success

Social and economic information influences the success of conservation planning (Stewart & Possingham 2005; Knight et al. 2008). Mounting research stresses that a major consideration for the future of marine conservation is the integration of biological interests with other social, economic and cultural factors (Stewart & Possingham 2005; Pomeroy et al. 2006; Charles & Wilson 2009; Parsons et al. 2014). This thesis addresses this research need by including anthropogenic and socio-economic data in systematic conservation planning in several ways. In **Chapter 3** the integration of multiple fishery stakeholders were combined into the opportunity cost. The main result from this chapter was that conservation priorities are greatly influenced by cost. I further expanded this example in **Chapter 6** to a complex case study of the whole of Israel's Mediterranean territorial waters, whereby multiple marine activities and economic objectives were included and trade-offs were explicitly quantified. Such work helps to deliver conservation outcomes where other users have greater acceptance and willingness to comply by regulations. Since Marxan is a decision support tool for initial planning discussions, explicitly accounting for other stakeholders in plans, many mean that implementation success is greater and action is not delayed.

The inclusion of multiple activities, stakeholders and their objectives are becoming increasingly important in conservation planning. In the marine realm, competing for ocean space is becoming a reality (Norse 2008; Agardy 2010). **Chapter 6** provided a case-study example of balancing activities within a crowded sea space. I found that when more marine activities are included in the planning process, conservation objectives were more costly and difficult to reach. The novelty of this study is that it includes numerous marine activities and for the first time explicitly includes marine hydrocarbon opportunity cost in a Marxan analysis. One of the challenges of this work was the translation of hydrocarbon information into an opportunity cost layer. As this was the first attempt to do so, and data were especially limited, these methods require refinement.

7.3 Practical contributions and advancements of this work for the Mediterranean Sea

This thesis advances systematic conservation planning and prioritisation is the Mediterranean Sea where previously no large-scale plans existed. I have provided three large-scale plans (**Chapter 2, 3, 4**) that encompass the entire Mediterranean Sea. An important contribution from these large-scale works was demonstrating the importance of between-country collaboration for conservation in the Mediterranean Sea and the monetary benefits it can deliver. I discovered that a plan where all countries of the Mediterranean Sea collaborate can save over two-thirds of the cost of a plan where each country acts independently (**Chapter 2**). Even partial collaboration in the Mediterranean region (e.g., between EU countries) can deliver huge savings. I also developed large-scale fishing opportunity cost maps for the region, and found that for less than 10% of the Sea's area, conservation targets (10% of the distribution of 77 threatened marine vertebrate species) could be achieved while incurring opportunity cost of less than 1% (**Chapter 3**). Another large-scale contribution is the identification of loggerhead sea turtle conservation priorities at a basin-wide scale (**Chapter 4**). My approach incorporated sea turtle breeding and feeding habitats, migratory connections via telemetry data and the conservation cost of reducing both sea and land threats. These priority areas are helpful for decision makers in determining where to invest for sea turtle protection in the Mediterranean Sea.

This thesis also contributes two case studies at a regional scale (**Chapter 5 & 6**). These case studies of Israel's Mediterranean coastal waters have worldwide relevance, but are particularly applicable to countries of the Mediterranean Sea which share many common challenges. **Chapter 5** was developed closely with the Sea Turtle Rescue Centre of Israel's Nature and Parks Authority, and has helped provide direction for the management of sea turtles along Israel's Mediterranean coast by: 1) identifying persisting nesting beaches, 2) identifying night lights as a threat, and, 3) identifying other indicators of sea turtle habitat such as cliffs, human population and infrastructure. The methodologies of this work are especially relevant for eastern Mediterranean countries (e.g., Lebanon, Turkey, Greece, Egypt), that have highly populated coastlines and support major nesting beaches of threatened sea turtles. I also presented a framework for helping achieve implementable marine conservation priorities within the Mediterranean Sea (**Chapter 6**). Using this framework I identified conservation priorities for Israel's territorial waters which explicitly incorporate multiple marine activities and seeks economic and conservation objectives. Given that Israel is currently striving to expand its network of marine protected areas this timely work can help influence the selection of areas for reservation. This research also highlights the importance of incorporating hydrocarbon exploration in conservation planning, which is an important newly-found resource for many Mediterranean countries (Shaffer 2011).

A secondary benefit that evolved along with this thesis was the formation of a Mediterranean group of scientists to ascertain the important gaps and limitations specific to the Mediterranean Sea (**Appendix 2**: Giakoumi et al. 2012). Since this group, large-scale planning efforts have increased in the region. The opportunity cost layers as developed in **Chapter 2** has been further applied in Giakoumi et al. (2013) to identify spatial priorities for the conservation of three key Mediterranean habitats; seagrass meadows (*Posidonia oceanica*), coralligenous formations and marine caves.

While the scarcity of data in the Mediterranean is a hurdle that still needs addressing (Levin et al. 2014), these broad scale examples across the entire Mediterranean Sea demonstrate a process that other countries around the Sea should embark upon with the refinement of better data. Conservation planning and prioritisation with this complex sea can not only assist countries in reaching their CBD targets (protecting 10% of their waters), but also help address emerging threats such as hydrocarbon extraction.

7.4 Challenges and limitations of large-scale marine conservation prioritisations

Large-scale marine conservation prioritisations that span multiple countries present several challenges concerning the availability of data. One major limitation is the current absence of conservation cost information at large-scales. To help overcome this shortcoming this thesis developed surrogate opportunity cost layers. Poor data availability and the reliance on surrogate measures meant that scale precision was compromised with large planning units of 10 km^2 for the whole Mediterranean region. The lack of consistent socio-economic data across the Mediterranean region meant that surrogates were based on rather crude measurements. I attempted to keep data temporally consistent where possible, adjusting by GDP or PPP per country in the metrics (Chapter 2, Chapter 3, and Chapter 4). I used distance from port or coast, which is considered a representative measure of fishing effort and has been used in other prioritisation approaches (Cabrera & Omar 1997; Caddy & Carocci 1999; Gelchu & Pauly 2007; Stelzenmüller et al. 2008; Sala et al. 2002; Stewart et al. 2010; Giakoumi et al. 2011). To test the sensitivity of my results, I used two annual reported measurements of commercial fishing, GFCM and the SAUP (Chapter 3), and tested a number of different exponential decay rates for decreasing fishing effort with distance from ports (Chapter 2). For recreational fishing I assumed effort is based on human population size (Ban et al. 2009; Stallings 2009). However, several assumptions were made based on a limited number of publications for the number of days fishing per year per fisher, the average prices of a recreational catch, the proportion of the population fishing and the depth of fishing. Similarly, assumptions were made in Chapter 6 due to limited data and information. In Chapter 6 I assumed the distribution of petroleum across leases was homogenous when in reality it would be more heterogeneous. I trialled several scenarios to test the sensitivity of my findings where possible, nevertheless, the ability to validate resulting surrogate layers was impossible with the lack of detailed information on commercial and non-commercial fishing in the Mediterranean Sea.

Another data challenge was the deficiency of species information. I used IUCN data in Chapter 2 and Chapter 3, for species distribution information. These range maps are considered very coarse

(Levin et al. 2014), but are currently the best available species distribution data at large-scales. An improvement of this method is to collect individual point data and build species distribution models as done in Chapter 4 for the distribution of sea turtle foraging habitat. However, the scarcity of data around the Mediterranean Sea for each species (IUCN's 77 marine vertebrate species), and in the marine environment in general (Hendriks et al. 2006; Robinson et al. 2011), questions the ability of the models to be more informative than current IUCN data. Similarly, the 163 species I used for zoning Israel's waters (Chapter 6), ideally should all be modelled with the acquisition of better data. Other data shortages within this thesis included sea turtle tracking data in Chapter 4. Given the large expense and time to conduct telemetry studies, obtaining these data were difficult. Thus, I had a small sample size of 34 telemetry tracks, and suggest that the findings of this chapter would have greater significance with a higher sample of tracks. Setting species targets within this thesis was also considered arbitrary. I made attempts to base targets on the species threat status (e.g., IUCN red list), the size of the species distribution range, and purposely set low achievable targets (e.g., 10%). Setting conservation targets can also change the final conservation planning output. I also trialled several different targets as a sensitivity analysis (e.g., Chapter 3 I set at 10% and 30% target), however targets setting remains a debated topic (Tear et al. 2005; Carwardine et al. 2009; Rondinini & Chiozza 2010).

Marxan does not consider uncertainty in the input data. Therefore the quality of data used in the analysis is reflected in the generated results. The result of conservation plans associated with poor data quality is poor decision making. While there are obvious benefits for reducing uncertainty in conservation planning, we are often limited by the cost and time to accumulate such data (Halpern et al. 2006; Grantham et al. 2009). For chapters within this thesis which explored large scale conservation planning, the use of coarse surrogates for both species and cost layers was necessary. It is important to distinguish the quality of data underpinning surrogates. For this thesis, cost surrogates are likely to under estimate opportunity costs, because total fisheries removals are often not recorded around the globe, these include; illegally caught fish, unreported landings and recreational removals (Bray 2000; Agnew et al. 2009). It could be argued that if the methods of deriving such surrogates were altered, resulting Marxan outputs could be different. For example, in Chapter 3 I used two method of commercial fishing catch for the Mediterranean Sea, GFCM and SAUP. The GFCM data is more accurate given that it is collected specifically for the region and divided into geographical sub-regions due to vessel movement between countries (also used in Chapter 2). In comparison, the SAUP data is from a global analysis and calculated per country, thus reported landings in one country may have actually been caught in another. Nevertheless, these two cost layers highlight that different spatial outcomes can occur when underpinning your cost layer with different sources of data. It must be noted here, that large monetary discrepancies existed between the two cost methods, yet such values were not intended to be comparable between methods. Throughout this thesis I was interested in relative cost differences rather than absolute values. This is a major point to emphasise in this thesis, as all chapters are not intended to deliver sound conservation plans for implementation in the regions. Rather such analyses are meant to further build upon conservation planning methods and discover ways to improve the efficiency, adequacy and implementation success of conservation plans. Other research also states that improving data quality is not always the best strategy because there is a trade-off between effectiveness and efficiency (Hermoso et al. 2013; Grantham et al. 2009). Similarly, assessments based on limited data can provide useful information (Smith et al. 2006), and baseline starting points for the conservation planning process (i.e., stakeholder negotiations). Conservation planning as presented in this thesis should be seen as an adaptive process where the incorporation of better data and information can help reshape the resulting outputs (Smith et al. 2009).

Assumptions and limitations were also made by using Marxan in this thesis. One major assumption that was made throughout this thesis is that a relatively minor spatial reduction in fishing areas means a reduction in fishing income. However, this may not necessarily occur, as fishing effort may redisperse into new areas whereby the fishing income is maintained. Such re-dispersal is largely unknown, and predicting this type of information into a Marxan analysis is a challenge for future work. Another limitation by Marxan is the outputs it produces. The best solution output is considered to be a very good solution that meets the constraints of the system within a continuum of options. However, the spatial configuration of the best solution output does not reflect how often that area is selected in Marxan reiterations. In comparison, the selection frequency output does considers the number of times the planning unit was selected, but the output does not give you a complete marine reserve network. Combining information from both these types of outputs is helpful, yet can be confusing for decision makers. In this study I have used a combination of both types of outputs to describe results and planning outcomes. I suggest that future work aims to improve Marxan outputs to better aid decision making.

Despite the limitations of conservation cost and species data when working across large spatial scales, this thesis contributes conservation planning platforms for future expansion rather than detailed conservation work plans. Each research chapter aimed to further develop conservation planning applications and methods such as quantifying the role of collaboration and improving species protection. In this context I was more interested in relative cost values than absolute values. Similarly, the species data provides a baseline approach to build upon with the increasing availability of data. Where possible I undertook sensitivity analysis to account for possible variations.

7.5 Future research directions resulting from this thesis

This thesis has provided new strategies and methods for advancing marine conservation planning and prioritisation. In addition, such work has inspired further research questions. Below I outline three important topics for future exploration:

7.5.1 Refining conservation cost

One of the challenges of this research was the reliance on surrogates for marine conservation cost. This research has provided some ways of approaching this lack of data (e.g., **Chapter 2**), however there is a need to refine such approaches. When planning for conservation actions (e.g., marine reserves, restoration) with limited resources it is important to quantify the potential costs involved. Marine conservation plans which have incorporated cost often only use opportunity cost, as also done in this thesis. This is a much better approach than ignoring cost or using area as a cost, as revealed in **Chapter 2**, however, there is need for the development of more rigorous methods and frameworks. Particularly, approaches which consider a suite of costs, such as management cost, monitoring costs and transaction costs that are often ignored in conservation prioritisation. Such costs could cause the cost layer to be slightly more homogenous in marine systems, as monitoring, control, and surveillance costs increases with distance away from population centres. Incorporating these other types of cost, particularly at large-scales will require either a great synthesis of data or reliance on robust surrogate measures. The difficulty of estimating management cost in marine systems is that it is dependent on numerous factors such as the distance from coast, the potential for disturbances (e.g., areas with more activities may require more surveillance time), the size of the area requiring management (e.g., large vs. small MPAs). Another important aspect that has not been explored is the temporal variation of cost. Current studies include cost as a fixed factor (e.g., a one-time payment), but cost can greatly fluctuate, especially with the discovery of new marine resources (e.g., hydrocarbon discoveries discussed in **Chapter 6**) which have the potential to drive ocean spaces to higher values. I suggest that developing structured procedures and methods to assist the integration of this cost into conservation planning will encourage its application in further conservation planning initiatives. Hence, aiding the delivery of better informed conservation plans to decision-makers for weighing up foreseeable conservation cost.

7.5.2 Incorporating the social dimension

The new era of conservation planning is moving towards the inclusion of more social and economic considerations (Polasky 2008; Christensen et al. 2009; Ban et al. 2009; Weeks et al. 2010b; Adams et al. 2011; Klein et al. 2013; Levin et al. 2013; Ulloa et al. 2013). The success of implementing marine conservation efforts is often dependent on the inclusion of stakeholders and socio-economic factors during the planning and design process (Lundquist & Granek 2005; Warner & Pomeroy 2012).Thus, marine conservation planners are moving from simple approaches that map biodiversity priorities with the inclusion of one simple cost layer, to approaches that incorporate multiple other activities and features economic constraints, ecosystem services, and other social objectives (e.g., **Chapter 2** and **Chapter 6**). For examples, perusing equitable solutions for stakeholders (Halpern et al. 2013), including poverty alleviation goals (Gurney et al. 2014) and incorporating local marine tenure objectives (Weeks et al. 2010a). Thus marine conservation planning is merging into somewhat of a hybrid approach akin to marine spatial planning, but maintaining conservation as the primary objective. However, the inclusion of social considerations means that new challenges and uncertainties may arise, for example the unpredictable nature of socio-economic factors such as illegal fishing, the uncertainty around offshore hydrocarbon exploration and the unknown expansion of other industries as they move offshore (e.g., desalination plants, aquaculture, sand mining). I suggest that future research should aim to develop better approaches for amalgamating social and economic aspects into conservation plans and further examining and quantifying what this means for conservation outcomes; are these outcomes efficient and feasible?

7.5.3 Developing a dynamic ecosystem approach

A future quest for conservation planning is its evolution into a dynamic tool. In this thesis I treat many temporally variable processes as static features, for example cost. The implementation of conservation actions such as an MPA, involves dynamic considerations such as spill-over effects and the redistribution of fishing effort with the occupation of a marine reserve. Similarly, conservation actions also provide benefits for other species in the ecosystem. Current methods of conservation planning do not take into account this dynamic nature intrinsic to marine systems. However, currently there is a substantial lack of data for the inclusion of dynamic interactions. This thesis has already highlighted the lack of data in the marine realm for simple species distributions and opportunity cost layers. Hence, data which quantifies dynamic interactions within a food web or ecosystem context is scarce. Thus, modelling approaches will be heavily relied upon to integrate dynamic processes into conservation planning. One possible approach I suggest for future work is to explore the integration of ecosystem models into systematic conservation planning tools, for example feeding Marxan good-solution outputs into a simulation model such as Ecospace (e.g., Christensen et al. 2008); a spatial and temporal dynamic model used for exploring the impacts of MPA placement (Pauly et al. 2000). Moving towards a dynamic ecosystem approach can provide decision makers with better insight into the benefits and consequences of possible marine reserve networks, contributing to better conservation decisions.

7.6 Concluding remarks

This thesis tackles three ways in which systematic conservation planning can be improved: efficiency, adequacy (species protection) and implementation success. My work further develops methodological approaches for improving the ability to make informed conservation decisions. Prioritisation and planning approaches as demonstrated in this thesis support conservation decisions, but do not make them. The final outcome is determined by stakeholders and decision-makers who play a critical role in designing protected areas and choosing conservation priorities, and thus need to be equipped with the best tools to do so. An important conclusion from my research is that there is no "one solution" to planning the protection of biodiversity within a complex marine environment such as the Mediterranean Sea. Moreover, the socio-economic factors and biological data incorporated will ultimately guide and shape a suite of near-optimal solutions for achieving a set objective. It is with the best use of available information that we can achieve practical solutions for aiding decisions. However, conservationists should be mindful that being a "crisis" discipline means one must act quickly and that tolerating some degree of uncertainty is often necessary (Soulé 1985; Regan et al. 2005; McDonald-Madden et al. 2008). To address the urgent protection of biodiversity, we must continue to develop prioritisation strategies that use scarce conservation resources effectively and efficiently. The success of such strategies ultimately hinges on our ability to apply them to the real world.

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Appendix 1.

Levin, N., Tulloch, A., Gordon, A., **Mazor, T.**, Bunnefeld, N. & Kark, S. (2013). Incorporating Socioeconomic and Political Drivers of International Collaboration into Marine Conservation Planning. *BioScience*, **63**, 547-563.

Incorporating Socioeconomic and Political Drivers of International Collaboration into Marine Conservation Planning

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International collaboration can be crucial in determining the outcomes of conservation actions. Here, we propose a framework for incorporating demographic, socioeconomic, and political data into conservation prioritization in complex regions shared by multiple countries. As a case study, we quantitatively apply this approach to one of the world's most complex and threatened biodiversity hotspots: the Mediterranean Basin. Our analysis of 22 countries surrounding the Mediterranean Sea showed that the strongest economic, trade, tourism, and political ties are clearly among the three northwestern countries of Italy, France, and Spain. Although economic activity between countries is often seen as a threat, it may also serve as an indicator of the potential of collaboration in conservation. Using data for threatened marine vertebrate species, we show how areas prioritized for conservation shift spatially when economic factors are used as a surrogate to favor areas where collaborative potential in conservation is more likely.

Keywords: marine conservation planning, international collaboration, Marxan, Mediterranean Sea, spatial prioritization

International collaboration has been shown to be a key to success in tackling a range of environmental issues (e.g., the Montreal Protocol on Substances that Deplete the Ozone Layer; Velders et al. 2007). Developing transboundary marine parks is one useful strategy used to facilitate a collaborative approach in conservation planning (Mackelworth 2012). This approach is often applied at the subregional scale and poses substantial challenges, because it depends on the availability of appropriate funding, resources, and political will, among other factors (Mackelworth 2012). A range of factors may be associated with a country's willingness or ability to take collaborative conservation actions (Sarkar et al. 2006, McDonald and Boucher 2011). These include socioeconomic factors (e.g., gross domestic product [GDP]) and political factors, such as governance—the competency, incorruption, and accountability of public administrations (Leftwich 1993). It is recognized that international protocols and legislative agreements for biodiversity conservation can legitimize sociopolitical interests (e.g., Groves et al. 2002, Sarkar et al. 2006).

In recent decades, wide application of systematic conservation planning has been in place, with the aim of designing and implementing protected area networks on the basis of specified conservation goals (Moilanen et al. 2009, Hooper et al. 2012). However, conservation goals that are focused on preserving a target proportion of endemic or threatened biodiversity in a given area are often ambitious and costly, and the funding available for conservation is usually less than what is required (Balmford et al. 2003, 2005). Limited funds therefore need to be spent in a cost-effective and efficient manner (Moilanen et al. 2009). It is increasingly acknowledged that collaborative conservation actions can lead to improved efficiency and economic savings (e.g., Rodrigues and Gaston 2002, Wells et al. 2010). For example, Kark and colleagues (2009) found that collaboration between countries can improve conservation efficiency and can potentially allow countries to save conservation funds and to achieve more conservation targets for the same area size. In marine environments, between-country collaboration and coordination is of special importance because of factors such as currents and the natural flow of material in the oceans (e.g., nutrients, pollution), the high mobility of many marine species (both native and alien), the common use of marine resources (Hardin 1968), and the varying levels of

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marine sovereignty (e.g., territorial waters, exclusive economic zones [EEZs]; Suárez de Vivero et al. 2009).

Traditionally, systematic conservation planning has been focused on achieving biodiversity targets, such as species richness and complementarity (Margules and Pressey 2000). However, various studies have illustrated that incorporating economic costs into conservation planning can achieve substantial conservation gains in terms of the biodiversity protected (e.g., Stewart and Possingham 2005, Naidoo et al. 2006). Nonetheless, only a few conservation-planning studies have incorporated the potential for intercountry collaboration in conservation prioritization (but see Kark et al. 2009, Moilanen et al. 2012, Mazor et al. 2013). Collaboration has many benefits to conservation, including the sharing of expertise and technical capacity, as well as knowledge (e.g., Lacher et al. 2012). In addition, collaboration can reduce the overall costs of conservation actions (Kark et al. 2009) and has the potential to reduce conservation costs by lowering transaction costs (e.g., those related to negotiations), which can be substantial (Michaelowa et al. 2003). Clearly, successful implementation of conservation plans requires the incorporation of socioeconomic, political, and demographic considerations into conservation planning (McDonald and Boucher 2011). This is especially important in regions in which resources are shared by multiple countries and particularly at the international scale. A range of activities (e.g., trade and resource extraction) can have direct impacts on biodiversity beyond a single country's boundaries. Trade between countries is often considered a vector for threats to biodiversity, especially in relation to threatened species, because of, for example, habitat loss or the hunting or fishing of threatened species, such as in the shark fin trade (Clarke 2004) and the ivory trade (Lenzen et al. 2012). However, trade may also facilitate successful collaboration in conservation efforts. Countries that develop strong commercial ties among one another may have greater potential to collaborate on additional factors, including environmental issues and conservation efforts in particular (Bunnefeld et al. 2011, Fulton et al. 2011; for examples of such collaboration, see Sandwith and colleagues [2001]).

International environmental regulations and agreements are important components of international collaboration in conservation (Donald et al. 2007, Rands et al. 2010). Numerous international and regional conservation-related agreements have been signed, such as the Convention of Biological Diversity and the Convention on International Trade in Endangered Species, in an attempt to stem the tide of species extinctions and loss of ecosystems (see supplemental appendix S1, available online at *http://dx.doi.org/10.1525/bio.2013.63.7.8*). International environmental agreements are important because they set international standards; draw global attention to environmental issues; lead to national legislation on conservation matters; and direct governmental funding, legal action, and awareness into environmental issues, and they may therefore lead to better governance (Bennett and Ligthart 2001, Biermann et al. 2012). Although collaboration may have substantial benefits in advancing conservation efforts, there are numerous barriers to effective collaboration between countries in conservation efforts (Kark et al. 2009, McDonald 2009). Such barriers include, for example, political, linguistic, and cultural differences. A history of political instability or military conflict has also been shown to lead to a reduced ability to participate in collaborative conservation programs and therefore hampers the political feasibility of between-country collaboration (Didia 1997, Neumayer 2002). In addition, political will, which, in itself, is a function of societal values, is required in order to provide funding for conservation (Brechin et al. 2002). New conflicts also arise in times of increasing usage and exploitation of natural resources, including newly discovered deep-sea hydrocarbons (e.g., natural gas; see Borgerson 2008), further emphasizing the urgent need for advancing collaborative conservation in marine areas.

In the present study, we quantify the strength of collaborative potential between countries with respect to various socioeconomic and political factors and explore methods and approaches for incorporating international collaboration between countries into systematic conservation planning in marine systems, including marine protected areas (MPAs). We focus on the Mediterranean Sea as a case study. The Mediterranean Sea is a unique ecosystem, being a largely enclosed internal sea surrounded by more than 20 countries spanning three continents (Europe, Asia, and Africa), all of them sharing its natural resources. The Mediterranean Sea's rich and endemic biodiversity faces increasing threats (Bianchi and Morri 2000, Coll et al. 2012). This has led to recent calls for the creation of an effective network of MPAs in the area (de Juan et al. 2012, Giakoumi et al. 2012, Micheli et al. 2013) and for large-scale conservation planning in the sea beyond the territorial waters.

The Mediterranean Sea is unique in the fact that once all countries declare their respective EEZs, there will be no international waters within it. Currently, coastal MPAs in the Mediterranean Sea cover less than 0.5% of the coastal area (Abdulla et al. 2008). Although the European Union can influence the establishment of new MPAs (e.g., through the Natura 2000 [EU 1992] initiative), so far, the network of MPAs in the Mediterranean Sea is lacking (Giakoumi et al. 2011, 2012). According to Abdulla and colleagues (2008), there are 93 MPAs (with a median area of 26 square kilometers [km^2]) in the Mediterranean Sea, all but one within coastal territorial waters (also, in part, because most countries have yet to formally declare their EEZs). The only international MPA in the Mediterranean is the Pelagos Sanctuary (shared among Italy, France, and Monaco), which was declared in 1999 and has an area of 87,500 km^2 (Notarbartolo di Sciara et al. 2008). Italy has the largest number of MPAs (25) and the largest total area (2738 km^2), compared with all other Mediterranean countries (appendix S4). MPAs larger than 500 km^2 ($n = 6$) are found only in the waters of Italy (2 large MPAs), Greece (1), Turkey (1), Croatia (1), and France (1). Aside from two MPAs in Spain that are between 100 and

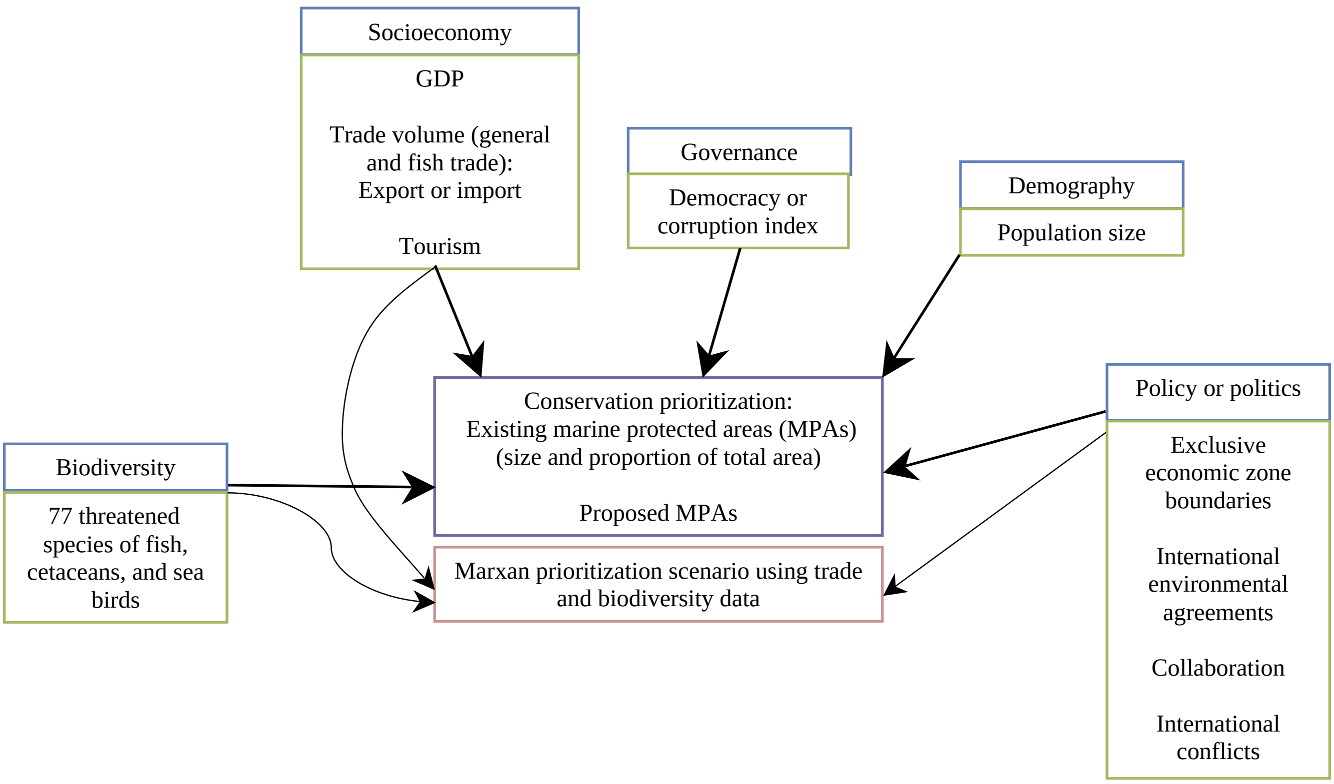


Figure 1. Schematic flowchart showing the framework and variables used in the present study. The variables used in the case study on marine protected areas (MPAs) in the Mediterranean Sea (see the "Mapping and quantifying collaboration" section) are connected with thin black lines. Abbreviation: GDP, gross domestic product.

500 km^2, all 16 other Mediterranean countries currently have only MPAs smaller than 100 km^2.

Here, we use biodiversity, demographic, socioeconomic, policy, and political characteristics of the countries bordering the Mediterranean Sea to examine the correspondence among the multiple factors with the extent of current conservation efforts, reflected by the total area and number of MPAs per country (figure 1). We present an approach for estimating the potential for collaboration between countries when taking conservation actions. Our working hypothesis is that neighboring countries with stronger commercial, social, and political ties will also be in a better position to collaborate in marine conservation efforts. Our main research questions in the present study are the following: What is the potential of economic and political factors to predict conservation efforts at the country level? How do existing economic and political collaborations between countries correspond with their collaboration in conservation? How can information about collaboration be applied in spatial conservation prioritization? Last, how does the incorporation of socioeconomic and political information affect spatial conservation-planning outcomes?

Mapping and quantifying collaboration

We collated a database of the biological, socioeconomic, and political characteristics of the countries bordering the Mediterranean Sea (shown schematically in figure 1). After analyzing the correlations between the countries' characteristics and their conservation efforts, we constructed matrices quantifying the strength of economic collaboration between all pairs of countries. Finally, we demonstrated how information about collaboration between countries can be used for spatial prioritization of conservation efforts using the Marxan conservation-planning software package.

Altogether, 23 countries (including Gibraltar and the Palestinian Authority; table 1) are located along the coast of the Mediterranean Sea. We created a binary matrix of the shared marine borders for all 23 countries that have a stretch of coast along the Mediterranean Basin (following Suárez de Vivero and Mateus 2002). We defined two countries as sharing an international boundary on the basis of their marine EEZ boundaries. Although most Mediterranean countries have not yet formally claimed or agreed on the spatial delimitation of their exact EEZ boundaries (Suárez de Vivero and Mateus 2002), for this analysis, we adapted the EEZ boundaries in the MARBOUND Marine Regions database (*www.vliz.be/vmdcdata/marbound*). We excluded Monaco from most analyses because of a lack of trade data (see below for details), which left us with 22 Mediterranean countries for the analysis. The data collected for each country included the following factors: biodiversity (the spatial distribution of *threatened* species), demography (human

Table 1. General geographic, demographic, and trade statistics for each of the Mediterranean countries.

Country	2008 population (in millions of people)	2008 GDP per capita (in dollars)	Shared borders	Total export: Absolute value (in billions of US dollars)	Total export: Providers' percentage of total trade	Total import: Absolute value (in billions of US dollars)	Total import: Users' percentage of total trade	Dependency: Percentage of goods exported	Dependency: Percentage of goods imported	Median number of shared species with other countries
Albania	3.6	4149	3	1.0	0.0	2.5	0.0	76.6[b]	53.3[b]	56[d]
Algeria	33.7	3520	5	24.2[f]	0.3	18.9[e]	0.6[f]	42.2	41.0	58[b]
Bosnia and Herzegovina	4.6	7274	3	2.3	0.0	4.9	0.0	46.1[f]	41.7	52
Croatia	4.5	8904	3[e]	5.9	0.1	9.6	0.1	51.5	33.8	57[c]
Cyprus	0.8	29,238[b]	6[b]	1.5	0.0	6.5	0.1	32.7	46.9[d]	48
Egypt	81.7[a]	3725	5[c]	11.5	0.5[f]	10.8	0.4	34.5	15.6	51
France	64.0[c]	22,223[c]	2[f]	142.3[b]	4.0[b]	145.9[a]	5.0[b]	24.5	21.1	56[d]
Gibraltar	0.03	56,425[a]	2[f]	0.1	0.0	2.8	0.0	31.1	32.0	56[d]
Greece	10.7	16,362	6[b]	10.3	0.6[e]	27.5	0.9[e]	36.5	28.8	58[b]
Israel	7.1	17,937	5[c]	10.1	0.3	8.4	0.3	11.9	11.9	49
Italy	58.1[d]	19,909[e]	12[a]	149.8[a]	8.1[a]	137.9[b]	8.3[a]	29.5	22.0	60[a]
Lebanon	3.9	4453	3[e]	0.9	0.0	5.5	0.1	26.1	28.7	48
Libya	6.1	2994	6[b]	35.6[d]	0.3	8.2	0.3	56.7[e]	45.3[e]	54[e]
Malta	0.4	20,497[d]	3[e]	3.7	0.1	4.4	0.1	22.5	43.5[f]	51
Montenegro	9.9	3620	3[e]	0.4	0.0	1.0	0.0	60.6[d]	33.2	56[d]
Morocco	34.3[f]	3465	4[d]	10.2	0.1	16.7[f]	0.5	47.4	40.0	57[c]
Palestinian Authority	4.1	2178	2[f]	0.0	0.0	0.1	0.0	90.5[a]	83.6[a]	No data
Slovenia	2.0	18,170	2[f]	9.1	0.1	11.3	0.2	35.1	33.5	53[f]
Spain	40.5[e]	19,706[f]	6[b]	92.3[c]	2.3[c]	111.8[c]	2.9[c]	35.0	26.0	58[b]
Syrian Arab Republic	19.7	8360	4[d]	4.1	0.1	4.4	0.1	43.2	24.8	45
Tunisia	10.4	6103	4[d]	12.1	0.1	13.4	0.3	64.6[c]	52.5[c]	56[d]
Turkey	75.8[b]	8066	3[e]	34.5[e]	2.2[d]	32.1[d]	1.5[d]	26.1	16.8	51

Note: The values are ranked in each column, from first to sixth (superscript a–f). The total number of *threatened* species analyzed was 77 (see supplemental appendix S2, available online at *http://dx.doi.org/10.1525/bio.2013.63.7.8*). For the export and important statistics, the *Providers* column represents the median share of imports in Mediterranean Basin countries from each exporting country (i.e., for this column, each country mentioned in the first column is an exporter, or *provider*). The *Users* column represents the median share of imports from Mediterranean Basin countries from each importing country (i.e., for this column, each country mentioned in the first column is an importer, or *user*).

population size), governance (democracy and corruption indices), economy (GDP, trade), tourism, politics (history of conflicts, international agreements), and the spatial extent of protected areas.

We used data on the occupancy of the native Mediterranean *threatened* and *near-threatened* marine vertebrate species compiled from the International Union for Conservation of Nature's (IUCN) Red List of Endangered Species (*www.iucnredlist.org/technical-documents/spatial-data*; appendix S2). These data comprised 63 fish species, 7 cetacean species, 5 seabird species, and 2 sea turtle species—altogether, 77 species (appendix S2). We overlaid the distribution ranges of each species and mapped its occupancy area within each country's Mediterranean EEZ. On the basis of these data, we derived a matrix of the number of shared species (ranging from 38 to 68) among the Mediterranean Basin countries. Our assumption here was that countries that share species may have a stronger incentive for collaboration in conservation.

To examine the existing set of MPAs in the study area, we combined information from the IUCN report by Abdulla and colleagues (2008) with country statistics of the percentage of each country's land covered by terrestrial protected areas or sea area covered by MPAs (WDPA 2010). We used terrestrial protected areas in our analysis because they may reflect a conservation-oriented tradition or conservation-related policy in that country.

To demonstrate how existing conservation plans and biodiversity-monitoring efforts are distributed within the Mediterranean Sea, we explored the following spatial layers: We digitized the map of existing and proposed MPAs for whales and dolphins in the Mediterranean and Black Seas from ACCOBAMS (the Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea, and Contiguous Atlantic Area; *www.cetaceanhabitat.org*; Rais et al. 2006). We then evaluated the spatial distribution and extent of proposed MPAs over the EEZs of the Mediterranean Sea countries. We also mapped the location of underwater surveys conducted by Sala and colleagues (2012).

We used demographic data (human population size, from *www.ggdc.net/maddison/content.shtml*) for all of the Mediterranean countries (following Maddison 2007) for calculating the per capita values of trade and tourism factors. We used the Corruption Perceptions Index at the country level, derived from the World Resources Institute (*http://www.transparency.org/research/cpi/overview*), and the Democracy Index 2011, from the Economist Intelligence Unit (*www.eiu.com/public/topical_report.aspx?campaignid=DemocracyIndex2011*), to test whether these measures are correlated with a country's conservation efforts.

We collated data on the signatories of 27 major international agreements and policies related to conservation issues (appendix S1) and created a matrix showing the number of shared international conservation agreements between Mediterranean countries. To complement this and to represent any negative relationships between countries, we also collected information about military conflicts between the countries in the past 50 years (from 1963 onward; Themnér and Wallensteen 2011; Uppsala Conflict Data Program, *www.pcr.uu.se/research/UCDP*). This included information on the total number and duration of military conflicts among the Mediterranean countries (including conflicts between nongovernmental militia forces from one country acting against another country).

We collated the GDP statistics of all of the Mediterranean countries (from *www.ggdc.net/maddison/content.shtml*). We used the trade volume between countries to examine their economic interdependencies. We used trade statistics from Trade Map (*www.trademap.org*), which were based on statistics from the United Nations Commodity Trade Statistics Database (*http://comtrade.un.org*). We used trade data from 2008, because this was the most recent year for which trade data were available for all of the Mediterranean countries (except Monaco). Trade matrices between countries were constructed for all commodity types and for trade only in marine products (including fish, crustaceans, mollusks, aquatic invertebrates; also from Trade Map).

On the basis of these matrices, we then calculated the relative share of each country's import from and export to each other Mediterranean country, both in absolute numbers and relative to the country's total import and export. Using these data, we aimed to determine which of the Mediterranean countries are major providers of exported goods or users of imported goods. We used the import and export trade matrices to determine which countries were more dependent on other Mediterranean countries for their trade ties and to what degree they were trading with other Mediterranean countries.

We collected data on tourism from the UN World Tourism Organization (UNWTO 2013) for each Mediterranean country in the year 2010, showing the number of tourists arriving (inbound) from and departing (outbound) to each other country. We calculated both the proportion of tourists per capita and the percentage of incoming tourists from other Mediterranean countries out of the total number of incoming tourists.

Analyzing the collaboration data

To help the reader visualize the connections, we present the matrices spatially as networks and, therefore, visualize the spatial patterns of collaboration between Mediterranean countries as networks. For example, we show the trade and tourism connections depicted as lines connecting the capital cities of each country (using an equal-area Lambert projection; see Lenzen and colleagues [2012] for a similar approach). To standardize the different factors for comparison, we ranked the values in each of the matrices (of, e.g., trade, tourism, shared species, shared agreements) by their order from highest to lowest (e.g., the country that imported the most from another country was ranked first for the trade import variable). In order to summarize all the trade and tourism statistics for each country into a single composite number, we calculated the median rank of all 231 possible trade and tourism connections for each country. This resulted in a single trade score and a single tourism score for each country, representing its trade and tourism connections with each of the other countries.

We calculated monotonic relationships between the different factors at the country level, such that the demographic, economic, and political variables served as the independent variables, and the area protected for conservation (in square kilometers as well as in the percentage of a country's area) served as the dependent variable. The above relationships were calculated using Spearman's rank correlation. We also performed a hierarchical cluster analysis at the country level, using several clustering methods (average, ward, and centroid) for measuring the distance between the countries using JMP7 statistical discovery software (SAS Institute; *www.jmp.com*). This was done for the following variables: population size, GDP, protected areas (area, number, and proportion), trade, tourism, shared legislation, shared species, and democracy and corruption indices.

Spatial prioritization of protected areas. In order to demonstrate the effects of collaboration between countries on the spatial prioritization of protected areas, we used the conservation-planning software Marxan (University of Queensland; *www.uq.edu.au/marxan*). Marxan is a decision support tool for conservation planning (Moilanen et al. 2009) and finds efficient solutions to the problem of selecting a least-cost system of spatially cohesive areas that meet a suite of biodiversity targets (Possingham et al. 2000). The proportion of times a spatial planning unit is included in the selected set of protected areas (selection frequency) can be used to determine its priority (irreplaceability) for conservation and to compare different scenarios (Leslie et al. 2003). We also used Marxan (Possingham et al. 2000) to demonstrate how information about collaboration between countries can be integrated into a systematic conservation-planning tool. We used square planning units of 100 km^2, corresponding with the spatial scale and accuracy of the species distribution data and following a study in the terrestrial Mediterranean Basin (Kark et al. 2009).

Comparing collaboration scenarios. In the conservation-planning analysis, we set biodiversity targets to be 30% of the distribution area for each of the 77 threatened marine species. We then compared how these targets could be achieved using (a) a scenario with no collaboration and (b) a scenario in which collaboration between neighboring countries was incorporated. In the no-collaboration scenario, the costs of all planning units were uniform. In the full-collaboration scenario, we used the median trade rank between neighboring countries as a surrogate for cost, assuming that collaboration in trade facilitates collaboration in conservation. Therefore, a high trade ranking between a pair of countries signifies lower costs for collaboration. This resulted in a higher prioritization of planning units between country pairs that have strong trade ties than of planning units between country pairs with weaker ties. We assumed in this scenario that collaboration in conservation can occur across a shared EEZ boundary. Although additional variables may also be included, we chose to use trade both because we hypothesized that it can serve as a surrogate for the political feasibility of collaboration between countries and as a demonstration of our methodological approach (see the section entitled "The implications of between-country collaboration for conservation in the Mediterranean" below).

We used the EEZ boundaries to create a layer of Thiessen polygons (Thiessen 1911), using the ALLOCATE algorithm within Idrisi Selva geographic information system software (version 17.0; Clark Labs; *http://clarklabs.org*). Thiessen polygons define individual areas of influence around a given set of points (in our case, these sets of points are defined by the EEZ boundaries). The Thiessen polygon boundaries then define the area that is nearest to each point relative to all other points. Mathematically, they are defined by the perpendicular bisectors of the lines between each point and every other point (see supplemental figure S1). Using the Thiessen polygon layer, we allocated each 100 km^2 planning unit to its nearest EEZ boundary. We then assigned the median ranking of the trade connections of a country pair as the *cost* to all the planning units allocated to the EEZ boundary of the country pair defined by the Thiessen polygons. We ran Marxan 1000 times for each collaboration scenario, with a boundary length modifier value of 2 in both scenarios (determined using a sensitivity analysis following Ardron and colleagues [2010]). We compared the selection frequency of the planning units in the two scenarios and calculated the change in the selection frequency of the planning units when trade connections were considered.

Spatial trends in socioeconomic and political factors

We discovered a clear distinction in most of the factors tested here between the EU Mediterranean countries of Italy, France, and Spain and all other Mediterranean countries. These three countries were also three of the six most-populated Mediterranean countries (France, Italy, and Spain had a combined population of 162 million people in 2008). The three other most-populated countries were all non-EU countries: Egypt, Turkey, and Morocco, which had a combined population of 192 million people in 2008 (table 1). The six highest-ranking countries in terms of GDP (with a per capita GDP above $18,000) all belonged to the European Union and also included France, Italy, and Spain (table 1). Of the 22 Mediterranean countries examined, Italy had by far the highest number of shared EEZ borders with other Mediterranean countries (sharing boundaries with 12 other countries because of its central location; figure 2), followed by Spain, Cyprus, and Libya (which had five shared EEZ borders each; table 1).

The countries that had signed the largest number of international conservation agreements included Italy (23 agreements), France (21), Spain (20), and Morocco (20),

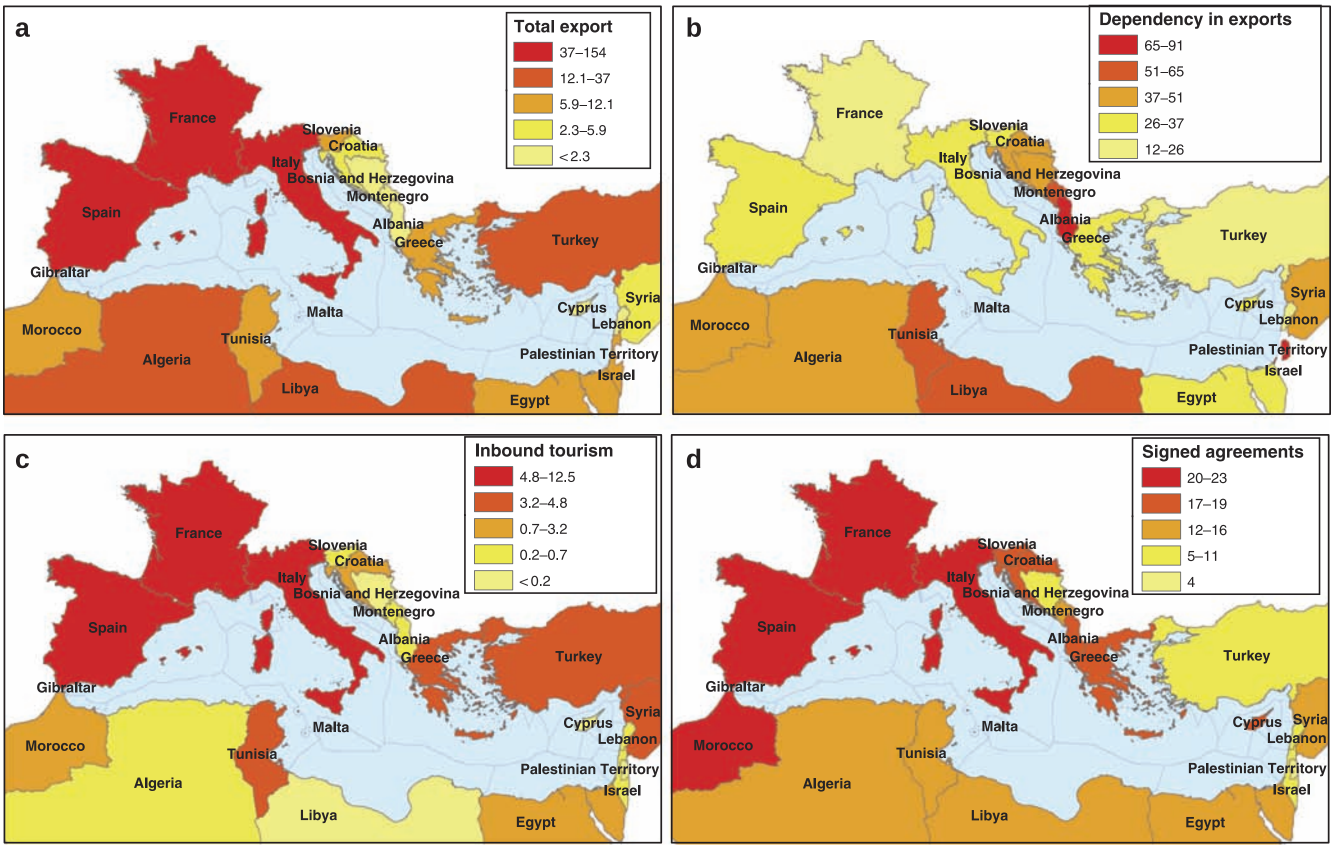


Figure 2. The distribution of the major socioeconomic and political factors at the country level. (a) Total export to other Mediterranean countries (in billions of US dollars). (b) Dependency in exports calculated as the percentage of total exports sent to other Mediterranean countries. (c) Inbound tourism from other Mediterranean countries (in millions of people). (d) The number of signed international agreements related to conservation and environmental issues.

and those with the fewest signed agreements were Bosnia and Herzogovina (9) and the Palestinian Authority (4) (figure 2d). Overall, on the basis of the 2011 Democracy Index, northern Mediterranean Sea countries were more democratic than those in the eastern and southern Mediterranean (figure 3).

In terms of the volume of trade with other Mediterranean countries, Italy, France, and Spain were again the top three Mediterranean countries in both their total import and total export volumes (table 1, figure 2a). When we calculated the proportion of trade between each country and the other Mediterranean Basin countries by their total trade volume (with all other countries), Italy was the leading exporter to other Mediterranean countries and provided the greatest proportion (8.1%, median value, of their total imports worldwide) of exports to other Mediterranean Basin countries and imported 8.3% (median value) of its total imports from other Mediterranean Basin countries (table 1, appendix S3). The major importer and exporter countries after Italy were France, Spain, and Turkey (table 1). When we examined the trade of marine products alone (e.g., fish), we found that Italy provided the greatest proportion of marine exports to other Mediterranean Basin countries (13.1%, median value), followed by Spain (4.9%, median value). Israel (0.3%, median value) and the Palestinian Authority (less than 0.1%, median value) had the weakest trade ties with other Mediterranean countries (table 1, appendix S3). The average share of a country in import (or export) with other Mediterranean countries was positively correlated both with the total value of its own import (or export; $r = .87$, $p < .001$) and with the number of its shared boundaries ($r = .64$, $p < .01$).

France, Spain, and Italy had the highest number of inbound tourists from other Mediterranean Basin countries (12.4 million, 11.8 million, and 9.3 million, respectively) and outbound tourists (19.8 million, 9.1 million, and 15.7 million, respectively, in 2010) to other Mediterranean Basin countries (figure 2c).

The factors most strongly and significantly correlated with the percentage of terrestrial area set aside as protected area included the democracy index ($r = .73$, $p < .001$), the per capita GDP ($r = .54$, $p < .01$), and the number of inbound

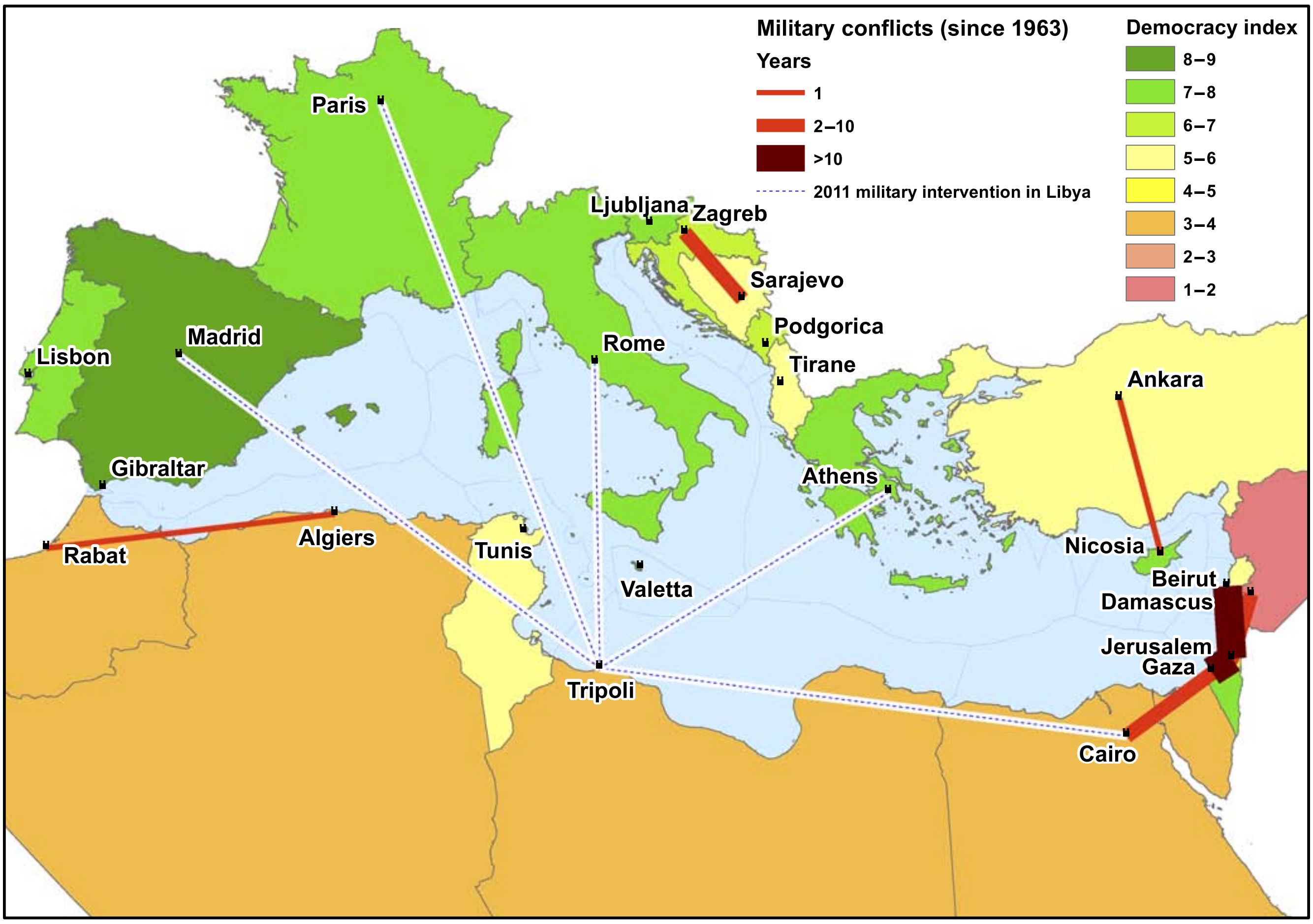


Figure 3. The number in years of military conflicts between the different Mediterranean countries since 1963, based on Themnér and Wallensteen (2011) and the democracy index of the Mediterranean countries (based on the Economist Intelligence Unit's Democracy Index 2011; see the "Mapping and quantifying collaboration" section). The dashed lines show the North Atlantic Treaty Organization intervention in Libya during 2011.

tourists per capita originating from other Mediterranean countries ($r = .52$, $p < .05$; table 2). The variables that were most strongly correlated with the percentage of marine area set aside as MPAs (within the territorial waters) and with the total area of MPAs within a country's EEZ included the total number of inbound tourists (rs = .59 and .73, respectively), the total exports of marine products to other Mediterranean countries (rs = .45 and .78, respectively), and the total imports from other Mediterranean countries (rs = .53 and .75, respectively; table 2).

A significant positive correlation was found between the size of the MPAs per country and the number of international conservation agreements to which a country was a signatory ($r = .48$, $p < .05$; table 2). The distinction between Mediterranean countries in their economic, political, and demographic variables was confirmed by a cluster analysis performed at the country level (supplemental figure S2). In all three dendrograms, Italy, Spain, and France were always separate from the other Mediterranean countries, regardless of the clustering method used (figure S2).

Socioeconomic and political connections between countries

In all of the factors that we analyzed, the strongest socioeconomic and political ties between countries were found in the northwestern part of the Mediterranean Basin, with the triangle of the strongest ties among Italy, France, and Spain appearing in the networks of trade and tourism, as well as in their shared species and shared international agreements (figure 4). More specifically, the connection between Italy and France was always ranked either first or second out of all of the 231 possible connections between Mediterranean countries for the following four variables: total import, total export, inbound and outbound tourism, and the number of shared agreements. In general, the connections were stronger among European Mediterranean countries than among non-European Mediterranean countries. The least connected countries were located in the southeastern region of the Mediterranean Basin. The eastern Mediterranean region had the highest number of military conflicts between Mediterranean countries in the past 50 years (figure 3).

Table 2. Socioeconomic and political variables correlated with the percentage of a country's total area that is set aside as terrestrial and marine protected areas (Spearman's rank correlation coefficient), the absolute area of marine protected areas in that country, and the number of conservation agreements signed by that country.

	Percentage of a country as terrestrial protected areas	Percentage of a country as marine protected areas	Absolute area of marine protected areas	Number of signed international conservation agreements
Gross domestic product per capita	.54	.34	.33	.42
Democracy index	**.73**	.48	.24	.48
Median percentage of imports from Mediterranean Sea countries	.19	.53	**.75**	.53
Total exports of marine products to Mediterranean Sea countries	.04	.45	**.78**	.64
Number of signed international environmental agreements	.41	.33	.48	–
Number of inbound tourists per capita	.52	.42	.45	.64
Total inbound tourism	.03	.59	**.73**	.60

Note: Correlations greater than .7 are shown in bold.

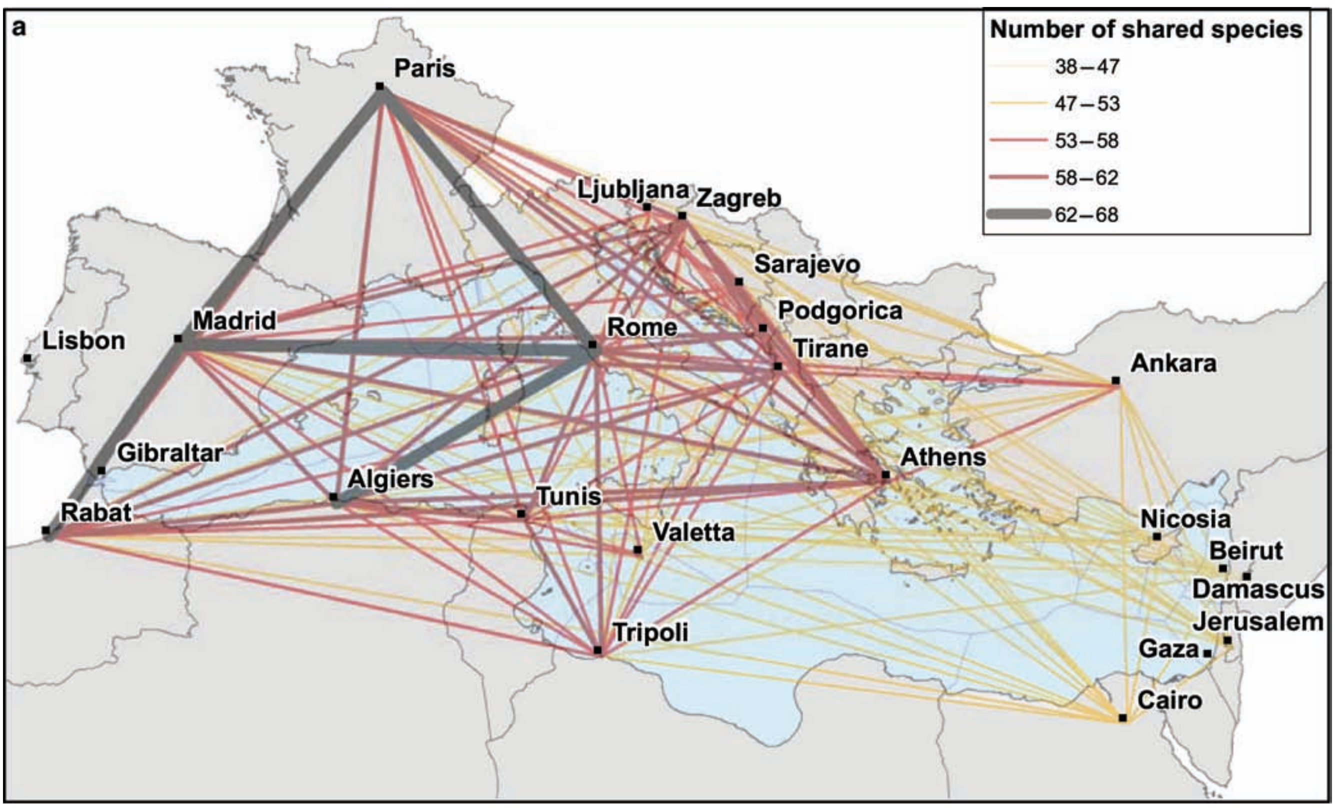


Figure 4. The spatial patterns of socioeconomic and political interactions between each pair of Mediterranean countries presented as a network. Between-country connections are depicted as lines linking the countries, and the line width and color represents the strength of the relationship. (a) The number of shared species between each pair of Mediterranean countries, calculated from the 77 threatened vertebrate species included in the study. (b) The median rank of all the variables used to calculate trade connections between Mediterranean countries. (c) The median rank of all the variables used to calculate tourism (inbound and outbound) connections between Mediterranean countries. (d) The number of shared international environmental agreements between country pairs. Low values represent a high ranking in (b) and (c).

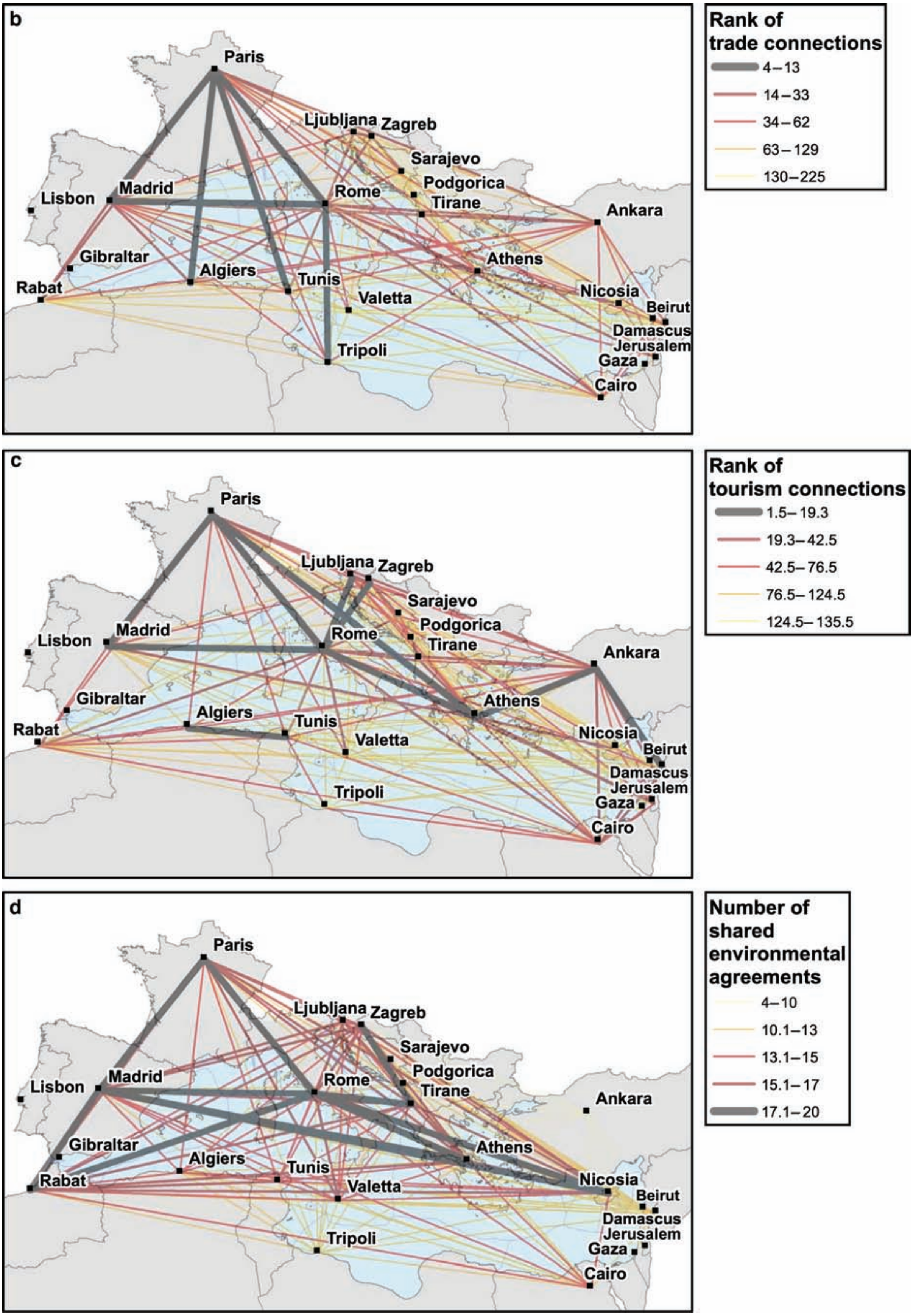


Figure 4. (Continued)

Interestingly, the geographical distance between pairs of Mediterranean countries was not significant in explaining their between-country trade or tourism connections.

Conservation prioritization outcomes when collaboration is considered

When uniform costs were used in the Marxan scenario of no collaboration among Mediterranean countries, the spatial pattern of the resulting selection frequency was driven by biodiversity patterns of the threatened species and species spatial aggregation, showing higher selection frequency (the number of times each grid cell is selected in the 1000 Marxan runs) and therefore higher conservation priority near the coast (figure 5b). However, there was no clear difference in selection frequency of the northern, southern, eastern, and western parts of the Mediterranean Sea.

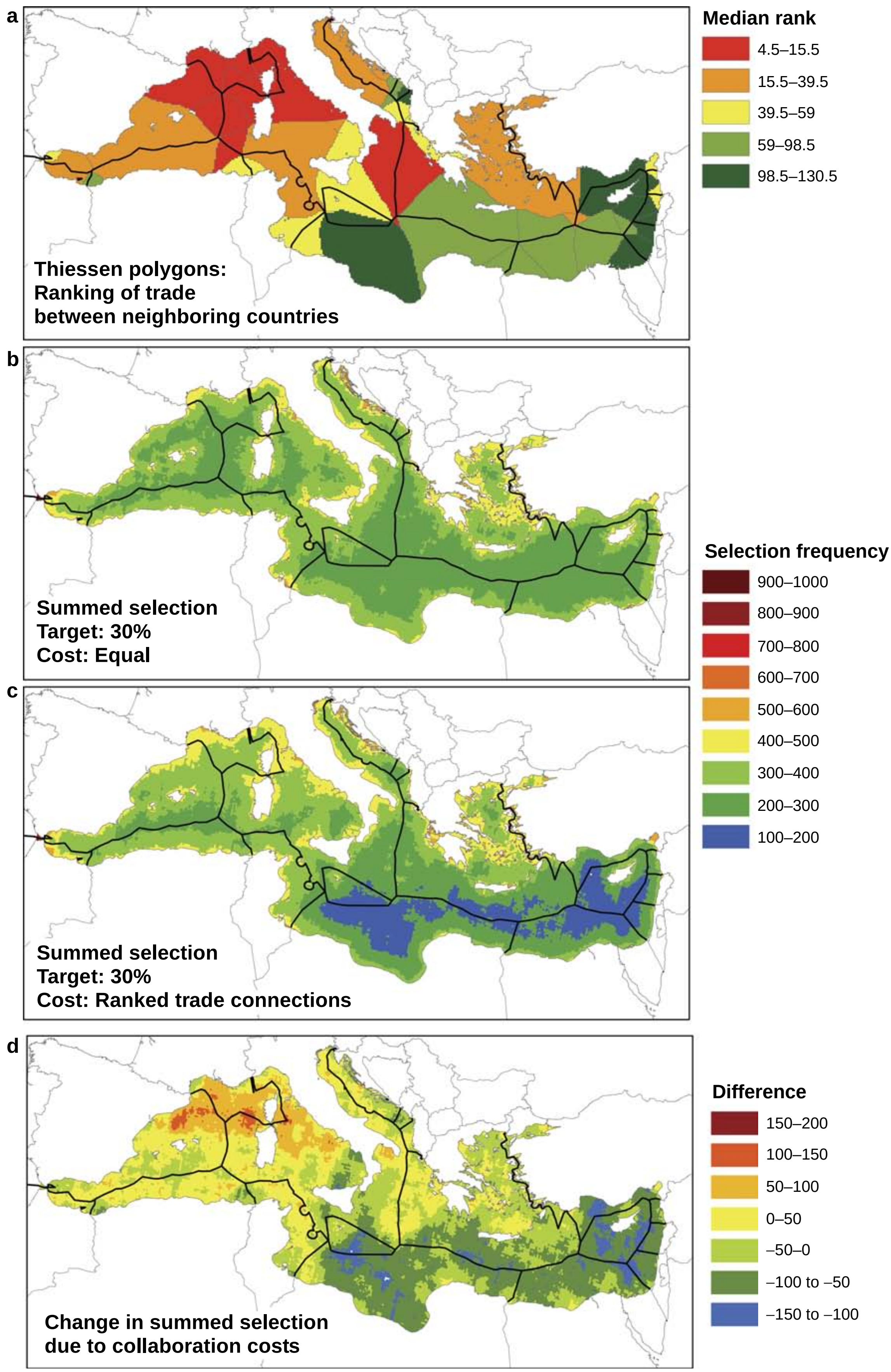
In the second scenario, we incorporated between-country collaboration in trade (outbound and inbound combined), spatially allocating the trade ranking using Thiessen polygons. The Thiessen polygons of neighboring countries that had strongest trade connections (e.g., Italy and France, France and Spain, Italy and Greece) are shown in figure 5a. When we used in our Marxan runs the median ranking of trade connections as a surrogate for higher feasibility of collaborative conservation efforts (low conservation costs), the selection frequency of planning units changed such that planning units in the southern and eastern Mediterranean Sea were selected less frequently (figure 5c) and areas in the northwestern area were selected more frequently (figure 5d). This shift in conservation prioritization corresponds with the difference in trade connections among the countries of the southeastern Mediterranean Basin and among the countries of the northwestern Mediterranean Basin (figure 5a).

The implications of between-country collaboration for conservation in the Mediterranean

In recent years, with the increasing availability of spatial quantitative and mapping tools, conservation planning has advanced rapidly, allowing more efficient spatial prioritization at large regional scales (Moilanen et al. 2009). Awareness of the importance of incorporating anthropogenic factors into conservation planning—in addition to biological factors—is also increasing (Kark et al. 2009, Klein et al. 2010, Bryan et al. 2011). Economic activity, such as trade (Lenzen et al. 2012) and large-scale tourism (Gray 1997), is often viewed as a threat to biodiversity. However, such factors can also serve as useful surrogates for determining where successful collaboration in conservation interventions is more likely. In the present study, we showed how such socioeconomic and political factors could potentially serve as helpful predictors of conservation efforts at the country scale and provided an example of how they can be incorporated into the conservation-planning process.

On the basis of our socioeconomic–political analysis, we found that in Mediterranean countries with higher GDPs, a larger volume of outgoing and ingoing trade (with other Mediterranean countries), a greater number of incoming Mediterranean tourists, and more-democratic political systems tended to allocate more terrestrial and marine area for conservation (i.e., in protected areas; see table 2) and were signatories to a larger number of international conservation agreements. We also found that collaborative potential (evident in a wide range of socioeconomic and political factors) was strongest among European countries situated along the northwestern coast of the Mediterranean Sea (figures 4 and 5). Interestingly, the northwestern countries also shared the largest number of threatened species, suggesting that these countries may have strong potential for defining common conservation targets and for collaborating in reaching them. There are several issues with the IUCN biodiversity information available for the present study, with most significant biases in the data being potentially due to unequal sampling efforts across different taxonomic groups, locations, or times and the use of species ranges rather than probabilities of occurrence. We used these data in the current study because of their availability at the full Mediterranean Sea scale, but as better information about species distributions becomes available, this analysis can be repeated with improved biodiversity data. Our findings correspond with those of Kark and colleagues (2009), who pointed to the European Union as a region in which conservation collaboration may be practical and feasible. Because the European Union already has in place a range of environmental agreements, efforts, and collaborations (e.g., EU 1992; see appendix S1), conservation efforts among EU countries may be an effective first step toward integrating socioeconomic and political factors into collaborative conservation efforts across the Mediterranean Basin (Kark et al. 2009). However, more area may be required to reach the same conservation targets if conservation is focused only on EU countries (Kark et al. 2009). Therefore, the next steps could involve countries and regions among which there are weaker economic and political ties, and collaborative conservation may be more challenging to initiate but may lead over time to effective impacts. Interestingly, it has been shown that collaborative environmental efforts may also, in some cases, lead to improved sociopolitical ties (e.g., through peace parks; see Sandwith et al. 2001).

Clearly, the size and geographic location of particular countries may influence their likelihood of implementing successful collaborative activities. For example, Italy, Greece, Libya, and Spain have the largest (potential) EEZ areas in the Mediterranean (65% of the total Mediterranean Sea marine area; supplemental appendix S4) and, therefore, will probably have important roles in the conservation of the sea's biodiversity and its threatened species. The countries most strongly connected to other Mediterranean countries (determined on the basis of their trade, tourism, and other variables) were also the three Mediterranean EU countries with the largest populations: Italy, France, and Spain. Italy's central location within the Mediterranean Sea appears to play a major role in determining its strong economic ties


a
Median rank
4.5–15.5
15.5–39.5
39.5–59
59–98.5
98.5–130.5
Thiessen polygons:
Ranking of trade
between neighboring countries
b
Summed selection
Target: 30%
Cost: Equal
Selection frequency
900–1000
800–900
700–800
600–700
500–600
400–500
300–400
200–300
100–200
c
Summed selection
Target: 30%
Cost: Ranked trade connections
d
Difference
150–200
100–150
50–100
0–50
–50–0
–100 to –50
–150 to –100
Change in summed selection
due to collaboration costs

with other Mediterranean countries and also played an important role historically with the expansion of the Roman Empire 2000 years ago. Italy emerged in our analysis as a pivotal Mediterranean country, being a key importer from and exporter to other Mediterranean countries. Italy also has the highest number of shared marine boundaries with other Mediterranean countries—more than double the number of any other Mediterranean country. In addition, Italy has the largest-size EEZ (covering 21.3% of the whole Mediterranean Sea) and the largest number of threatened marine species shared with other Mediterranean countries (a median of 60; table 1). In contrast, some countries were found to be relatively isolated from other Mediterranean countries, with relatively weak economic ties to other Mediterranean countries (e.g., Israel). When evaluating the potential for collaboration between stakeholders, especially between nations, we also need to take into account historical and political factors such as governance instabilities and changing economic situations and crises. Given the history of armed conflicts between countries in the southeastern Mediterranean, new developments such as the recent findings of natural gas and oil in the deep sea will pose new challenges for marine conservation in the southeastern Mediterranean (Shaffer 2011, Khadduri 2012).

A unique example of potential Mediterranean collaboration in conservation is that of the only international MPA in the Mediterranean Basin, the Pelagos Sanctuary (Notarbartolo di Sciara et al. 2008; figure 6). Three other cross-boundary MPAs for marine mammals have been proposed by ACCOBAMS (see figure 6), which, if they are approved, will be shared among Spain, Morocco, and Algeria; between Italy and Malta; and between Greece and Turkey (Rais et al. 2006), all involving at least one country from the northern part of the Mediterranean Sea (figure 6). Collaboration to achieve conservation benefits already exists between some Mediterranean countries. An example for collaborative research is the set of marine surveys by Sala and colleagues (2012), which were conducted in the four countries with the most sites in the northern Mediterranean Sea: Spain (59 survey sites), Italy (52), Greece (30), and Morocco (6) (figure 6).

A clear link between the state and history of peace within a country and between countries and factors such as governance, economics, environmental awareness, and conservation has been demonstrated both in earlier studies (e.g., Neumayer 2002) and here (table 2). Democracy and a higher income were found to be favorable for promoting internal peace in various countries (Collier and Rohner 2008). It is also known that democratization reduces the risk of war (Gleditsch and Ward 2000). Trade has also been shown to promote peace between countries, because of the negative costs associated with violence that might deter countries from engaging in war (Hegre et al. 2010). These trends reinforce our suggestion that trade connections and the level of democracy can be used as surrogates for the potential success in conservation collaboration. Previous studies have mostly emphasized the negative impacts of economic activity on biodiversity, such as the increased density of invasive plants with trade imports in the Mediterranean (Vilà and Pujadas 2001) and the high risk of biological invasions resulting from the complex global network of cargo ship routes (Drake and Lodge 2004, Molnar et al. 2008, Kaluza et al. 2010). However, in our view, strong trade relations may also facilitate collaboration in other fields that may benefit conservation. In addition, trade may drive better environmental outcomes through multinational enterprises—for example, when multinational firms implement advanced environmental standards in developing countries (Rondinelli and Berry 2000). A lack of prior knowledge and the disregard of socioeconomic and political factors may be the cause of some conservation failures (Brechin et al. 2002, Bunnefeld et al. 2011, Fulton et al. 2011). Theory and tools are currently being developed to help better balance socioeconomic and conservation trade-offs in spatial conservation planning (Klein et al. 2010).

The proxies used here for predicting collaborative potential in conservation provide information that planners and decisionmakers can incorporate to account for political feasibility when setting up international marine conservation projects. Most previous studies have not accounted for this in the prioritization of conservation actions. In the example presented here, we showed how trade can be incorporated into a systematic conservation site selection tool as a surrogate of collaborative potential. In our analysis, using Marxan, we changed the selection likelihood of planning units by increasing or decreasing their cost, using the trade variable as a surrogate for the level of collaboration. We assumed that collaboration in conservation would be easier (i.e., the cost would be lower) between countries that also collaborate in other realms. Our collaboration scenario showed how the selection frequency for marine conservation shifts from the southern and eastern parts of the Mediterranean toward the northern and western parts of the Mediterranean

Figure 5. Results of the two Marxan prioritization scenarios aiming to conserving 30% of the occupancy area of 77 threatened species in the Mediterranean Sea while either ignoring collaborative potential or including it as a cost (see the "Mapping and quantifying collaboration" section). (a) Thiessen polygons dividing the Mediterranean Basin area, based on the nearest exclusive economic zone boundary (shown in thick black lines). Planning units within each Thiessen polygon were assigned a cost on the basis of the median value of the ranked trade variables between each pair of neighboring countries. (b) The selection frequency of planning units when no costs are included. (c) The selection frequency of planning units when costs were based on trade connections between countries. (d) The difference in the selection frequency between the two collaboration scenarios.

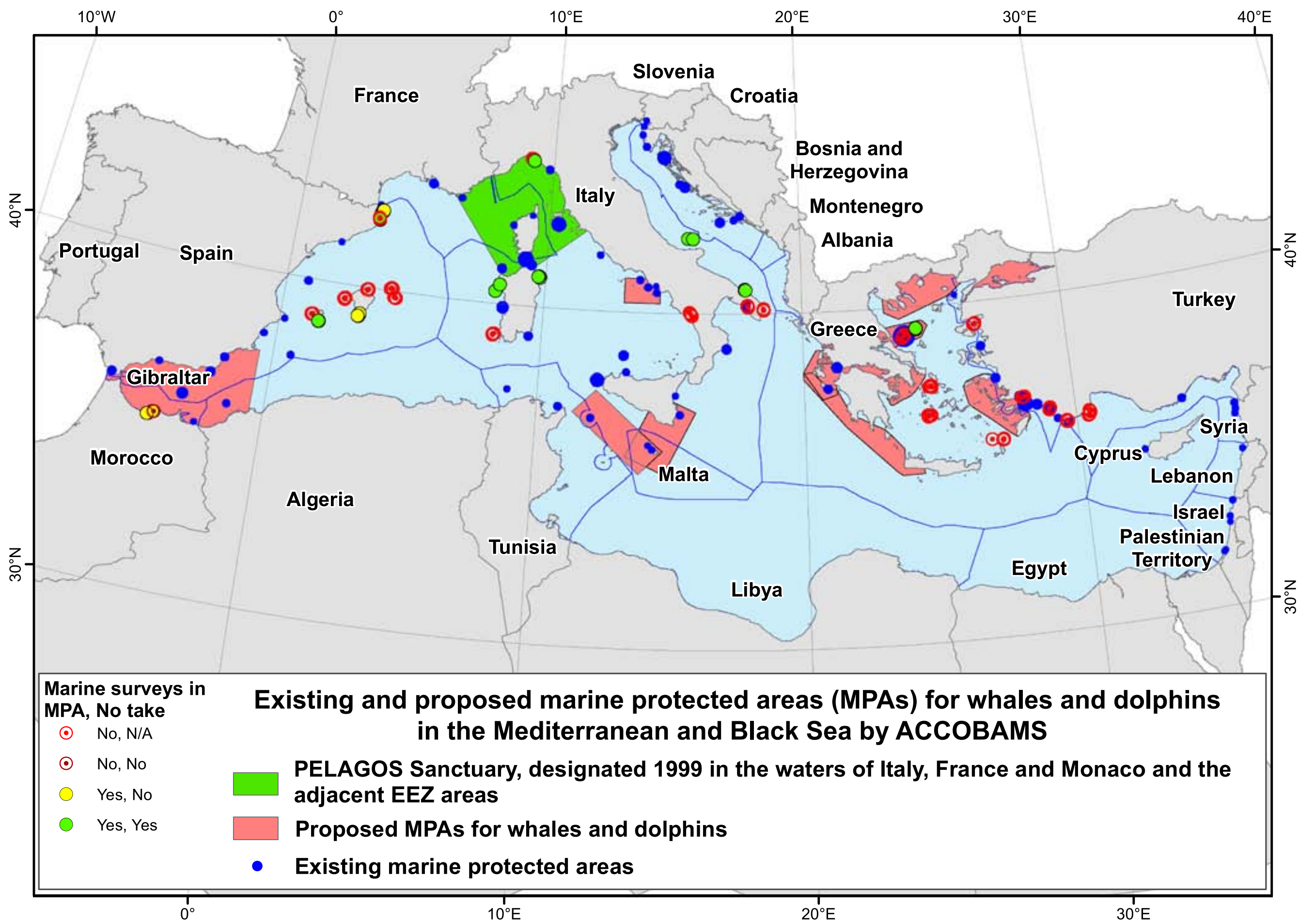


Figure 6. Current and proposed marine conservation activities in the Mediterranean Sea. The lines in the sea depict exclusive economic zones. Existing marine protected areas (from Abdulla et al. 2008) are shown with blue dots. The polygons show existing (green in the legend) and proposed (red in the legend) large protected areas for marine mammals. The locations in which marine surveys were conducted by Sala and colleagues (2012) are shown with four different point symbols based on the level of protection of the site. No take refers to no-take marine protected areas.

(figure 5d) when proxies for collaboration were taken into account. This spatial bias in current conservation efforts in the Mediterranean Sea is also reflected by the present spatial distribution of proposed conservation areas across the region (figure 6; Abdulla et al. 2008).

Conclusions

Transboundary conservation programs are increasing globally in both the terrestrial (Halpern et al. 2005) and marine (Mackelworth et al. 2012) realms, and new approaches are required for estimating the potential for collaboration success between stakeholders when taking conservation action. We have demonstrated one approach at a multinational level, and similar analyses accounting for different aspects of uncertainty and socioeconomic information are possible at smaller scales—for example, using bioeconomic modeling (Stewart and Possingham 2005), applying more complex models predicting probability of collaboration success, and including the growing literature on opportunity costs in conservation planning (Adams et al. 2011). The example of marine conservation in the Mediterranean Sea presented here can be used as a framework for incorporating a range of socioeconomic factors into conservation planning in other complex regions. Unraveling these socioeconomic factors into meaningful collaborative ties for conservation can help facilitate successful international collaboration and can ultimately help achieve more cost-effective conservation outcomes.

In the conservation-planning case study analyzed here, we used trade as our surrogate for collaborative potential between countries. Additional factors worth exploring in future studies include countries that are not immediate neighbors and how decisions might change using other proxies that might reduce the estimated costs of collaboration in conservation. These other proxies include tourism, shared international agreements, or the history of conflicts between countries. The cost of conservation could also be adjusted in accordance with the difficulty of implementing conservation actions—for example, the willingness of an actor (in the present study, a country) to take an environmental action (e.g., Knight et al. 2010). Proxies such as the degree of democracy and governance or the percentage of a country set aside for terrestrial protected areas might be useful for assessing this, although the causal link between effectiveness of environmental actions and governance has not yet been clearly demonstrated (Bäckstrand 2006). Finally, costs can be modeled using weighted distance functions (Levin et al. 2007), which are inversely related to the distance from the coastline (assuming that negative impacts of terrestrial activity on marine systems mostly originate from the coast).

In summary, in the present study, we present a framework for integrating collaborative potential into systematic conservation planning. Our analysis shows that taking surrogates for collaborative potential into account can alter our spatial priorities. Within the Mediterranean Sea, where collaboration between countries is essential for protecting its unique biodiversity, the approach proposed here can help identify areas in which future transboundary MPAs and collaborative initiatives for marine conservation may be more likely to succeed (or less costly). The approach can also be a guideline for international nongovernmental organizations (NGOs) to determine where their funding allocations may be more successful. Alternatively, these results can be used to indicate areas in which extra resources and time are required to facilitate collaborative conservation planning and management.

Existing sociopolitical and economic ties between northwestern European countries may enhance the potential of future conservation efforts among these countries. Because, as was discussed above, the European Union already has in place many of the institutions required for building these collaborations, concrete actions might be put into place in the very near future without much outside international facilitation. Other parts of the Mediterranean Basin may require more international support (e.g., of international conservation NGOs) in order to facilitate potential collaborative conservation efforts. One of the first steps that should be taken in order to advance cross-boundary conservation planning and the establishment of large cross-boundary MPAs in the Mediterranean would be the mutual agreement between countries of their EEZs. The framework developed in the present study for the Mediterranean Sea can be further applied to other complex marine and terrestrial regions in which multiple countries share ecosystems, conservation targets, and other environmental resources, such as in the Coral Triangle, the Caribbean Sea, and the Black Sea.

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Appendix 2.

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POINT-OF-VIEW

Advancing marine conservation planning in the Mediterranean Sea

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Abstract Twenty leading scientists in the field of marine conservation planning attended the first international workshop on conservation planning in the Mediterranean Sea. This globally significant biodiversity hotspot has been subjected to human exploitation and degradation for 1,000s of years. Recently, several initiatives have tried to identify priority areas for conservation across the Mediterranean Sea. However, none of these efforts have led to large-scale actions yet. The aim of the workshop was to establish a network of scientists who are involved in large-scale conservation planning initiatives throughout the Mediterranean basin to promote collaboration and reduce redundancy in conservation initiatives. The three focus groups of the workshop build on existing efforts and intend to deliver: (1) a roadmap for setting conservation priorities, (2) a methodological framework for linking threats, actions and costs to improve the prioritization process, and (3) a systematic conservation planning process tailored to complex environments such as the Mediterranean Sea. Joining forces and involving more scientists (especially from the South-eastern part of the region) in following meetings, the participants endeavour to provide guidelines on how to bridge the science-policy gap

Please refer the ''Appendix'' section for the workshop participants-authors.

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and hence aid decision-makers to take efficient conservation actions.

Keywords Biodiversity · Collaboration · Marine conservation planning · Mediterranean Sea · Marine protected areas

Background

Despite the agreement by most Mediterranean countries to conserve 10 % of the sea by 2020 under the Convention on Biological Diversity (http://www.cbd.int/convention/), only ~4 % of the Mediterranean Sea is currently included in marine protected areas (MPAs) and merely 0.01 % is designated as no-take reserve (Abdulla et al. 2008; Portman et al. in print). The Mediterranean Sea is a biodiversity hotspot with nearly one-fifth of the total known number of marine species world-wide, which has been subjected to human exploitation for centuries (Bianchi and Morri 2000; Coll et al. 2010). Current MPAs only partially protect fundamental biodiversity traits of the Mediterranean (Mouillot et al. 2011). Therefore, the need to expand and increase the number of spatially managed areas in the region to progress towards an ecosystem-based approach to marine resources (Pikitch et al. 2004) and ecosystem-based marine spatial management (Katsanevakis et al. 2011) is essential. Such necessity is also highlighted by the European Commission with the adoption of the Marine Strategy Framework Directive (2008/56/EC). The directive specifically calls for the establishment of a network of MPAs in European waters.

Recently, several large-scale conservation initiatives for the entire Mediterranean Sea have suggested increasing the number and extent of MPAs in the region (see Oceana 2011 and references therein). Intergovernmental bodies have identified priority areas for conservation (e.g. the European Union, the Regional Activity Centre for Specially Protected Areas of the United Nations Environmental Programme—Mediterranean Action Plan), NGOs (such as Greenpeace, Oceana), scientific committees (Scientific, Technical and Economic Committee for Fisheries and General Fisheries Commission for the Mediterranean), regional scientific commissions and Agreements (CIESM-The Mediterranean Science Commission, ACCOBAMS-Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and contiguous Atlantic area) and research groups (Coll et al. 2012; Micheli et al. http://globalmarine.nceas.ucsb.edu/mediterranean/). However, little or no action has yet been taken to act on these large-scale plans or recommendations in response to their valuable findings. The socio-economic, political and cultural complexity in the Mediterranean Basin, which comprises 21 countries, could be a possible explanation for this delay (Kark et al. 2009).

The great deal of current research in the region often lacks coordination. With the increasing threats to the region's biodiversity, scientists and managers are faced with the need to integrate their efforts to more efficiently conserve the biodiversity of the Mediterranean Sea. The production of multiple maps with both alternative and converging information, while useful in serving specific aims, should provide clarity and coordination among plans to better inform decision-makers. Furthermore, there is a need to explicitly account for the complexity of the Mediterranean Basin and to advance the bridge between science and policy issues. Basic science and theory need to be translated into real action in order to foster marine protection and secure the valuable ecosystem services that the Mediterranean Sea provides.

Within this context, the first workshop on advancing marine conservation planning in the Mediterranean Sea was organised last April in Santorini (Greece) to create a network of scientists who are involved in large-scale spatial conservation planning initiatives throughout the basin. The aim of the workshop was to promote collaboration and reduce redundancy in future scientific contributions. Building on existing efforts, the workshop participants discussed: (1) a roadmap for setting conservation priorities, (2) a process for integrating and incorporating threats, actions and costs into the prioritization process, and (3) recommendations for a systematic conservation planning process in complex environments such as the Mediterranean Sea.

The workshop

The twenty workshop participants came from several Mediterranean countries—from Spain and Italy, to Greece and Israel. Experts from the USA, Australia and Canada also attended. This brought together knowledge and expertise from NGOs, universities, research institutes and the European Union.

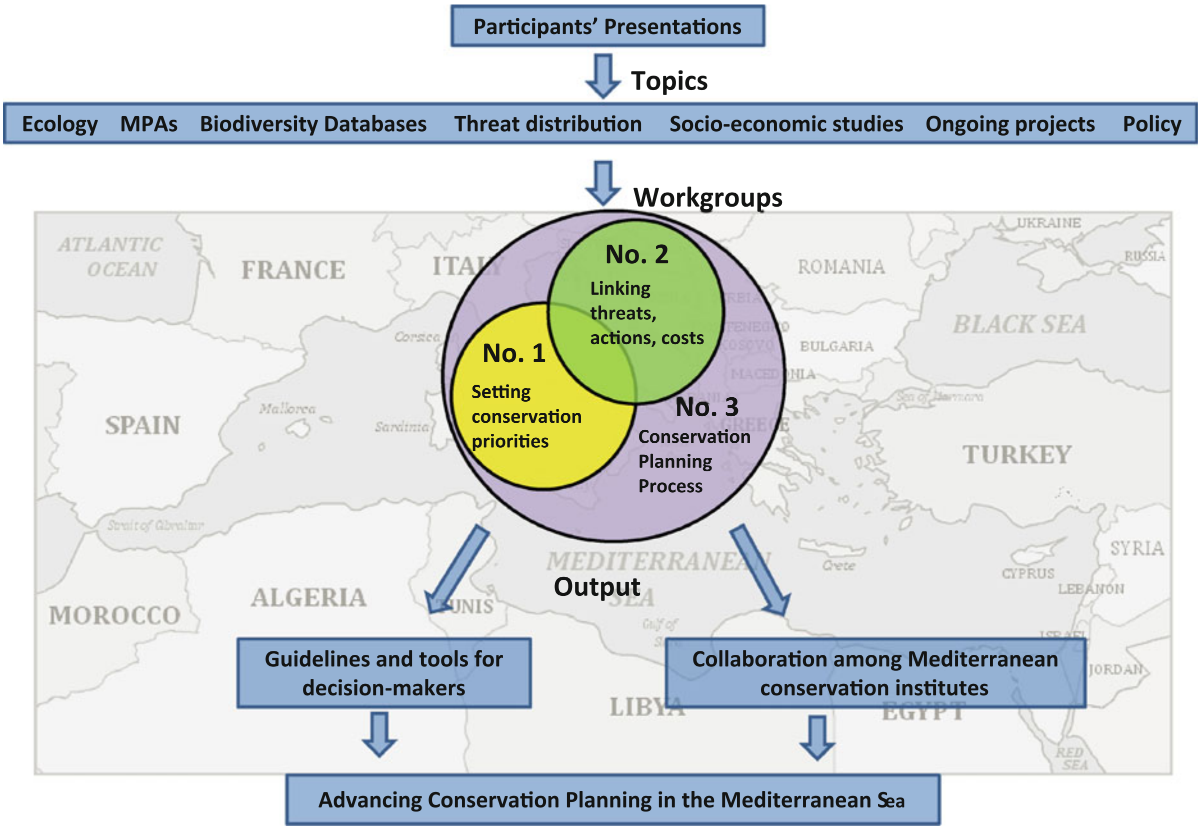

Fig. 1 Steps towards advancing marine conservation in the Mediterranean Sea. Broad topics presented by the workshop participants are illustrated on the *top* of the *chart*; the subjects of the three working groups appear in the *circles* in the *centre* of the *chart* whereas the outputs of the workshop at the *bottom* of the *chart*

During the first part of the workshop, each of the participants presented their own research on several interrelated topics (Fig. 1). In these presentations, the majority of speakers brought the current status of marine conservation in the Mediterranean Sea to attention. A map of the sea's current MPAs was referred to in many presentations to highlight the need for greater protection of biodiversity (Abdulla et al. 2008, 2009) and also to make the existing situation clear. From this, the presenters led into their efforts and projects to improve conservation within the basin. NGO representatives referred to their efforts to establish new MPAs in the Mediterranean and improve the management of existing ones, with special emphasis and efforts being directed towards eastern and southern areas of the Mediterranean Sea (Marshall et al. 2009). Several academic researchers explained the range of systematic tools and approaches for the identification of priority areas for conservation in the Mediterranean Sea, while other researchers presented available spatial data to identify threats to marine species and habitats at a Mediterranean Basin scale (Micheli et al. http://globalmarine.nceas.ucsb.edu/mediterranean/; Giakoumi et al. 2011; Coll et al. 2012; Sala et al. 2012; Portman et al. print).

Experience and up-to-date computational and modelling tools such as Marxan (Ball et al. 2009) and ecological modelling software (Christensen and Walters 2004) used in other parts of the world such as Australia, the USA and Canada were introduced within the workshop. Participants discussed how modelling tools should be considered and adapted to regional conditions, while new examples of applications to Mediterranean ecosystems were presented (Fraschetti et al. 2009; Giakoumi et al. 2011; Coll and Libralato 2012). Successful examples of large-scale MPA networks, such as the Great Barrier Reef MPA network (Fernandes et al. 2005) could provide guidance on how to deal with fundamental conservation planning issues. The adaptations of current approaches and planning methods and the development of novel ones to address the complexity of the region have been largely discussed in the workshop.

Participants also discussed EU programs (MEDITS and Med SEA), web-platforms (EMIS and EASIN), the ecosystem approach of the Mediterranean Action Plan and EU projects (CoCoNet, NETMED) relevant to basin-scale conservation planning. Larger and smaller scale efforts were also discussed, via global models of the world's oceans (e.g. Christensen et al. 2012), and specific regional studies such as the Adriatic Sea (Mackelworth et al. 2011; Mackelworth and Caric 2010), the Aegean and Ionian Seas (Giakoumi et al. 2011, in revision; Stelzenmüller et al. in print).

A subsequent plenary session focused on the following subjects:

(1) *Scarcity of data in the southern and eastern Mediterranean Sea*

One impediment to prioritizing initiatives for the entire Mediterranean has been the lack of data and the poor representation of the eastern Mediterranean Sea (Claudet and Fraschetti 2010; Fraschetti et al. 2011; Coll et al. 2010, 2012). Methods should be devised to overcome this problem. One example could be the use of surrogates for biodiversity, threats and cost in data poor regions, i.e. geo-morphological and oceanographic data (Giakoumi et al. 2011). Data uncertainty and subsequent biases can be taken into account in prioritization schemes so lack of complete data is not an excuse for inaction. Moreover, diverse approaches may be required in different ecoregions according to data availability.

(2) *The complexity of the region should be taken into account in conservation planning initiatives*

The Mediterranean Sea, almost completely enclosed by land, has been an important route for merchants and travelers since antiquity allowing for trade and cultural exchange among civilizations. Currently 21 modern states share the Mediterranean coastline. These states present important differences in terms of economic status, political regime, culture and religion. This heterogeneity has generated significant collaborative achievements but also severe conflicts. In such a complex environment, opportunities and obstacles for collaboration in conservation efforts among States should be considered (Marshall et al. 2009; Kark et al. 2009).

(3) *From MPA planning to action*

Why is the Mediterranean Sea receiving so little protection? Explicit and quantitative consideration of socio-economic activities when identifying priority conservation areas could aid decision-making. Up to now systematic planning approaches, explicitly considering opportunity cost, have been applied only to national-scale projects (Fraschetti et al. 2009; Maiorano et al. 2009; Giakoumi et al. 2011; in revision). Furthermore, identification of threats to biodiversity, habitats and ecosystem processes is crucial (Coll et al. 2012; Micheli et al. http://globalmarine.nceas.ucsb.edu/mediterranean/). Equally crucial is the distinction between threats that can be mitigated (e.g. fishing pressure) and those that cannot (e.g. climate change). A homogenous plan of conservation actions throughout the Mediterranean Basin may not be possible. The socio-economic complexity of the Mediterranean Sea requires different strategies for conservation planning, adapted to different contexts by region.

Outreach

A representative from the cabinet of the European Commissioner on Marine Affairs and Fisheries, Mrs Maria Damanaki, attended the workshop in order to report to the European Commission about the efforts required to increase the proportion of the Mediterranean Sea currently protected. The workshop was reported on Greek television, in an attempt to raise people's awareness on marine conservation issues.

In the framework of the workshop, a website was created to host the material presented, provide background information on the workshop, and to form a basis for collaboration and discussion within and among the working groups. The site is accessible to the public at large: https://sites.google.com/site/conservationmediterraneanws1/.

Priorities and future opportunities

Three main topics concerning conservation planning in the Mediterranean Sea emerged from the general discussion. To explore possible solutions on these subjects, the participants were divided into three working groups, each proposing a different strategy for action. The outcomes from these groups will be published in peer-reviewed journals.

Group 1: setting conservation priorities—a cookbook approach

This working group aims at reviewing existing large-scale conservation initiatives and suggestions for the Mediterranean Sea in order to identify areas that emerge as priorities regardless of the planning criteria and methods used. When setting priorities, biases due to data uncertainty must be accounted for. This working group intends to identify these biases and suggest ways to incorporate them, while also detecting a suitable scale for prioritizing actions in the Mediterranean Sea. Another question that emerged during discussions was whether conservation actions and their sequence should be the same throughout the Mediterranean region. Ecological, economic and social divergence among Mediterranean countries dictates that a holistic approach for the entire Mediterranean Sea is likely to fail. The targeted outcome of the group is a roadmap to guide the setting of priorities in complex regions such as the Mediterranean Sea.

Group 2: linking threats, actions and costs

Recent efforts to identify and map threats to biodiversity and habitats of the Mediterranean Sea are very informative, but it is still unknown how this information can be used to advance conservation planning. This group discussed how to summarize available information on threats and move towards a framework for linking threats to conservation actions and further quantify the costs of mitigating the threat as a useful way to use available data. The group is working on a study that will describe the commonalities of existing systematic analyses of threats that have been done, while also showing that there is room for refinement by looking at synergies, better resolution of data, deep-water data and other critical aspects of the analyses that are currently lacking. Documenting successful stories of recoveries in the Mediterranean Sea following the work of Lotze et al. (2011), the group is presently analysing the specific actions involved in past experiences that led to reverting or arresting trajectories of changes resulting from threats. In addition, the use of two case studies on the endemic seagrass *Posidonia oceanica* and loggerhead turtle *Caretta caretta* will illustrate in detail how simple actions can limit habitat degradation and help conserve species of importance, while linking specific threats to actions and to costs.

Group 3: providing a general framework for systematic conservation planning and policy in complex regions

The third group focuses on developing a model planning process that can expedite marine conservation in the Mediterranean within existing institutional frameworks. Currently, spatial planning is invariably hampered because stakeholders often cannot agree on the boundaries of the areas within which planning will occur. However, the exact boundaries at a local scale are of limited importance especially when designing a network of MPAs. This group agrees that the appropriate scale for developing the first detailed marine zoning plans in the Mediterranean is the ecologically or biologically significant areas (EBSAs) (Notarbartolo di Sciara and Agardy 2009; UNEP-MAP-RAC/SPA 2010). Some scientists and decision makers in the region argue that marine spatial planning in the Mediterranean Sea cannot move forward because of insufficient data. This group aims at demonstrating that data available on the EBSAs are sufficient to make credible plans and that insufficient data is no longer an excuse for inaction. A case study will be described to illustrate the main conclusions of the group's work.

After this workshop, participants and hence the institutions they represent are ready to network and exchange information concerning their projects, datasets, research protocols and approaches, sources of literature, on-line databases and other resources. Several new collaborations among scientists within the framework of ongoing Mediterranean-scale conservation projects have already been started as a result of this exchange.

Summarizing, the first international workshop on conservation planning in the Mediterranean Sea gave experts from different parts of the Mediterranean Basin and overseas institutions an opportunity to establish fruitful collaborations. Ongoing EU projects concerning marine conservation planning will help identify gaps and create further opportunities for collaboration among Mediterranean research institutions.

Participants are already organizing a second workshop for spring 2013 in a Mediterranean country. In this next meeting, more scientists will be involved, especially from the southern and eastern Mediterranean regions. We invite scientists and managers interested in the area to join the effort, more information can be found: https://sites.google.com/site/conservationmediterraneanws1/. We believe that collaboration among experts and institutions as well as coordination of ongoing conservation planning projects are necessary for bridging the science-policy gap and the uptake of conservation action across the Mediterranean region.

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Appendix

Workshop participants

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