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Marine forest restoration in a fast-changing Mediterranean Sea

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ABSTRACT

Marine forests are amongst the most important rocky shores habitats worldwide. In the Mediterranean Sea, forests composed by fucalean brown algae of the genus *Cystoseira sensu latu* have a crucial role for both intertidal and subtidal ecosystems, supporting marine biodiversity and providing several ecosystem services. In the last twenty years, cumulative impacts of local anthropogenic pressures, combined with global climate changes are driving the decline of these forests in many areas of the basin with cascading effects and severe consequences on the whole marine-coastal ecosystem. Despite the robust legislative framework orbiting around them, specific conservation measures for the protection of these habitat-forming species have never been implemented and restoration represents one of the most promising approach to halt their decline.

To make restoration interventions consistently successful, i) the assessment of their actual status, ii) the evaluation of the stressors that caused their decline or disappearance, iii) a detailed mapping of their present and past distribution, and iv) the identification of sites where restoration interventions are more likely to be effective, are urgently needed. Present thesis has been conceived to address these issues adopting different approaches with strongly connected chapters. In the first Chapter, a Machine Learning technique to develop a Habitat Suitability Model for *Cystoseira s.l.* forests and the analytical tool NEAT (Nested Environmental status Assessment Tool) were applied to assess macroalgal forests status; in the second Chapter, a multivariate statistical approach was adopted to explore critical stressors that may cause forests regression across different spatial scales; in the third Chapter, a systematic literature review allowed to deepen the understanding about past and current forests distribution; in the fourth Chapter, a spatial prioritization analysis was performed to support the planning of future restoration efforts. All analyses were performed at Mediterranean scale with the exception of Chapter 3 where the Italian coasts were selected as case study to assess drivers of changes. Taken as a whole, my thesis support macroalgal forest restoration at basin scale, providing guiding principles and criteria to scale-up and step-up the effectiveness of restoration actions.

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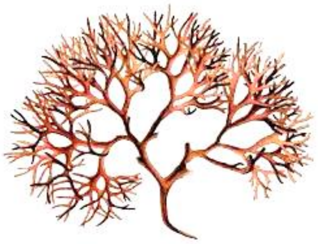

GENERAL INTRODUCTION

I Marine Ecosystems Restoration

Awareness of the need for environmental repair and solution for ecosystem recovery is growing across the globe (Gann et al., 2019). Terrestrial and marine biodiversity is decreasing at unprecedented rates as a result of the influence of human activities (Baillie et al., 2004; Micheli et al., 2013; Airoldi et al., 2020), and natural recovery can be difficult in a reasonable time frame (Dobson et al., 1997; Lotze et al., 2006). Although protecting remaining native ecosystems is critical to conserving the world's natural heritage, protection alone is insufficient, given past and current degradation, resulting in a global escalation of ecological restoration and related efforts (Gann et al., 2019)

Ecological restoration, i.e., the process of actively assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SER, 2004), is considered an effective strategy to supplement current conservation and management actions, providing benefits relating to biodiversity, climate, economic growth, and physical and mental well-being (Fraschetti et al., 2021). The United Nations Sustainable Development Goals (UN SDGs) for 2030, The Convention on Biological Diversity (CBD, 2020), and the EU Biodiversity Strategy for 2030 (EC, 2020) call for restoration as a prominent tool to halt and reverse trends of loss and degradation, bringing back biodiversity and the associated ecosystem services (Bevilacqua et al., 2021; Tamburello et al., 2022). The concept of restoration in many national and international initiatives and agreements is very broad and in 2019 the "United Nation Decade on ecosystem restoration 2021–2030" has been declared (Waltham et al., 2020).

Nevertheless, repairing degraded ecosystems is a complex task requiring significant time, resources, and knowledge. Many restoration interventions, despite well intentioned, have underperformed. There is an urgent need for principles to underpin restorative activities defined by the UN Decade, which have to be applicable across sectors, biomes and regions (FAO, IUCN

CEM and SER, 2021). The Society of Ecological Restoration (SER) and international partners produced "The International Principles and Standards for the Practice of Ecological Restoration" (the Standards, Gann et al., 2019) providing a cross-sectoral framework of principles (Figure I.1) that can improve ecological restoration outcomes across all ecosystems.

Figure I.1 Eight principles for ecological restoration. Source Gann et al. (2019)

However, whilst there is a relatively good understanding of elements supporting terrestrial systems restoration (Wortley et al., 2013), similar knowledge is lacking in coastal and marine systems (Blignaut et al., 2013). Besides differences in the nature and scale of degrading processes, which historically affect the way we manage biotic resources on land and in the sea (Carr et al., 2003), this discrepancy is also due to marine ecosystems being largely "out-of-view," leading to a lack of understanding of the degree of degradation and the needs for restoration practices (Parsons et al., 2014), as well as the intrinsic difficulties and elevated costs associated with interventions in the marine environment (Bayraktarov et al., 2016), especially in remote areas (Van Dover et al., 2014). A global analysis by Fraschetti et al. (2021) has recently highlighted the main reason behind failures of restoration interventions in the marine realm, providing recommendations to drive future marine restoration projects. Firstly, contextual factors, such as the ecosystem type and geographic location, are found to be determinant for restoration success, demonstrating the vital importance of site selection, for example, in terms of knowledge of the pressures present within the region, its historical context, and the wider seascape. Finding suitable and priority areas where to plan active restoration actions is a crucial step in this process and requires the adoption of the principles and tools generally adopted in the framework of marine spatial planning, conservation planning, and spatial ecology. At this purpose, there is an urgent need of fine-scale data about past and present distribution of the target of restoration and of human uses together with the knowledge of environmental variables and local socio-economical features potentially affecting the outcomes of restoration.

Secondly, absence of protocols and best practices for supporting the recovery of ecosystems resulted as major issues too, including the lack of standardized criteria for assessing and measuring the “success” of restoration actions. This also requires the definition of specific restoration targets built on baseline ecological information on species and habitats, combined with the knowledge about their status and causes of degradation.

Ultimately, restoration must be implemented at multiple scales to achieve measurable effects despite large scale pressures. Most restoration projects were carried out over relatively small areas (<1 ha) which fails to match the scale of human disturbance. Upscaling marine restoration in space and time is imperative to provide desired ecological benefit, ensuring net-positive environmental changes. In this respect, an inclusive participation with a concrete convergence among scientists, private sectors, and stakeholders leading to a shared vision, should be promoted throughout the process. This broad engagement can be enabled only by policies and measures, including intersectoral policies, fostering the replication and the scaling-up of restoration actions over the long term.

II Marine Forests in the Mediterranean Sea

Habitat formers are common targets of marine and coastal ecosystem restoration, not only to reverse species local decline, but also to improve and provide habitat for other species of commercial values (Tamburello et al., 2022), contrasting the shift from complex three-dimensional habitat to simpler and less productive ones (Airoldi et al., 2008; Claudet and Fraschetti, 2010).

Marine forests composed by fucalean macroalgae (*Cystoseira sensu latu*, including the genera *Cystoseira*, *Ericaria* and *Gongolaria*; Molinari and Guiry, 2020) are critical habitats of intertidal and subtidal reefs in the Mediterranean Sea (Sala et al., 2012) and an excellent case study to show gaps strongly limiting restoration interventions in the marine environment. They dominate several assemblages from the very shallow to deep waters (−50 m) (Feldmann, 1937; Giaccone and Bruni, 1973; Ballesteros et al., 1998; García-Fernández and Bárbara, 2016). providing shelter, food and nursery areas to a multitude of marine communities. In addition, they increase the three-dimensional complexity and spatial heterogeneity of rocky substrates, enhancing biodiversity and primary productivity in coastal ecosystems, and maintaining healthy the functioning of rocky habitats (Cheminée et al., 2013; Gorman et al., 2013; Piazzi et al., 2018). As a consequence, several species (*C. sedoides*, *E. amentacea* var. *stricta*, *E. mediterranea*, *E. zosteroides*, *G. montagnei*) have been protected since 1982, with the enforcement of the Bern Convention (1979). In 2009, an amendment of the Mediterranean Action Plan (Annex IV, SPA/BD Protocol - United Nations Environment Programme) adopted within the framework of the Barcelona Convention (1976), identified the conservation of all but one (*C. compressa*) Mediterranean *Cystoseira s.l.* species as a priority. Despite the robust legislative framework, specific conservation measures for the protection of these habitat-forming species have never been implemented (Fraschetti et al., 2011). For example, the selection of marine sites deserving protection under the Natura 2000 Sites network[1] in the Mediterranean is generally based on the presence of *Posidonia oceanica* meadows, while the presence of *Cystoseira s.l.* is only incidental, as brown macroalgal forests are not listed in the Habitat Directive annexes (Directive 92/43 EEC).

The cumulative impacts of local anthropogenic pressures combined with other global stressors are driving the decline of *Cystoseira s.l.* forests and their associated communities in several regions of the Mediterranean Sea subject to increased human threats (Airoldi and Beck, 2007; Thibaut et al., 2015). Decrease in water quality (Soltan et al. 2001, de Caralt et al. 2020), climate-driven stressors (Verdura et al. 2021), coastal urbanization (Thibaut et al. 2005, Mangialajo et al. 2008, Sales and Ballesteros 2009, Perkol-Finkel and Airoldi 2010),

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[1] https://ec.europa.eu/environment/nature/natura2000/index_en.htm

aquaculture (Boudouresque et al. 2021) and overgrazing by sea urchins and herbivorous fish (Fraschetti et al. 2001, Sala et al. 2011, Guarnieri et al. 2014, Vergés et al. 2014, Tamburello et al. 2022) are among the main reasons causing their regression and their replacement by turf-forming, filamentous or other ephemeral seaweeds. Due to their role in supporting biodiversity and food webs, the loss of these long-lived brown algae is leading to a decrease in critical ecosystem services such as fisheries (Thibaut et al., 2015; Buonomo et al., 2018) and also to a reduction in the potential to sequestrate carbon dioxide and to mitigate climate change.

Since natural recovery of *Cystoseira s.l.* forests hase rarely been observed (Perkol-Finkel and Airoldi, 2010; Cebrian et al., 2021) and considering the low dispersal capacity of fucalean algae (Verdura et al. 2018, Riquet et al. 2021), the protection of remnant algal forests within Marine Protected Areas (MPAs) may be insufficient for prompting their recovery (Sala et al. 2012). For this reason, an active restoration has been proposed as a promising approach to halt their decline (Gianni et al., 2013), and several European projects (e.g., MERCES[2], and AFRIMED[3]) developed and tested new techniques to reintroduce *Cystoseira s.l.* species after local extinction, regenerating self-sustaining populations (Verdura et al., 2018).

Yet, to make restoration interventions consistently successful, a detailed knowledge of their present and past distribution, the individuation of donor populations, and the identification of the stressors that caused their decline or disappearance together with the evaluation of their mitigation state is still lacking. Furthermore, criteria, targets, and methods applied until now have been highly heterogeneous, making results from various macroalgal restoration projects scarcely comparable. Filling gaps of knowledge is a prerequisite to reverse current patterns of regression supporting the planning of restoration interventions and informing the selection of: 1) the restoration sites; (2) the target species; (3) donor sites and specimens; (4) restoration techniques; and (5) complementary actions (Figure II.1, Cebrian et al., 2021). These crucial steps, together with the development of interdisciplinary synergies and cooperation among decision-makers and stakeholders, are paramount to improve the restoration of the Mediterranean macroalgal forests.

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[2] http://www.merces-project.eu/

[3] http://afrimed-project. eu/

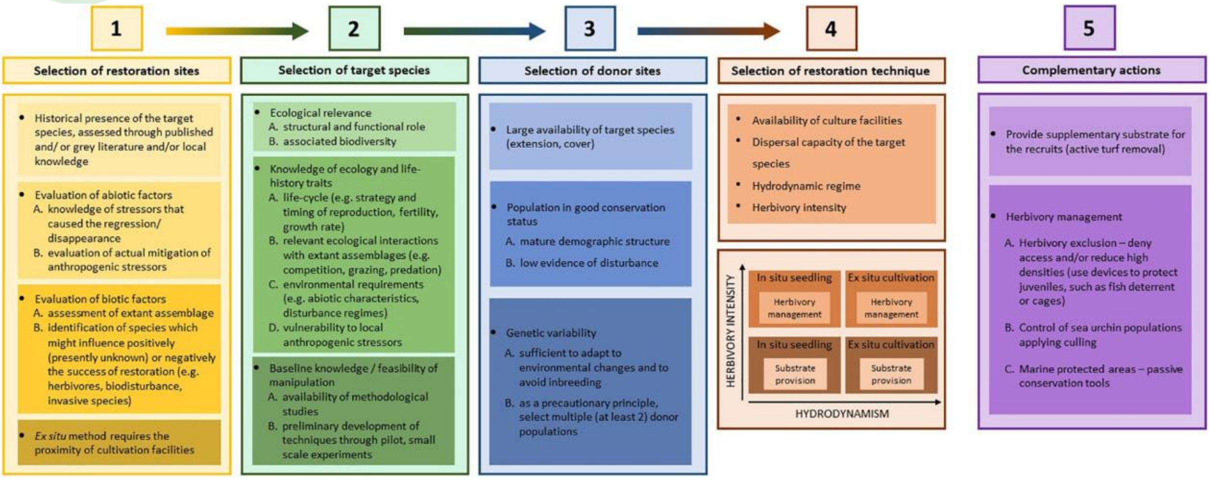


Figure II.1 Conceptual model of the crucial steps to guide the restoration of macroalgae forests. Source Cebrian et al. (2020)

III Aims and structure of the thesis

The present thesis has been developed as a series of studies already published for publication. The common thread binding all chapters is the need to fill the huge gap of knowledge in this field, using macroalgal forests of the Mediterranean Sea as a perfect example to test how to scale-up and step-up marine restoration. Each chapter, infact, disentangles specific issues linked to different steps of the restoration process described above (see paragraph II). Analyses were all performed with the idea to provide assessment and guidelines at basin scale (except those of Chapter 3 zooming into the Italian coasts).

Figure III.1 Thesis structure. Each part of the diagram represents the topic of the corresponding chapter/manuscript of the thesis: 1) "Assessing the status" corresponds to Chapter 1 divided into the two parts: Part I "*Modeling macroalgal forest distribution at Mediterranean scale: present status, drivers of changes and insight for conservation and management"* and Part II "*An integrated assessment of the Good Environmental Status of Mediterranean Marine Protected Areas"*; 2) "Identifying drivers of changes" corresponds to Chapter 2 "*Effects of natural and anthropogenic stressors on fucalean brown seaweeds across different spatial scales in the Mediterranean Sea"*; 3) "Mapping areas of loss" corresponds to Chapter 3 "*Can we preserve or restore overlooked macroalgal forests?"*; 4) "Planning restoration actions" corresponds to Chapter 4 "*The challenge of setting restoration targets for macroalgal forests under climate changes"*.

In the Figure III.1, each part of the diagram represents the main topic addressed in the corresponding chapter/study of the thesis: 1) "Assessing the status" corresponds to Chapter 1 divided into the two parts: Part I "*Modeling macroalgal forest distribution at Mediterranean scale: present status, drivers of changes and insight for conservation and management"* and Part II "*An integrated assessment of the Good Environmental Status of Mediterranean Marine Protected Areas*"; 2) "Identifying drivers of changes" corresponds to Chapter 2 *Effects of natural and anthropogenic stressors on fucalean brown seaweeds across different spatial scales in the Mediterranean Sea*; 3) "Mapping areas of loss" corresponds to Chapter 3 *Can we preserve or restore overlooked macroalgal forests?*; 4) "Planning restoration actions" corresponds to Chapter 4 *The challenge of setting restoration targets for macroalgal forests under climate changes*. A conceptual order drives the sequence by which the chapters are here presented and the topic discussed in each chapter is part of a single process leading to an informed planning of restoration actions. Aims of each chapter/study of the thesis and connections among them are specifically described in the next four paragraphs. As authorship, I am the first author of manuscript presented in Chapter 1 Part I, second author of manuscript presented in Chapter 1 Part II, co-author of the manuscript presented in Chapter 2, second author of the manuscript presented in the Chapter 3 and first author of the manuscript presented in the Chapter 4.

III.1 Assessing the status (Chapter 1)

In the first Chapter of the thesis, the manuscript entitled *Modeling macroalgal forest distribution at Mediterranean scale: present status, drivers of changes and insight for conservation and management* is presented in Part I. It was published on Frontiers in Marine Science journal in 2020 (Fabbrizzi et al., 2020). The aims of this study were i) to synthesize knowledge about the distribution of *Cystoseira s.l.* species at Mediterranean scale and ii) to develop a Habitat Suitability Model (HSM) for *Cystoseira s.l.* species living in the shallow rocky substrates by means of a Random Forest technique (RF) (Breiman, 2001), a very effective Machine Learning technique. The developed HSM has been used for the assessment of species distribution on shallow rocky shore predicting the occurrence of *Cystoseira s.l.* canopies at locations where information was not available, identifying suitable and unsuitable areas for holding *Cystoseira s.l.* species. Also, the importance of different environmental predictors in shaping their spatial distribution was discussed.

Part II of this Chapter is represented by the manuscript entitled *An integrated assessment of the Good Environmental Status of Mediterranean Marine Protected Areas*, published on Journal of Environmental Management in 2022 (Fraschetti et al., 2022). The aim of this work was to assess if Marine Protected Areas (MPAs) reach the GES and if their effects are local or can be detected at ecoregion level or up to a Mediterranean scale. To do this, an ecosystem approach, evaluating the status of several ecosystem components, was adopted. This study, taking into account macroalgal forests among the ecosystems components driving GES achievement, was crucial to assess their status across the Mediterranean Sea, particularly with regard to the role of MPAs. The analyses were undertaken using the software NEAT (Nested Environmental status Assessment Tool), a tool which allows an integrated assessment of the status of marine systems.

The two parts of this chapter, both assessing the present status of macroalgal forests across the Mediterranean basin, has provided a critical baseline to understand the drivers of changes (Chapter 2), refine the knowledge about forests distribution and the role of MPAs in protecting them (Chapter 3), improve site prioritization for their management, conservation and restoration (Chapter 4).

III.2 Identifying drivers of changes (Chapter 2)

The second Chapter is represented by the manuscript entitled *Effects of natural and anthropogenic stressors on fucalean brown seaweeds across different spatial scales in the mediterranean sea* published on Frontiers in Marine Science journal in 2021 (Orfanidis et al., 2021). This study explore the key drivers that influence the distribution and abundance of fucalean forests to determine reasons behind their persistence or loss across different areas of the Mediterranean Sea. Findings of this study improved knowledge about the main drivers of changes in distribution patterns of macroalgal forests, consolidating results from Chapter 1 (Part I).

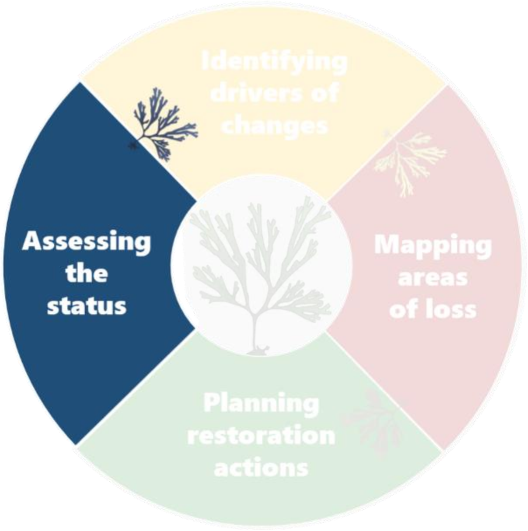
III.3 Mapping areas of loss (Chapter 3)

In Chapter 3 the study entitled *Can we preserve or restore overlooked macroalgal forests,* published on Science of the Total Environment journal in 2022 (Tamburello, et al., 2022), is presented. This study, zooming on the coasts of Italy, aims at i) documenting the historical and recent occurrence of *Cystoseira s.l.* species reported in the literature; ii) assessing the current knowledge on the extent of decline and extinction of *Cystoseira s.l.* populations and on the drivers responsible for these changes; iii) exploring whether the present network of protected areas is efficiently contributing to the conservation of brown macroalgal forests. This study refined the information about forests distribution for the Italian coastline, capitalizing and improving knowledge gathered in Chapter 1, developing a huge dataset mapping both intertidal and subtidal *Cystoseira s.l.* occurrence, both in the past and in the present. Maps produced in this study allowed the identification of areas of loss. Coupling this information with those obtained in Chapter 2 was crucial to inform the planning process developed in Chapter 4.

III.4 Planning restoration actions (Chapter 4)

Chapter 4 is represented by the manuscript entitled *The challenge of setting restoration targets for macroalgal forests under climate changes* submitted on the Journal of Environmental

Management (Fabbrizzi et al., 2023). The aims of this study were to assess the consequences of adopting different restoration targets and criteria at a Mediterranean scale on spatial restoration prioritization. Spatial priorities were identified by the software Marxan (the most widely used open-source decision-support tool in conservation, Watts et al. 2017) under six planning scenarios considering three main restoration targets, conceived to reflect the objectives of the EU Biodiversity Strategy for 2030. Fine-scale data about their present and past distribution across the Mediterranean Sea (integrating information from Chapter 3) were combined with data about their environmental requirements gathered using the Habitat Suitability Model (HSM) outputs provided in Chapter 1 (Part I) and validated in Chapter 2. In addition, to take into account future climate changes the distribution of Sea Surface Thermal Anomalies (SSTA) hotspots across the Mediterranean Sea was included in the spatial planning. The spatial planning conceived provided guiding principles and criteria to find suitable and priority areas for macroalgal forests restoration supporting the enhancement and the upscaling of future interventions



CHAPTER 1

Part I

MODELING MACROALGAL FOREST DISTRIBUTION AT MEDITERRANEAN SCALE: PRESENT STATUS, DRIVERS OF CHANGES AND INSIGHT FOR CONSERVATION AND MANAGEMENT

1.I.1 Abstract

Macroalgal forests are one of the most productive and valuable marine ecosystems, but yet strongly exposed to fragmentation and loss. Detailed large-scale information on their distribution is largely lacking, hindering conservation initiatives. In this study, a systematic effort to combine spatial data on *Cystoseira* C. Agardh canopies (Fucales, Phaeophyta) was carried out to develop a Habitat Suitability Model (HSM) at Mediterranean scale, providing critical tools to improve site prioritization for their management, restoration and protection. A georeferenced database on the occurrence of 20 *Cystoseira* species was produced collecting all the available information from published and grey literature, web data portals and co-authors personal data. Data were associated to 55 predictor variable layers in the (ASCII) raster format and were used in order to develop the HSM by means of a Random Forest, a very effective Machine Learning technique. Knowledge about the distribution of *Cystoseira* canopies was available for about the 14% of the Mediterranean coastline. Absence data were available only for the 2% of the basin. Despite these gaps, our HSM showed high accuracy levels in reproducing *Cystoseira* distribution so that the first continuous maps of the habitat across the entire basin was produced. Misclassification errors mainly occurred in the eastern and southern part of the basin, where large gaps of knowledge emerged. The most relevant drivers were the geomorphological ones, followed by anthropogenic variables proxies of pollution and urbanization. Our model shows the importance of data sharing to combine a large number of spatial and environmental data, allowing to individuate areas with high probability of *Cystoseira* occurrence as suitable for its presence. This approach encourages the use of this modeling tool for the prediction of *Cystoseira* distribution and for supporting and planning conservation and management initiatives. The step forward is to refine the spatial information of presence-absence data about *Cystoseira* canopies and of environmental predictors in order to address species-specific assessments.

Keywords: *Cystoseira* canopies, Habitat Suitability Model, Mediterranean Sea, Random Forest, species distribution

1.I.2 Introduction

Increasing human pressures such as coastal development, habitat destruction, pollution, maritime traffic, fisheries and illegal fishing together with climate change are strongly affecting the distribution of marine coastal species and habitats (Claudet and Fraschetti, 2010; Coll et al., 2010). New and emerging uses of marine resources (e.g., seabed mining, aquaculture) are also expected as additional sources of disturbance for marine coastal ecosystems (Wolff et al., 2018). This changing scenario calls for science-based information to understand the processes driving present trajectories of ecological change. In this framework, a solid knowledge of the distribution of species and their habitats over large spatial and temporal scales is critical to support all stages of marine spatial planning, to inform action prioritizations for scientists and decision-makers and to provide guidance for sustainable exploitation of marine resources, minimizing the negative impacts of present and future human activities (Douvere, 2010; Levin et al., 2014; Martin et al., 2014).

Recently, increasing efforts to carry out systematic collections of spatial data have been conducted from different areas of the world with significant progresses for a variety of species and habitats (Duarte, 2002; Orth et al., 2006; Waycott et al., 2009; Van Soest et al., 2012; Yesson et al., 2017). In Europe, a network of organizations is currently working together to integrate and share information to combine available marine data across EU countries (European Marine Observation and Data Network, EMODnet[4]). In this respect, the Mediterranean Sea should be a hub of information: it has been intensely studied since the last century and several initiatives have been carried out to document and combine the available knowledge about the occurrence and status of key target species/habitats such as coralligenous outcrops and maërl beds (Martin et al., 2014), *Posidonia oceanica* meadows (Telesca et al., 2015), coral assemblages (e.g., *Cladocora caespitosa*) (Chefaoui et al., 2017), sea pens (Bastari et al., 2018), and gorgonian species (Boavida et al., 2016; Ghanem et al., 2018). Despite these efforts, Gubbay et al. (2016) document a substantial lack of quantitative information on definition, distribution and temporal trends of the status of most of Mediterranean habitats: data collected has often limited spatial and temporal scope, scattered over different institutions in small datasets for specific species groups or habitats (Portman et al., 2013), with important gaps

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[4] http://www.emodnet.eu/

in different levels of information, such as the evaluation of conservation status (Coll et al., 2012, 2010).

Real data very often do not cover large spatial extensions so that modeling approaches and extensive extrapolations are needed to fill gaps in knowledge (Martin et al., 2014). Different tools have been used to address a deeper understanding of large-scale distribution of species and to fill gaps in actual knowledge, in relation to the type and resolution of available data (presence-only data, presence-absence data and data on predictive environmental variables) (Guisan and Zimmermann, 2000). Maximum entropy (Maxent algorithm), Random Forest (RF), generalized linear and additive models (GLMs and GAMs) are some of the modeling techniques used in these studies to develop predictive occurrence maps for target species/habitats (hereafter referred to as Habitat Suitability Models, HSMs).

All these examples, and others not listed here, demonstrate the wide range of applications for which systematic conservation planning can be applied at different scales, based on data from field surveys, expert knowledge and model-based estimations of species distribution (Guisan and Zimmermann, 2000; Levin et al., 2014). Despite limitations and associated uncertainties, HSMs can be a cost-effective approach to integrate real data, as they can help documenting where sensitive marine species and habitats are expected to occur (McArthur et al., 2010; Gorman et al., 2013; Martin et al., 2014), and predicting their possible shifts in distribution under global climate change (Guisan and Wilfried, 2005; Rodríguez et al., 2007; Martínez et al., 2018).

Macroalgal forests represent a paradigmatic example of key threatened benthic habitat featured by sparse but increasing spatial information, deserving further efforts to improve the management of those pressures determining their increasing loss across the Mediterranean and elsewhere (Benedetti-Cecchi et al., 2001). Macroalgal forests are one of the most productive and valuable, yet undervalued habitats, undergoing dramatic changes (Mangialajo et al., 2008; Thibaut et al., 2005, 2015; Blanfuné et al., 2016; Mancuso et al., 2018). Along temperate rocky coasts worldwide, large canopy-forming algae (Laminariales and Fucales) were dominant in both intertidal and subtidal habitats, providing shelter, food and nursery areas to a multitude of marine communities, increasing three-dimensional complexity and spatial heterogeneity of rocky substrates, enhancing biodiversity and productivity in coastal ecosystems (Ballesteros et al., 1998; Fowler-Walker and Connell, 2002; Steneck et al., 2002; Cheminée et al., 2013; Gianni et al., 2013; Gorman et al., 2013; Piazzi et al., 2018). However, the cumulative impacts of local anthropogenic pressures combined with other global stressors are driving the decline of brown algae and their associated communities in several regions of the world subject to increased human threats (Airoldi and Beck, 2007; Lamela-Silvarrey et al., 2012; Sala et al., 2012; Gianni et al., 2013; Strain et al., 2014; Mineur et al., 2015; Thibaut et al., 2015; Bianchi et al., 2018).

In the Mediterranean Sea, the genus *Cystoseira* C. Agardh (Fucales, Phaeophyta) is one of the most representative of the Sargassaceae family and includes a total of 45 taxa (Guiry and Guiry, 2010) with habitat-forming species dominating several assemblages from the very shallow to deep waters (−50 m) (Feldmann, 1937; Giaccone and Bruni, 1973; Ballesteros et al., 1998; ESRI, 2012; García-Fernández and Bárbara, 2016). Reductions in their cover and biomass, prompted by the integration of multiple natural and artificial effects, promote their replacement by turf-forming, filamentous or other ephemeral seaweeds (Murray and Littler, 1978; Benedetti-Cecchi et al., 2001, 2015; Thibaut et al., 2005; Ballesteros et al., 2007; Pinedo et al., 2007; Airoldi et al., 2008; Connell et al., 2008; Falace et al., 2010; Perkol-Finkel and Airoldi, 2010; Fraschetti et al., 2011; Giakoumi et al., 2012). Moreover, macroalgal forests can be overgrazed to barrens by urchins or fish (Fraschetti et al., 2001; Sala et al., 2011; Guarnieri et al., 2014; Vergés et al., 2014). Due to their role in supporting biodiversity and food webs, the loss of these long-lived brown algae is leading to a decrease in critical ecosystem services such as fisheries (Thibaut et al., 2015; Buonomo et al., 2018) and also to a reduction in the potential to sequestrate carbon dioxide and to mitigate climate change.

There is an increasing attention toward the state of macroalgal forests from both a conservation (Annex II of the Barcelona Convention, COM/2009/0585/FIN) and a restoration point of view (MERCES project, 2016[5]; AFRIMED project, 2018[6]), to better understand the potential of reverting present trajectories of change through active restoration. Furthermore, *Cystoseira* assemblages are being considered as habitats of critical importance for the EU (Directive 92/43/EEC; Annex I, included in “Rocky reefs”) and as indicators to assess ecological status in the context of the Water Framework Directive (WFD; Directive 2000/60/EC). Despite the

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[5] http://www.merces-project.eu/

[6] https://ec.europa.eu/easme/en/afrimed-algal-forest-restoration-mediterranean-sea

increasing interest, there are still many information gaps in the spatial distribution of *Cystoseira* canopies across the Mediterranean Sea and the drivers involved in the observed shifts. Understanding causal relationships and filling these gaps of knowledge is a crucial step to reverse current patterns of regression.

The aims of this study are (1) to synthesize knowledge about the distribution of *Cystoseira* species and (2) to develop a HSM for *Cystoseira* species living in the shallow rocky substrates through the Random Forest technique (RF) (Breiman, 2001), considering areas where both presence and absence data were available at Mediterranean scale. RF is a Machine Learning technique which, starting from a set of selected predictor variables and combining an ensemble of classification trees, is able to identify suitable and unsuitable areas for holding *Cystoseira* species. The developed HSM has been used to identify environmental predictors which are related to *Cystoseira* spatial distribution and to predict the occurrence of *Cystoseira* canopies at locations where information was not available. Our model can be regarded as a valuable tool for the assessment of species distribution on shallow rocky shores to guide their management, conservation and active restoration.

1.I.3 Materials and Methods

1.I.3.1 Georeferenced data for *Cystoseira* species

A systematic review was conducted consisting of three steps: (1) articles identification using two databases [ISI Web of Science (WOS) and Scopus] searched for the 1985–2018 time frame (cut-off date 31 March 2018), (2) abstract screening, and (3) review of pertinent articles. The aim of this activity was to collect all the information about the georeferenced occurrence of the genus *Cystoseira* at Mediterranean scale. The systematic literature screening was carried out by searching in the "Title," "Abstract," and "Keyword" fields using Web of Science Core Collection. The following combination of terms was included in the search: ("*Cystoseira*" OR "*Cystoseira* canopies" OR "Fucales" OR "brown algae" OR "macroalgal forest*" OR "habitat form*") AND ("distribution" OR "occurrence" OR "shift" OR "habitat loss" OR "decline") AND "Mediterranean Sea". We also searched the citation lists of selected articles, using the same search terms. Supplementary Figures S1.I.1, S1.I.2 show respectively the number of publications per year and the number of publications per country obtained from the literature screening. Unpublished information, gray literature and maps have been also searched and cataloged (Supplementary Table S1.I.1).

The georeferenced database with the spatial distribution of *Cystoseira* across the Mediterranean Sea, initially produced in the framework of the FP7 EU project CoCoNet (FP7, Grant agreement no: 287844), was also used. In addition, co-authors of this manuscript personally contributed with their data (Supplementary Table S1.I.2). New data were also acquired from the monitoring program CARLIT (CARtography of LITtoral and upper-sublittoral benthic communities) (Ballesteros et al., 2007). The EMODnet biology data portal (Guardia, 2018[7]) which contains a dataset on the distribution of *Cystoseira* across the Mediterranean Sea, the Black Sea and the Eastern Atlantic Ocean was also used.

Most collected data were only available in .jpeg or .pdf format, or only as a description in a text. Therefore, this information was digitized as shapefile points or polylines in order to be associated with a map, using the Open Source QGIS software (QGIS Development Team, 2018. QGIS Geographic Information System. O3.Open Source Geospatial Foundation[8]). The two resulting vector shapefiles of points and polylines showed the georeferenced distribution of *Cystoseira* species along Mediterranean coasts. Each data entry was accompanied by the geographic coordinates, data origins and data providers, sampling method, date and depth, publication date and which species of the genus were sampled, when this information was available.

Absence records, which were only available for a limited number of locations, were assembled in a line shapefile mainly generated from maps found in the collected articles, but also from expert opinion.

In order to develop a HSM for *Cystoseira* canopies across the Mediterranean Sea, we firstly extracted from the whole dataset only those records documenting the presence of species in the shallow rocky substrates, excluding those corresponding to species that are only found at deeper stands: *C. algeriensis* Feldmann, *C. corniculata* (Turner) Zanardini, *C. crinitophylla* Ercegovic,

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[7] http://www.emodnet-biology.eu
[8] https://www.qgis.org/it/site/

C. dubia Valiante, *C. montagnei* J. Agardh, *C. sauvageauana* Hamel, *C. schiffneri* Hamel, *C. sedoides* (Desfontaines) C. Agardh, *C. squarrosa* De Notaris, *C. tamariscifolia* (Hudson) Papenfuss, *C. zosteroides* (Turner) C. Agardh. Occurrence data of *C. amentacea* (C. Agardh) Bory, *C. barbata* (Stackhouse) C. Agardh, *C. brachycarpa* J. Agardh, *C. compressa* (Esper) Gerloff & Nizamuddin, *C. crinita* Duby, *C. elegans* Sauvageau, *C. foeniculacea* (Linnaeus) Greville, *C. humilis* Schousboe ex Kützing and *C. mediterranea* Sauvageau were included in the model subset, together with all records collected as *Cystoseira* spp. produced by sampling surveys in which the CARLIT method was applied, since it is focused on the identification of shallow rocky substrates (Ballesteros et al., 2007).

These data, combined with the available absence records, were assembled in a single vector shapefile. Then the vector layer was converted to the (ASCII) raster format, using the same procedure, grid resolution, geographical extent and coordinate system as the layers of the predictor variables which were selected as input to the model (see next section). The resulting *Cystoseira* raster layer was featured by values covering the entire Mediterranean coastline and was composed of three types of pixels: “absence” pixels (coded as 1), “presence” pixels (coded as 2) and “no-data” pixels (coded as 3), with the last one corresponding to all the sections of Mediterranean shoreline where no information was available.

1.I.3.2 Modeling *Cystoseira* occurrence: selection of predictor variables

A set of 55 predictor variables was associated with the dataset with the occurrence of the Cystoseira spp. as input to the modeling procedure. Most of these predictor variables derived from Halpern et al. (2008) and Wolff et al. (2018). Some have been obtained from data geoportals, e.g., the Copernicus Marine Environment Monitoring Service[9], and others were instead based on GIS calculations (the complete list of predictor variables, together with their sources, is shown in the Table 1.I.1).

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[9] http://marine.copernicus.eu

Table 1.I.1 Predictor variables used for the HSM with their units, source and the abbreviated name

	Name	Source	Abbreviated name
1	Artisanal fishing	Halpern et al., 2008	artfish
2	Aspect of the seafloor	GIS calculation based on variable 3 (bathymetry)	gebasp
3	Bathymetry	EMODNet Hydrography Portal, 2013	gebmed
4	Bottom salinity	Boyer et al., 2005	botsalin
5	Bottom temperature	Boyer et al., 2005	bottemp
6	Bottom type	Halpern et al., 2008	bottype
7	Calcite concentration	Tyberghein et al., 2012, based on Feldman and McClain, 2010	calcite
8	Chlorophyll *a* concentration (mean)	Tyberghein et al., 2012, based on Feldman and McClain, 2010	chmean
9	Chlorophyll *a* concentration (annual range)	Tyberghein et al., 2012, based on Feldman and McClain, 2010	chrange
10	Climate change (sea surface temperature)	Halpern et al., 2008, based on Casey et al., 2010	climsst
11	Climate change (UV)	Halpern et al., 2008, based on McPeters et al., 1998	climuv
12	Diffuse attenuation coefficient	Tyberghein et al., 2012, based on Feldman and McClain, 2010	dacmean
13	Dissolved oxygen concentration	Tyberghein et al., 2012, based on Boyer et al., 2009	dissox
14	Distance to 200 m isobath	GIS calculation based on variable 3 (bathymetry)	dist200m
15	Distance to coast	GIS calculation based on Wessel and Smith, 1996	distcoast
16	Distance to ports	GIS calculation based on NG-IA, 2011	distport
17	Distance to river mouths	GIS calculation based on ESRI, 2012, and Wessel and Smith, 1996	disriver
18	Euphotic depth	Feldman and McClain, 2010	zeumean
19	Human impact to marine ecosystems	Halpern et al., 2008	impact
20	Nitrate concentration	Tyberghein et al., 2012, based on Boyer et al., 2009	nitrate
21	Nutrient input (fertilizers)	Halpern et al., 2008	nutrient
22	Ocean acidification	Halpern et al., 2008	oceacidif
23	pH	Tyberghein et al., 2012, based on Boyer et al., 2009	ph
24	Photosynthetically available radiation	Tyberghein et al., 2012, based on Feldman and McClain, 2010	parmean
25	Phosphate concentration	Tyberghein et al., 2012, based on Boyer et al., 2009	phosphate
26	Pollutants (inorganic)	Halpern et al., 2008	inorpol
27	Pollutants (organic)	Halpern et al., 2008	orgpol
28	Pollution (ocean-based)	Halpern et al., 2008	pollution
29	Population pressure	Halpern et al., 2008	popress
30	Salinity	Tyberghein et al., 2012, based on Boyer et al., 2009	salinity
31	Topographic coastal slope (in degrees, 30 arc-seconds resolution)	GEBCO	cst
32	Sea surface temperature (mean)	Tyberghein et al., 2012, based on Feldman and McClain, 2010	sstmean
33	Sea surface temperature (annual range)	Tyberghein et al., 2012, based on Feldman and McClain, 2010	sstrange
34	Shipping intensity	Halpern et al., 2008	shipping
35	Silicate concentration	Tyberghein et al., 2012, based on Boyer et al., 2009	silicate
36	Slope of the seafloor	GIS calculation based on variable 3 (bathymetry)	gebslo

37	Urban area	Google Earth imagery and location-tagged photographs from the web-service Panoramio; MODIS 500-m global map of urban extent dataset	urban
38	Coast material classes	Google Earth imagery and location-tagged photographs from the web-service Panoramio	coast_material
39	Mean wave height (in cm)	Dataset produced as part of the RISES-AM projected by CMCC (Euro-Mediterranean Center on Climate Change)	waves
40	Max High Water (in m)	Pickering, 2014	Tide_maxHW
41	High Water (in m)	Pickering, 2014	Tide_MHW
42	Minimum Low Water (in m)	Pickering, 2014	Tide_minLW
43	Low Water (in m)	Pickering, 2014	Tide_MLW
44	Mean Tidal Range (in m)	Pickering, 2014	Tide_MTR
45	Saltmarshes (in m)	UNEP-WCMC saltmarshes	saltmarshes
46	Tourist arrival for 2014	World Bank	tour_arr
47	Height above mean sea level in 10 years (in m)	GTSR dataset	GTSR_10
48	Height above mean sea level in 100 years (in m)	GTSR dataset	GTSR_100
49	River	Derived from google earth	river
50	Forest area (in km2)	European Space Agency and Université Catholique De Louvain (UCL). Global Land Cover Map for 2009	forestarea
51	Arable area (in km2)	European Space Agency and Université Catholique De Louvain (UCL). Global Land Cover Map for 2009	ArableArea
52	Open space (in km2)	European Space Agency and Université Catholique De Louvain (UCL). Global Land Cover Map for 2009	openarea
53	Urban area (in km2)	European Space Agency and Université Catholique De Louvain (UCL). Global Land Cover Map for 2009	urbanarea
54	Vertical Land Movement (in mm/yr)	Peltier et al., 2014	VerticalMovement
55	Max stress of wind from 2008 to 2017 (in Pa)	Copernicus Marine Environment Monitoring Service	maxWind

Predictor layers were selected on the basis of the coverage of the information they provided. We only used variables with data coverage over the entire Mediterranean basin and with the highest available resolution. Following this criterion, we collected values for environmental variables (e.g., wind stress and the waves energy along the coastline) and anthropogenic variables (e.g., pollution, population density and shipping intensity). All the 55 selected layers were converted to a common raster format, having as geographical extent the Mediterranean Sea, in WGS 84 coordinate system and with a resolution of 0.004166 decimal degrees (i.e., each pixel was about 460 m along the latitudinal axis, and from about 330 m to 380 m along the longitudinal one, depending on latitude). The rasterization process was carried out using the package “raster” (Hijmans, 2017. R package version 2.6–7[10]) in the R open source data analysis

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[10] https://CRAN.R-project.org/package=raster

software (R Core Team, 2016[11]). The same raster format was applied to *Cystoseira* spp. occurrence data to ensure the appropriate matching with the predictor layers.

Furthermore, since most layers coming from the above sources did not provide data for the shoreline, where *Cystoseira* species potentially occur, we developed a procedure to estimate the value of the pixels of the shoreline. We considered the value of the first non-empty pixel of the layer for qualitative predictors and the mean value of neighbor pixels for quantitative predictors, within a search radius of 10 pixels at most. In particular, the mean value was obtained on the Moore neighborhood or, otherwise, on the smallest frame with at least a non-empty pixel. Thereby, all the predictor variables provided data for each pixel over the Mediterranean shoreline where *Cystoseira* records were distributed (presence, absence, and no-data).

1.I.3.3 Modeling approach

From the *Cystoseira* raster layer we extracted a subset including only regions where both presence and absence data were available. Hence, the distribution of *Cystoseira* canopies across the Mediterranean Sea was modeled focusing on a dataset composed by 8,143 pixels: 5,475 “presence” pixels and 2,668 “absence” pixels. The Machine Learning (ML) method selected to model the distribution of *Cystoseira* canopies was the RF (Breiman, 2001). This technique, based on a set of classification trees, needs two subsets of data: a training set to tune the model and a test set to validate model performances. We split our dataset in a training and a test set (with the 20% of the dataset assigned to the test set and the remaining 80% to the training set) following the next steps: at first, we superimposed a grid of 0.50 decimal degrees square cells to the whole Mediterranean basin. Then, we randomly assigned cells to the test set excluding those containing less than 100 records (i.e., 100 pixels of the *Cystoseira* raster layer). This number corresponded to the first quartile of the distribution of records among all the cells. Only cells where the presence/absence records were not too unbalanced (no more than 80% of presence records) were selected in order to better reflect the overall characteristics of the dataset. Moreover, care was taken to avoid geographical segregation in assigning cells, so as to be representative, as much as possible, of multiple environmental conditions occurring across

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[11] https://www.R-project.org/

Mediterranean regions. This procedure minimized the influence of spatial autocorrelation in the test procedure, as only at the boundaries of the test set cells a negligible number of pixels closely resembling those in the training set cells could be found.

The RF predicted the class (i.e., presence or absence) of each record in the training and the test set taking the majority voting (the 50%, at least) of the overall trees which composed the forest. We trained several RFs through a Fortran 90 program obtained from the original Fortran 77 source code by Leo Breiman and Adele Cutler[12]. The only changes to the original code were in the input of training parameters from file and in the dynamic allocation of the arrays, while the algorithm and the output file format were not modified.

RFs, by combining multiple classification trees in a single output, require the tuning of different parameters which affect the forest growth with repercussions on model accuracy: the number of trees in the forest and the number of cases in the terminal leaves of the trees, which have to be tuned in order to minimize the generalization error and to avoid overfitting; the number of predictor variables randomly selected at each split, which, remaining constant during the forest growth, shows large effects on the strength of each individual tree and on the correlation between any pair of them (Breiman, 2001). RFs were grown using 250 and 500 trees in the forest, a number of predictor variables per split at each node of the trees ranging between 4 and 14 and six different number of cases in the terminal leaves of the forests (1, 10, 25, 50, 100, and 150).

Since our dataset was unbalanced in the number of presence and absence records, the cut-off value (t = 0.50) was optimized by analyzing the Receiver Operating Characteristic (ROC) curve (Zweig and Campbell, 1993), in order to find the best compromise between the predictions of true and false positives, i.e., between sensitivity and specificity of the model. Then, we calculated the Kappa statistics (Cohen, 1960) to identify the best model among those we trained, considering also the Area Under the ROC curve (AUC) as a measure of the overall accuracy of the models. Furthermore, measuring the Kappa statistics over the test set (built to be as independent as possible from the training set) implies evaluating the model robustness just as a null model does with parametric modeling approaches. Indeed, this coefficient measures the

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[12] https://www.stat.berkeley.edu/~breiman/RandomForests/cc_software.htm

agreement among raters (or between model output and observed data) by assessing the deviation from random agreement (McHugh, 2012). Kappa statistics and ROC curve were calculated using the “caret” package (Kuhn, 2017. R package version 6.0–78[13]) and “pROC” package (Robin et al., 2011. BMC Bioinformatics, 12, p. 77. doi: 10.1186/1471-2105-12- 77[14]) in the R environment.

The importance of predictor variables was assessed by comparing the permutation importance measure (or the Mean Decrease Accuracy, MDA) (Breiman, 2001) with the Gini importance measure (or the Mean Decrease Impurity, MDI) (Breiman, 2003), both calculated at the end of the training phase. In the first instance, cases out of the bootstrap samples used in the training phase of the RF (OOB records), are randomly permuted in the values of predictor variables. The difference between the misclassification rate of original and permuted OOB values, divided by the standard error, is used as a measure of the importance of predictor variables, as the increase in this difference is proportional to variable importance. In the second case, every time a node is split according to a predictor variable, the Gini Importance for the two descendent nodes is less than the one for the parent nodes. An alternate measure of variable importance is so provided by the total decrease in node impurity for each variable averaged overall the trees in the RF. In any case, the two measures of variable importance were often consistent with each other.

We also validated the performance of the selected model by getting the predictions for regions where only presence records were available, i.e., records that were not used either in the training or in the test set for the RF. The performance of the models on these records was evaluated by looking at the total number of trees in the forest that voted for presence.

At the end, the model was used as a tool to predicting presence and absence of *Cystoseira* species for areas where no information was available (i.e., no-data records). In the Supplementary Figure S1.I.3, a flow diagram illustrating the main steps from data access to HSM and the analysis of the variables relative importance is presented.

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[13] https://CRAN.R-project.org/package=caret
[14] http://www.biomedcentral.com/1471-2105/12/77/

1.I.4 Results

1.I.4.1 Distribution and coverage of *Cystoseira* species across the Mediterranean Sea

The currently known distribution, obtained by combining line (presence: 12,968; absence: 564) and point (presence: 19,782) records, is shown in Figure 1.I.1, depicting areas where *Cystoseira* canopies are known to be either present or absent. Spatial coverage is geographically biased since most data derived from a specific monitoring activity within the EU Water Framework Directive on macroalgae (CARLIT index), carried out only on the western Mediterranean Sea. Types and number of collected records are specified for each of the 22 Mediterranean countries in the Supplementary Table S1.I.3. The table shows for which countries occurrence data were available and where information was reported at species level. In addition, both the presence and the absence shapefiles are available on request to the corresponding author, allowing the examination of finer details which were hard to represent in the map shown in Figure 1.I.1, aimed at giving an overview of the collected dataset.

Presence data, from both line and point records, covered 15 out of 22 Mediterranean countries. For only four out of these 22 countries absence data were also available (Albania, France, Italy, and Spain). *Cystoseira* canopies were found to be present along the Mediterranean coastline for 6,342.41 km out of a total coastal length of 46,000 km. *Cystoseira* absence, expressed as line records only, accounted for 1,328.27 km. No presence or absence data were available along the rest of the coasts of the basin. Obviously, the linear length estimates were only based on line records.

An exhaustive list of all the *Cystoseira* species included in the dataset, with their respective number of occurrences, is also provided in the Supplementary Table S1.I.4. The species with the highest number of available records are *C. amentacea*, *C. mediterranea*, and *C. compressa* which represent about the 40% of the dataset collectively, considering both point and line records. This information is mostly available for shallow rocky infralittoral habitat, where a lot of ecological studies and systematic monitoring have been conducted, depicting in this way a biased scenario of the presence of *Cystoseira* at basin scale. In addition, most studies did not provide information at species level and a total of 18,742 records were mapped at genus level as *Cystoseira* spp.

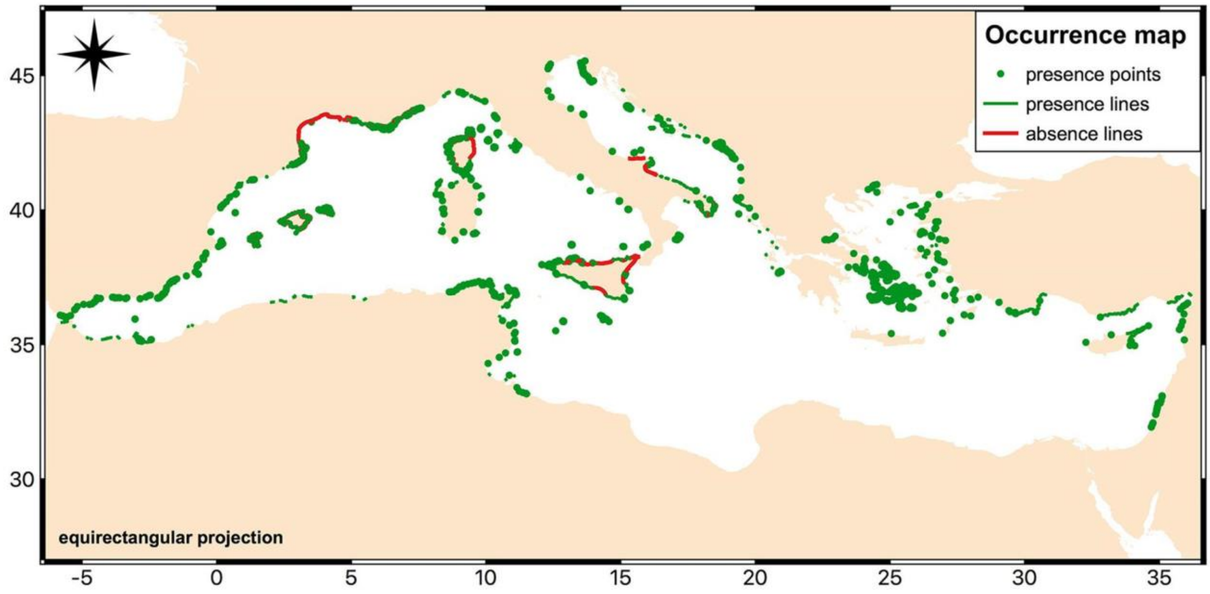


Figure 1.I.1 Distribution of *Cystoseira* canopies across the Mediterranean Sea. Map created with QGIS software (QGIS Development Team, 2018. QGIS Geographic Information System. O3.Open Source Geospatial Foundation: https://www.qgis.org/it/site/).

The *Cystoseira* raster layer, produced in order to develop the HSM for *Cystoseira* canopies based on a RF and obtained from the rasterization of the infralittoral species subset, was composed of 113,021 pixels stretched over all the Mediterranean coastline. Of these, 100,609 were coded as “no-data” pixels, 9,744 were “presence” pixels and the remaining 2,688 were “absence” pixels. The subset mainly included records of canopy-forming species (90% of all records) but also records of species which may not be “forest”-forming (e.g., *C. compressa*).

1.I.4.2 Habitat Suitability Model for *Cystoseira* canopies

To develop the HSM for *Cystoseira* canopies we used only data from areas where both presence and absence information were available. These areas included the Karaburun Peninsula and Sazani Island in Albania, Apulia and Sicily regions in Italy, Corsica and the southern coast of France, the eastern coast of the Iberian Peninsula with the Balearic archipelago in the western Mediterranean Sea. Figure 1.I.2 shows the aforementioned areas and, for reason of scaling, we illustrated the amount of the available data with the presence/absence ratio using cells of different dimensions and colors. Practically, from the “*Cystoseira*” raster layer we extracted a subset of data composed by 8,143 pixels: 5,475 “presence” pixels and 2,668 “absence” pixels. Then, we split this dataset in two further subsets, the training set and the test set, needed, respectively, to train and validate the RF model. The training set was composed of 6,531 records, of which 4,402 were reported as presence and 2,129 as absence. The remaining 1,612 records, divided into 9 separated Mediterranean areas, were assigned to the test set. Of these records, 1,073 were reported as presence and 539 as absence.

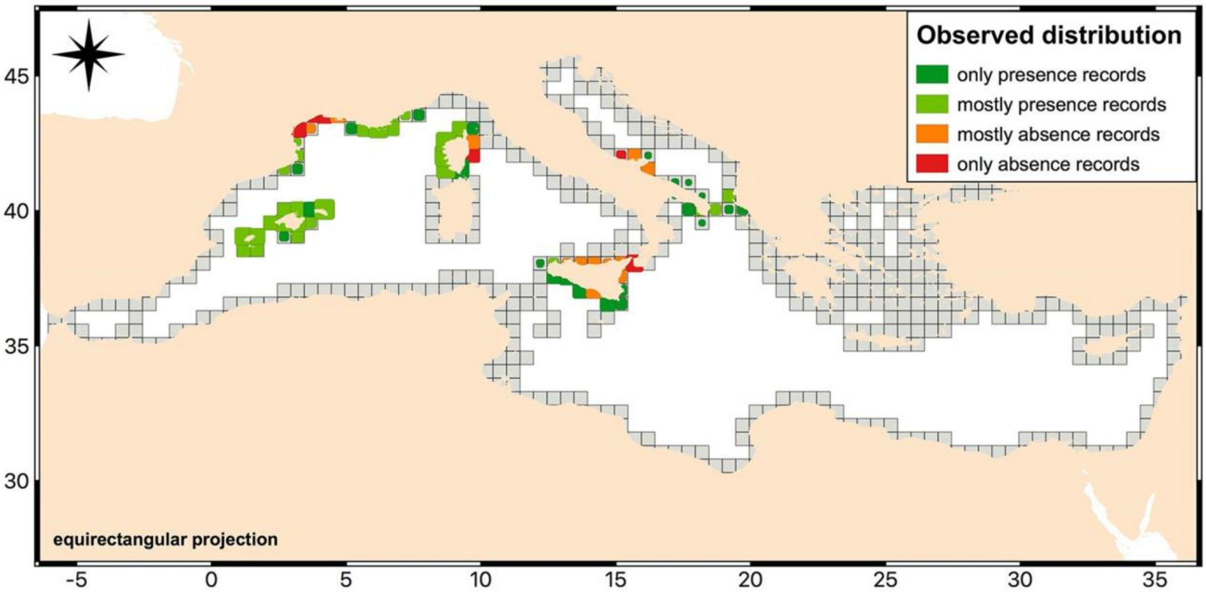


Figure 1.I.2 Observed distribution of records of *Cystoseira* infralittoral species used in order to develop the HSM. The map shows only the areas where both presence and absence information were available. The colors used reflect the ratio between presence and absence data per cells. The dimension of cells is representative of the amount of available data. Map created with QGIS software (QGIS Development Team, 2018. QGIS Geographic Information System. O3.Open Source Geospatial Foundation: https://www.qgis.org/it/site/).

The modeling process was trained with all the 55 predictor variable layers (Table 1.I.1), associated with presence/absence data, since a RF is able to select the most relevant predictor variables out of the whole set of those available. Moreover, its efficiency is not impaired by correlations between variables as the best split at the nodes of each tree is selected from a random subset of them. This property is one of the strengths of the RF technique, which selects only relevant variables even in presence of non-informative ones (Ishwaran, 2007; Louppe et al., 2013; Catucci and Scardi, 2020), thus becoming insensitive to the collinearity issues that might hinder other modeling methods.

A total of 132 RFs, tuned with different combination of parameters, was trained. Model elected as the best was obtained from the following combination of parameters: 500 trees in the forest, 25 cases in the terminal leaves of the forest and 9 predictor variables per split at each node of the trees. RF's outputs were analyzed by the computation of Kappa statistics, which gave an evaluation of the overall accuracy of the models. Kappa statistics of the selected model, calculated for the default cut-off value ($t = 0.50$), were 0.919 for the training set and 0.573 for the test set. Since our dataset was unbalanced in the number of presence and absence records, we optimized the cut-off value through the analysis of the ROC curve based on the test set, in order to overcome the bias that affected RF predictions toward presence (given that around the 70% of records in the training set were for presence). The optimized cut-off value was found to be 0.61 (Figure 1.I.3B). It did not affect the Kappa statistics for the training set, which remained quite unchanged (K = 0.917), but allowed an improvement for the test set (K = 0.637). These values resulted the largest in comparison with those obtained from any other RF we run and are indicative of a good model accuracy. Confusion matrices for the training and the test set calculated before and after the optimization of the cut-off value are shown in the Table 1.I.2. The optimized cut-off value improved predictions over false positives for the training set and, to a larger extent, for the test set. While the decrease in false positives was inevitably accompanied by an increase in false negatives, the overall accuracy of model predictions grew by adopting the optimal cut-off value.

Through the ROC curve analysis, we also examined the AUC value as indicator of model accuracy. As shown in Figure 1.I.3, the AUC was 0.988 for the training set (Figure 1.I.3A) and 0.875 for the test set (Figure 1.I.3B). These values, which were not affected by the cut-off value, confirmed our evaluation of model output since they resulted considerably large not only for the training, but also for the test set.

Table 1.I.2 Confusion matrices and Kappa statistics for the RF output before and after the optimization of the cut-off value.

Confusion matrix and Kappa statistic with *t* = 0.50							
Training set				**Test set**			
		Observed				**Observed**	
		Presence	Absence			Presence	Absence
Predicted	Presence	4358	44	**Predicted**	Presence	1025	48
	Absence	182	1947		Absence	242	297
Kappa = 0.919				Kappa = 0.560			

Confusion matrix and Kappa statistic with *t* = 0.61

Training set				**Test set**			
		Observed				**Observed**	
		Presence	Absence			Presence	Absence
Predicted	Presence	4301	101	**Predicted**	Presence	932	141
	Absence	142	1987		Absence	143	396
Kappa = 0.919				Kappa = 0.560			

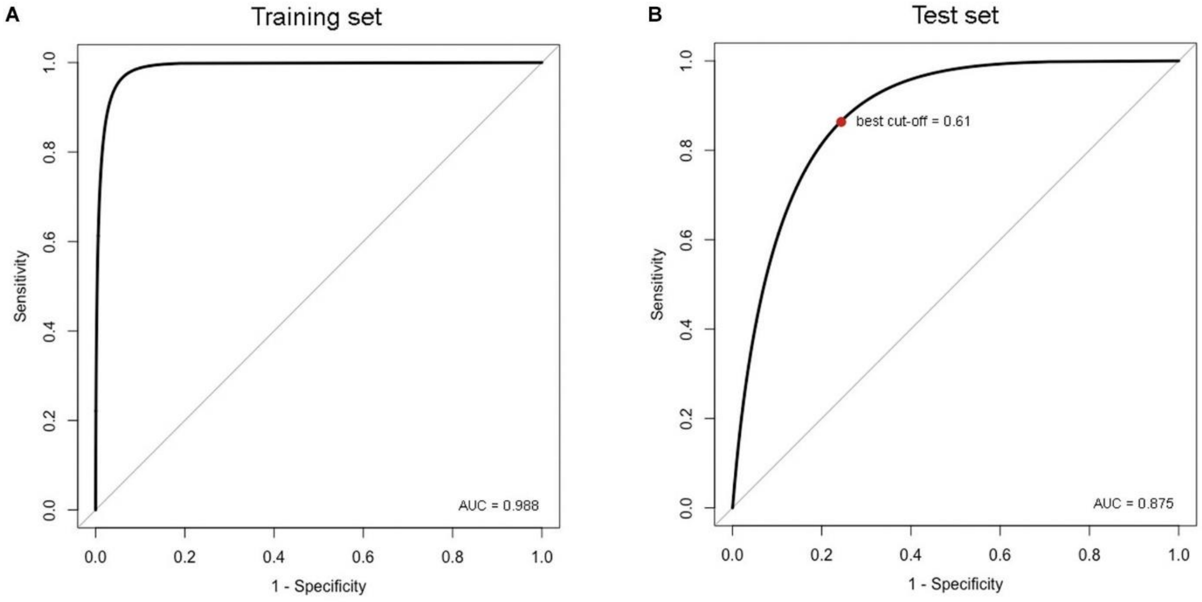


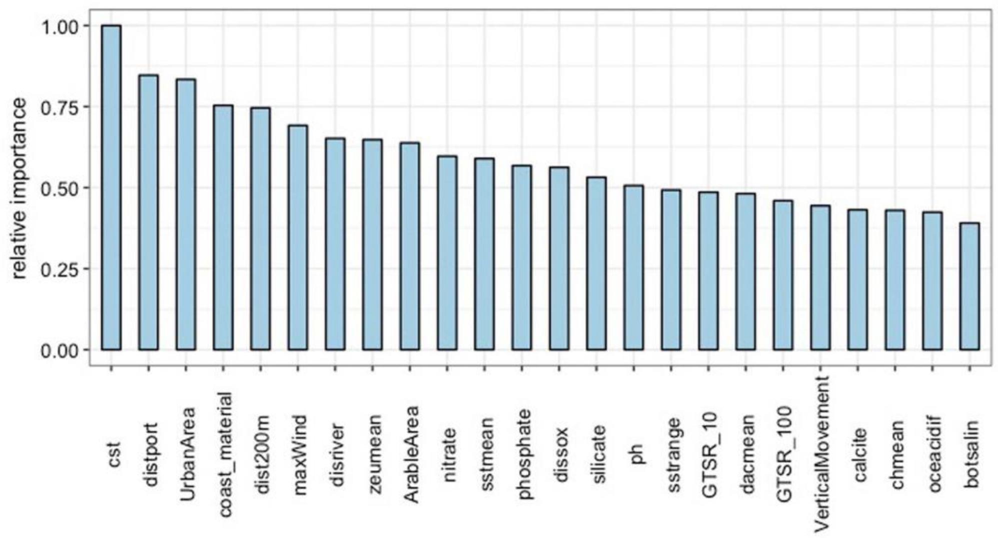
Figure 1.I.3 ROC curves of the best RF trained with the AUC for both the training (A) and the test set (B). The optimal cut-off, represented by the red dot in the Figure 2.3B, was calculated on the analysis of the ROC curve based on the test set.

1.I.4.3 The relative importance of predictor variables

Variable importance was assessed on the basis of the comparison between the permutation importance measure and the Gini importance measure, both calculated according to the original RF algorithm during the training procedure. From this analysis we identified a group of predictor variables which played a major role, although ranking differently, in increasing classification accuracy according to both the importance measures used (Figure 1.I.4). The most relevant predictor variables were the topographic coastal slope (“cst”) and coast materials (“coast_material”), which both can be regarded as an expression of coastline geomorphology.

Looking at the permutation importance measure (Figure 1.I.4A), the following factors linked to human pressures were identified among the variables mostly affecting the RF output: distance to ports (“distport”) and coastal development expressed as urban areas (“Urban Area”). Variables related to natural pressures, as the distance to the 200 m isobath (“dist200 m”), the max stress of the wind (“maxWind”) and the distance to river mouths (“distriver”), were also detected as important according to the permutation measure. On the other hand, the Gini importance measure (Figure 1.I.4B) highlighted that some seawater physical variables, for instance the euphotic depth (“zeu mean”), the diffuse attenuation coefficient (“dac mean”), the mean of chlorophyll *a* (“chmean”) and the nitrate concentration (“nitrate”), contributed significantly to the splits in the RF trees.

A Permutation importance measure



B Gini importance measure

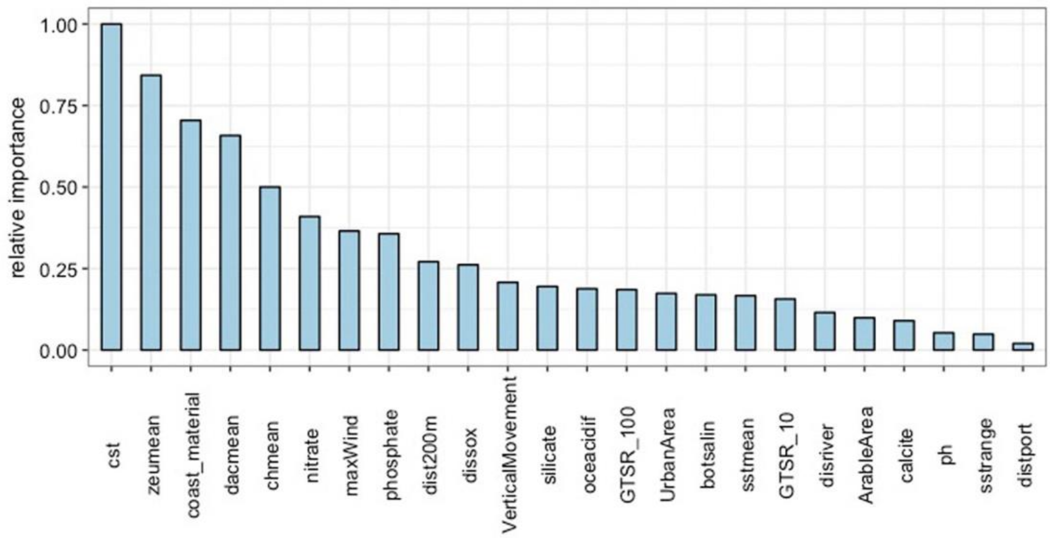


Figure 1.I.4 Most relevant variables according to importance values obtained from both the permutation (A) and the Gini importance measures (B). Name of the variables in alphabetical order: “ArableArea” = arable area (in km2); “botsalin” = bottom salinity; “calcite” = calcite concentration; “chmean” = chlorophyll a concentration (mean); “coast_material” = coast material classes; “cst” = topographic coastal slope (in degrees, 30 arc-seconds resolution); “dacmean” = diffuse attenuation coefficient; “disriver” = distance to river mouths; “dissox” = dissolved oxygen concentration; “dist200 m” = distance to 200 m isobath; “distport” = distance to ports; “GTSR_10” = height above mean sea level in 10 years (in m); “GTSR_100” = height above mean sea level in 100 years (in m); “maxWind” = max stress of wind from 2008 to 2017 (in Pa); “nitrate” = nitrate concentration; “oceacidif” = ocean acidification; “ph” = pH; “phosphate” = phosphate concentration; “silicate” = silicate concentration; “sstmean” = sea surface temperature (mean); “sstrange” = sea surface temperature (annual range); “UrbanArea” = urban area (in km2); “VerticalMovement” = vertical land movement (in mm/yr); “zeumean” = euphotic depth.

1.I.4.4 Model validation and purpose

While confusion matrices and statistics based on comparison between observed and modeled occurrences were evaluated on presence and absence records in the test set, we used all the presence records excluded from the training and test set to investigate model accuracy. These records corresponded to 4,269 “presence” pixels of the *Cystoseira* raster layer (Figure 1.I.5). For each record, the number of trees in the forest that voted for presence was analyzed. This number was then scaled into a (0,1) range and assumed, taking into account the optimized cut-off value, as estimates for *Cystoseira* presence to be analyzed in order to improve the assessment of the strength of model predictions. Thus, records associated with a RF output larger than 0.61 were considered correctly predicted, in accordance with the optimal cut-off value. We analyzed RF output values distribution by grouping presence records in nine Mediterranean regions aiming at assessing model accuracy on spatially independent areas (Figure 1.I.6). Distributions obtained from seven of these regions showed quite high probability of presence for most records, meaning that these areas were predicted to have suitable conditions for *Cystoseira* occurrence, consistently with the observed status. Sardinia resulted having the highest frequency of correctly predicted cases with the median RF output value around the 0.9. The west coast of north Italy, the eastern coast of Adriatic Sea, the Aegean Sea, the African coasts, the western and the southern areas of Mediterranean Sea followed immediately after Sardinia, with RF output concentrated between the 0.7 and the 0.8 classes. These areas were essentially predicted as suitable by the model, but with less confidence than the first one. For the remaining regions, i.e., the eastern Mediterranean regions and the northern Adriatic Sea, the number of cases in which trees voted for presence significantly decreased, pointing out that the model identifies these areas as almost unsuitable for holding *Cystoseira* species. Indeed, the median RF output value for the latter cases shifted between the 0.3 and the 0.6 (i.e., less than the optimal cut-off value).

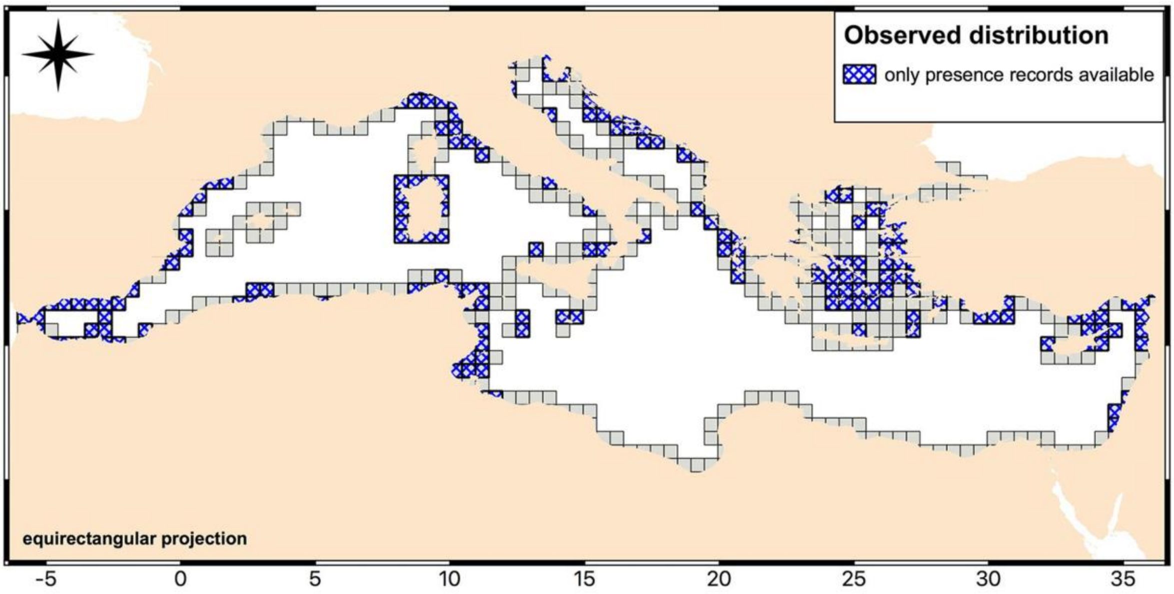


Figure 1.I.5 Observed distribution of *Cystoseira* records used in the model validation. The map shows the areas where only presence information was available. Map created with QGIS software (QGIS Development Team, 2018. QGIS Geographic Information System. O3.Open Source Geospatial Foundation: https://www.qgis.org/it/site/).

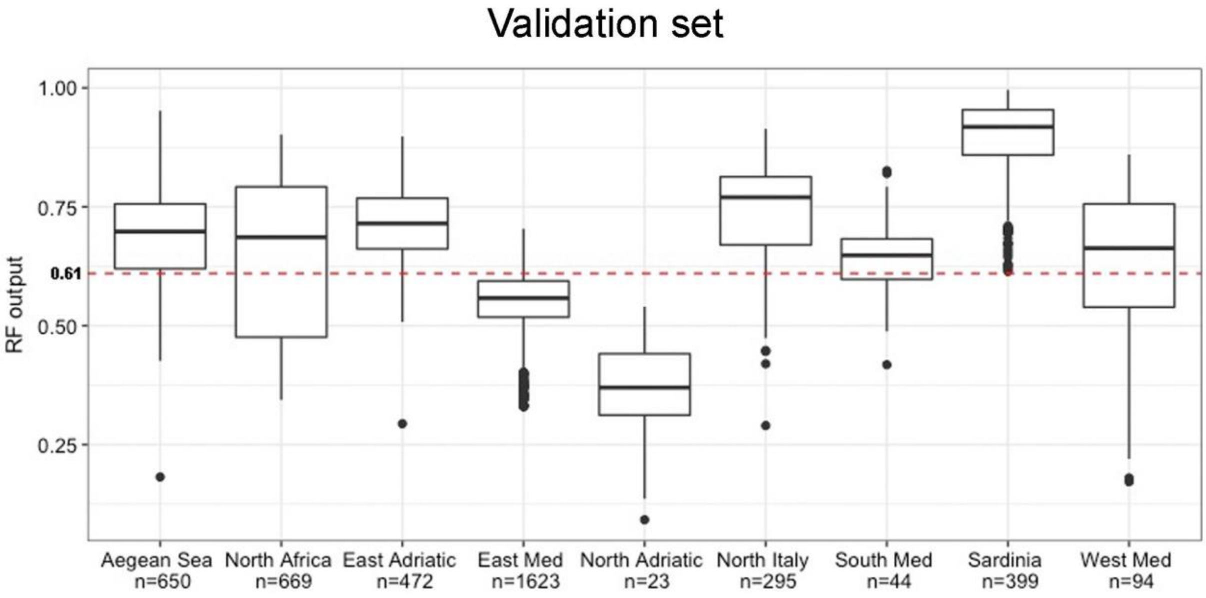


Figure 1.I.6 Distribution of the RF output values for the validation set, i.e., for the Mediterranean areas where only presence records were available. The red dashed line represents the optimal cut-off ($t = 0.61$).

Ultimately, developing a HSM for species of the *Cystoseira* genus living in the shallow rocky infralittoral habitat enabled also to carry out an exploratory analysis with the purpose of figuring out how *Cystoseira* forests are distributed along all those sites where no data were available across the Mediterranean Sea. Considering the spatial resolution of this study, we mapped 100,609 records for which no information occurred (i.e., “no-data” pixels of the *Cystoseira* raster layer). Using model predictions and proceeding as for “presence” pixels considered for model validation, we investigated, for each record, the number of trees in the RF that “voted” for presence, which is related to the predicted probability of presence. Our model classified 47,783 of these records as “absence” given that the RF output was lower than the optimized cut-off value and 52,826 as “presence” with an RF output larger than or equal to the cut-off value. To understand the distribution of *Cystoseira* occurrences across the basin, we firstly classified pixels on the basis of the abiotic properties of the coastline (i.e., “coast_material” variable), considering them as a limiting factor able to control canopies occurrence. Hence, we grouped records falling on “rock/unerodible” pixels, where canopies could potentially occur, and separated these records from those which covered “sand/mud” pixels, considered as unsuitable for *Cystoseira* growth. Figure 1.I.7A shows the distribution of the RF output values for the first group of pixels, which were regarded as possibly suitable for holding *Cystoseira* living in the shallow rocky infralittoral habitat species: the modal class of RF output values lies on the right of the optimal cut-off value, shown as a red dashed line in the figure. The left tail of the distribution represents all records predicted as “absence” in spite of potentially suitable geomorphological conditions. On the other side, Figure 1.I.7B shows the distribution of the RF output values for “sand/mud” pixels and reveals that these pixels, as expected, are mostly predicted as unsuitable, with the modal class of RF output on the left of the optimal cut-off value, in accordance with the unsuitable coast material. In this case, the right tail of the distribution shows records predicted as “presence” even though geomorphological conditions are unsuitable and could be regarded as model misclassifications, most probably related to the coarse resolution of our pixels (which might include small suitable areas), to the particular coast exposure or composition, or even to the lack of sufficient data in developing the HSM.

In the Figure 1.I.8 we presented an overview of our habitat suitability assessment. High resolution details of the model predictions are provided in the raster ASCII format as a zip file in the Supplementary Data.

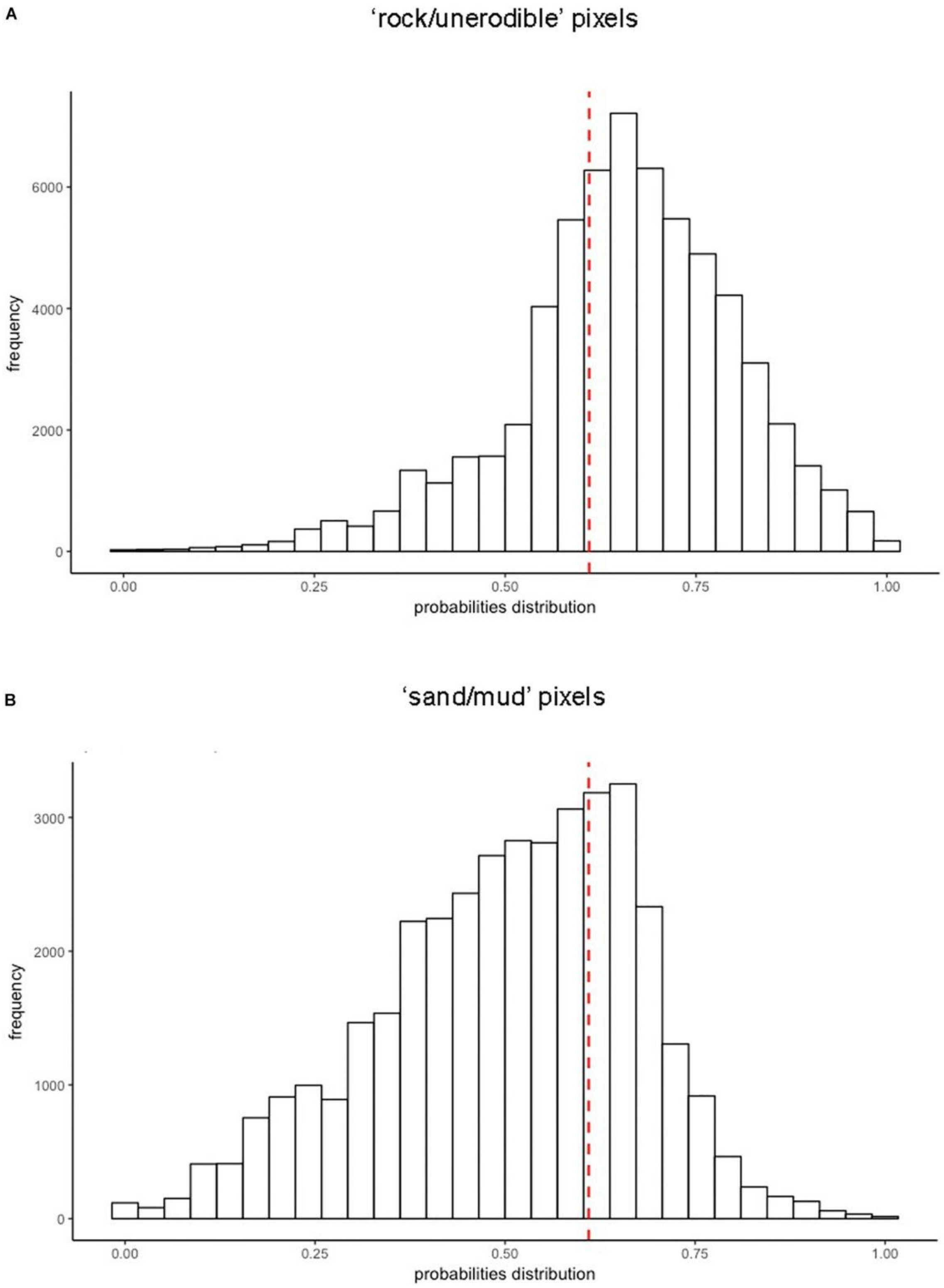


Figure 1.I.7 Distribution RF output values for "no-data" records, grouped on the basis of the type of coastline: (A) "rock/unerodible" pixels; (B) "sand/mud" pixels. Red dashed lines represent the optimal cut-off ($t = 0.61$).

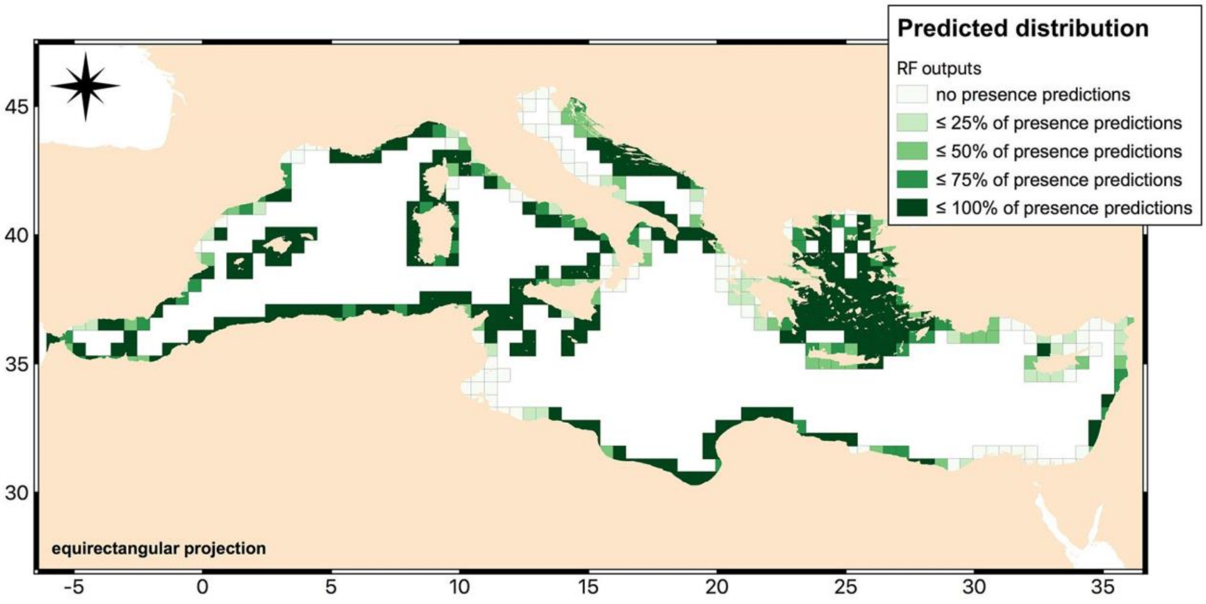


Figure 1.I.8 Map of the predicted distribution of *Cystoseira* infralittoral species according to RF outputs. The map is based on a combined dataset of canopy-forming species and records of species which may not be "forest"-forming (e.g., *C. compressa*). It has been created with QGIS software (QGIS Development Team, 2018. QGIS Geographic Information System. O3.Open Source Geospatial Foundation: https://www.qgis.org/it/site/).

1.I.5 Discussion

We produced the first effort able to deliver a georeferenced dataset of *Cystoseira* forests strongly affected by human pressures and deserving conservation priorities across the whole Mediterranean basin. This first crucial step enabled us to deepen our understanding on canopies distribution and to fill gaps in our knowledge by developing a HSM for a subset of *Cystoseira* species distributed on the shallow rocky shores.

To develop the HSM, we considered areas where both presence and absence records were available, using all other Mediterranean regions to validate and test model performances. Our model showed a quite high accuracy level in reproducing *Cystoseira* distribution, endorsed by a large number of occurrence and environmental data, but also enhanced by the identification of the species of the shallow rocky shores as a rather uniform ecological target to be modeled. On the other hand, the spatial resolution of our study, imposed by the available resolution for predictor variables, precluded a species-specific distribution assessment. Fine-scale data are indeed needed in order to improve the model in this sense (Cefalì et al., 2018). Our database suffers from all the limitations already described across the literature. Research efforts (published and unpublished) differ among countries (Coll et al., 2010), and large data gaps emerged in the eastern and southern part of the basin. Furthermore, heterogeneous sampling methods or false absences in occupancy surveys can lead to underestimation if the imperfect detection of the species is not accounted for Katsanevakis et al. (2017). These issues might have affected our spatial representation of *Cystoseira* canopies, resulting in an incorrect estimate of species distribution.

Despite the limits, the chosen approach based on the RF technique allowed us to highlight the most relevant predictor variables affecting the HSM and therefore those variables better candidates to explain *Cystoseira* canopies distribution and potential for regression. Considering both the importance measures obtained as RF outputs, the topographic coastal slope and the nature of substrate along the coast were identifies as the main factors in controlling *Cystoseira* canopies distribution, in accordance with the specific coastal conditions required for the infralittoral species development, limited to intertidal and shallow subtidal rocky shores (ESRI, 2012; Mancuso et al., 2018). Some anthropogenic variables emerged as relatively important from this analysis, but they followed the importance of the geomorphological ones. These variables (i.e., the distance to ports and to the urban areas), proxies of pollution and urbanization, have been claimed to drive the loss of *Cystoseira* forests in many Mediterranean regions in the last 20 years (Cormaci and Furnari, 1999; Benedetti-Cecchi et al., 2001; Thibaut et al., 2005; Mangialajo et al., 2008; Sales and Ballesteros, 2009; Perkol-Finkel and Airoldi, 2010). Finally, environmental variables, such as the distance to river mouths, the euphotic depth, the diffuse attenuation coefficient, the mean of chlorophyll *a* and the nitrate concentration, also displayed a significant role in the RF growth. Actually, these predictors can be regarded as indicators of nutrient enhancement, water turbidity and eutrophication levels which are included in several studies amongst the main causes for the regression of *Cystoseira* species (Cormaci and Furnari, 1999; Arévalo et al., 2007; Sales and Ballesteros, 2009; Fraschetti et al., 2011; Sala et al., 2012; Mancuso et al., 2018). Nevertheless, attention should be drawn to the fact that RF, given its potential to reflect the resemblance between the local spatial structure of predictor variables and species distribution, does not allow to reveal causal relationships. In our model, RF highlights that anthropogenic pressures, directly or indirectly, could have important roles in affecting the distribution of *Cystoseira* species, identifying several factors to be prioritized in conservation actions devoted to this genus. However, experimental studies are needed to identify the drivers for the observed canopies regression, as well as a shift from a single-threat approach toward a multiple-stressor one should be adopted in order to understand patterns of distribution and trajectories of change in *Cystoseira* forests (Benedetti-Cecchi et al., 2015; Mancuso et al., 2018). Notwithstanding, recently developed methods based on the hybridization of experimental and observational data provide novel opportunities to leverage the scope and causal inferential strength of large-scale studies (Benedetti-Cecchi et al., 2018). For example, these methods allow integrating empirical estimates of biological interactions into species distribution models, effectively increasing the predictive accuracy and ability to attribute causality of these models. This might be particularly critical for *Cystoseira* since another potentially strong but yet unexplored predictor for the presence of macroalgal forests is the emergence of highly effective grazers, invasive Indo Pacific rabbitfish (especially *Siganus luridus*), which in the southeast Mediterranean (Turkey, Israel) already decimate all erect, edible, macroalgae down to turf barrens (Rilov et al., 2018). The fishes are rapidly spreading to the west and north and the rate of spread of is probably strongly related to water temperature (and is thus probably facilitated by ocean warming).

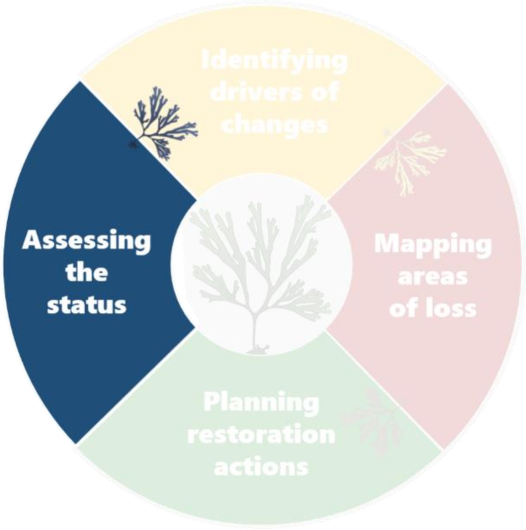
From the validation phase of the modeling procedure we determined the predictive accuracy in spatially independent Mediterranean areas in order to better evaluate model performances. Indeed, testing the model in a wider variety of spatial context means to better define the range of applications for which the model predictions are suited (Guisan and Zimmermann, 2000). In most cases model accuracy was high and presence records matched with pixels classified as suitable for holding *Cystoseira* canopies. In particular, Sardinia resulted having the highest frequency of correctly predicted cases and this result possibly derives from the low presence of anthropogenic pressures affecting the area. Model predictions showed an evident misclassification rate only for the eastern Mediterranean regions and the northern Adriatic Sea. Reasons for model biases probably lie in distinctive conditions, including the presence of human threats or unsuitable environmental conditions, characterizing these coastal areas. A finer resolution of predictor variables would enhance the RF ability to correctly reproduce *Cystoseira* distribution, pointing out environmental heterogeneities hidden under a coarser resolution used in this study. Data on the environmental drivers affecting intertidal and nearshore ecosystems (e.g., human impacts or the type of shoreline) are largely incomplete (Halpern et al., 2008), reducing our ability to assess the present and the future state of marine habitats. Moreover, it should be stressed that especially for the eastern Mediterranean regions, outdated occurrence data may have led to an inaccurate representation of the current distribution of the canopies.

1.I.6 Final remarks

Underpinned by model outputs, we will better direct our management and restoration efforts on the basis of the predictions on the presence/absence of the species, also combined to the information about human pressures. A first model attempt was performed by Buonomo et al. (2018) analyzing intertidal *Cystoseira* populations in order to predict their future ranges according to different climatic scenarios in the Mediterranean Sea. According to that model an important loss of suitable areas is expected across the range of distribution of the habitat-forming seaweed species already by 2050, with cascading effects on the whole ecosystem and the services that it provides.

In this respect, the model output from this study allows to investigate areas classified as suitable with high probability of *Cystoseira* occurrence, to assess if the predicted status of presence matched the real one and thus to define new suitable locations for restoration plans. As a result, the HSM could be seen as a useful baseline tool for the assessment of *Cystoseira* distribution and for the establishment of future-oriented marine planning initiatives from both conservation and restoration point of view, at least as far as the species on the shallow rocky shores. However, there is still a long way until we can use these predictions for true management and (mainly) restoration. Actually, both have to rely on species-specific actions and a (more) spatially accurate information on the environmental factors at the places to be managed or restored. In the case of restoration, it is also pivotal that the pressures that drove the disappearance of the canopies have been mitigated. For this reason, acting on reducing and carefully planning the distribution of local pressures should be considered a priority (Buonomo et al., 2018). In any case, to assess the spatial generality of models, an exhaustive evaluation of how the quality of its output varies within different regional context is required (Gorman et al., 2013). As stressed from the model validation, predictions for Mediterranean regions where model performances are quite limited due to environmental distinctiveness and heterogeneities (e.g., the eastern Mediterranean areas) may not reproduce the actual canopies distribution. In this regard, the large proportion of “no-data” records is an important limit in the development of the HSM and therefore in the understanding of the potential distribution of *Cystoseira* forests across all the Mediterranean coastlines.

Improving model outputs with a finer resolution of predictor variable layers and better dataset with species occurrences would allow more reliable predictions also for these regions and would promote the identification of species-specific suitable and unsuitable areas making our model more sensitive to ecological differences among species.



CHAPTER 1

Part II

AN INTEGRATED ASSESSMENT OF THE GOOD ENIRONMENTAL STATUS OF MEDITERRANEAN MARINE PROTECTED AREAS

CHAPTER 1 Part II

An Integrated Assessessment of the Good Environmental Status

1.II.1 Abstract

Local, regional and global targets have been set to halt marine biodiversity loss. Europe has set its own policy targets to achieve Good Environmental Status (GES) of marine ecosystems by implementing the Marine Strategy Framework Directive (MSFD) across member states. We combined an extensive dataset across five Mediterranean ecoregions including 26 Marine Protected Areas (MPAs), their reference unprotected areas, and a no-trawl case study. Our aim was to assess if MPAs reach GES, if their effects are local or can be detected at ecoregion level or up to a Mediterranean scale, and which are the ecosystem components driving GES achievement. This was undertaken by using the analytical tool NEAT (Nested Environmental status Assessment Tool), which allows an integrated assessment of the status of marine systems. We adopted an ecosystem approach by integrating data from several ecosystem components: the seagrass *Posidonia oceanica*, macroalgae, sea urchins and fish. Thresholds to define the GES were set by dedicated workshops and literature review.

In the Western Mediterranean, most MPAs are in *good/high* status, with *P. oceanica* and fish driving this result within MPAs. However, GES is achieved only at a local level, and the Mediterranean Sea, as a whole, results in a *moderate* environmental status. Macroalgal forests are overall in bad condition, confirming their status at risk. The results are significantly affected by the assumption that discrete observations over small spatial scales are representative of the total extension investigated. This calls for large-scale, dedicated assessments to realistically detect environmental status changes under different conditions.

Understanding MPAs effectiveness in reaching GES is crucial to assess their role as sentinel observatories of marine systems. MPAs and trawling bans can locally contribute to the attainment of GES and to the fulfillment of the MSFD objectives. Building confidence in setting thresholds between GES and non-GES, investing in long-term monitoring, increasing the spatial extent of sampling areas, rethinking and broadening the scope of complementary tools of protection (e.g., Natura 2000 Sites), are indicated as solutions to ameliorate the status of the basin.

Keywords: Good Environmental Status, Thresholds, Ecosystem approach, NEAT, Monitoring, Science-policy gap

List of abbreviations:

EC: Ecosystem Component

EU: European Union

FRA: Fishery Restricted Area

GES: Good Environmental Status

MPA: Marine Protected Area

MSFD: Marine Strategy Framework Directive

NEAT: Nested Environmental status Assessment Tool

OC: Other Controls

OECM: Other Effective area-based Conservation Measures

SAU: Spatial Assessment Unit

SDG: Sustainable Development Goals

UN: United Nations

WFD: Water Framework Directive

1.II.2 Introduction

Local, regional and global targets have been set to guarantee the long-term sustainability of human activities in the ocean, while protecting marine ecosystems. The Aichi Biodiversity Targets and the UN Sustainable Development Goals (SDGs) (UN, 2015) were designed to reconcile environmental protection with socioeconomic development, with SDG 14 specifically introduced for the conservation of the ocean and its sustainable use (Cormier and Elliott, 2017). However, achieving SDGs and, importantly, ensuring that these targets turn into actual biodiversity conservation require substantial steps in bridging the gap between policy and science, rectifying inefficiencies and inadequate management practices (Katsanevakis et al., 2020).

Europe has set its own policy goals to achieve a sustainable development in the European Union (EU) seas, through the implementation of the Water Framework Directive (WFD, 2000/60/CE) and of the Marine Strategy Framework Directive (MSFD, 2008/56/EC), environmental pillars of the EU integrated maritime policy (Fraschetti et al., 2018). The WFD was the first attempt to provide a single system of water management. The MSFD has been conceived to attain the full economic potential of the seas, while integrating environmental protection with a sustainable use of marine resources in a way that they can be preserved in the future, in accordance with SDG 14. Its main objective was to achieve the Good Environmental Status (GES) of marine ecosystems across member states by 2020, using a coordinated approach to monitor and assess their status (Fraschetti et al., 2018). The concept and the normative definitions of GES are based on 11 Descriptors, in line with the Drivers-Activities-Pressures-State-Impact-Welfare-Response approach (Patrício et al., 2016), relating anthropogenic activities and pressures to the state of the marine environment (Elliott et al., 2007). The target is to ensure that no significant risks or impacts are posed on marine biodiversity, marine ecosystems, human health, or legitimate uses of the sea (Smith et al., 2016).

Measuring progress towards meeting targets for ecosystem health is not an easy task and a clear quantitative definition of GES for a marine area is far from being attained (but see Borja et al., 2013). The identification of targets for assessing ecosystems' health requires the adoption of reference conditions, appropriate indicators, systematic monitoring delivering harmonized data with an adequate spatial and temporal coverage, as well as the knowledge of ecosystems' responses to human pressures (Claudet and Fraschetti, 2010). On top of that, ecosystems may shift abruptly in response to environmental perturbations (Oprandi et al., 2020; Scheffer and Carpenter, 2003), but very little information on critical thresholds and on their variability across space and time is available (Boada et al., 2017; Rindi et al., 2017). Our limited knowledge regarding the response of specific structural and functional features of ecosystems to multiple stressors and disturbances (Gissi et al., 2021; Micheli et al., 2013), the inherent spatial and temporal variability in the distribution of ecological features and stressors, and the challenging detection of critical thresholds that lead to regime shifts, are still restraining our potential to quantify and, consequently, achieve and maintain good ecological conditions (Nõges et al., 2016).

Despite its limitations, MSFD offers a strategic framework and an invaluable opportunity for the EU to work towards achieving SDG 14. The MSFD clearly defines Marine Protected Areas (MPAs, that include both fully protected, where all extractive uses are forbidden, and partially protected where some extractive uses, such as fishing, are permitted under regulation) as a main tool for implementing marine biodiversity conservation and promoting healthy ecosystems, while providing opportunities for sustainable local development. Also, Natura 2000 Sites are at the core of the biodiversity conservation strategy of the EU (Evans, 2012). They are based on the Habitats and Birds Directives (92/43/EEC; 2009/147/EC) and do not usually include fully protected zones (Mazaris et al., 2017), having the main target of regulating and managing human activities, contributing to an ecosystem-wide conservation with other national and supranational initiatives (Guidetti et al., 2019).

MPAs play a critical role in the achievement of GES in European seas, even though it is assumed that the GES should be attained also in unprotected areas (Boero et al., 2016): MPAs should be considered sentinel observatories of the effects of multiple human activities, and more broadly of the status of the marine environment as a whole (Grorud-Colvert et al., 2021; Rilov et al., 2020). In addition to MPAs, Fishery Restricted Areas (FRAs) are widely used as fisheries management tools in the framework of different regulatory approaches (Dimarchopoulou et al., 2018). FRAs can be considered as 'Other Effective area-based Conservation Measures' (OECMs) (Petza et al., 2019) including a vast array of different applications that range from temporary to permanent fishing bans and may regard one or more fishing gears. No-trawl areas have been created in the Mediterranean with the purpose of rebuilding overexploited fishery resources and addressing conflicts between fishery sectors, and their effectiveness on fish biomass has been clearly demonstrated (Dimarchopoulou et al., 2018; Pipitone et al., 2014). Given these results such areas can be considered tools for the attainment of GES, more specifically by means of Descriptor D3 on commercially exploited fish. Fish biomass is considered an element of marine waters assessment and of the determination of GES (articles 8 and 9 of MSFD) along with the physical disturbance of the seabed and the extraction of living resources.

The aim of this study is to bridge the science-policy gap by exploring if MPAs and FRAs achieve GES in the Mediterranean Sea, meeting the targets set at EU level. We combined an extensive dataset of well-known interconnected ecosystem components, such as the seagrass Posidonia oceanica, macroalgal forests, sea urchins, and fish, across five Mediterranean ecoregions including 26 MPAs, their control areas, and a notrawl case study to conduct a comparative assessment of environmental health under protected vs. unprotected conditions. This was undertaken by implementing the analytical tool NEAT (Nested Environmental status Assessment Tool, http://www.devotes-project.eu/ neat/), which allows an integrated assessment of marine environmental status.

This work aims at answering the following questions: (i) do Mediterranean MPAs and FRAs contribute significantly to the achievement of GES? (ii) are their effects local or can they be detected at ecoregions up to a Mediterranean scale? (iii) which are the ecosystem components mostly contributing to GES achievement? and, if no GES is achieved, (iv) which ecosystem components deserve urgent conservation actions? (v) which are the gaps for the identification of health status and thresholds of change? and (vi) how solutions and recommendations can be developed to improve the conceptual framework in defining GES?

1.II.3 Materials and methods

1.II.3.1 The case studies

The 26 Mediterranean MPAs analyzed in this study are listed in Table 1.II.1, reporting the ecoregions they belong to, the year of establishment, the ecosystem components analyzed in each MPA, the surface subject to protection, and the extent of the control areas. Table S.1 shows the complete list of controls. Additional Non-Protected Areas (OC = Other Controls), >20 km distant from the MPAs, were also included in the analyses. The eventual presence of a Natura 2000 Site and the Cumulative Human Impact score (CI), based on Halpern et al. (2015) to describe the status of control areas are also indicated.

A no-trawl area has been included as a case study and subjected to an *ad hoc* NEAT assessment to evaluate if and to what extent a year-round trawl ban may contribute to the attainment of GES in the Mediterranean. This case study is made up of a no-trawl area created in 1990 in the Gulf of Castellammare (GCAST, NW Sicily, central Mediterranean) and two trawled control areas along the same Sicilian stretch of coast (the Gulfs of Termini Imerese, GTERM and Sant'Agata, GSANT). Previous studies suggest that fish biomass in GCAST has increased dramatically after the ban (Pipitone et al., 2014). The observed values used in the NEAT assessment (kg km^{-2}) derive from two trawl surveys carried out in 2004–2005 on the continental shelf of the three gulfs. The worst, best and threshold (*moderate/good*) values derive from trawl surveys carried out in the Italian seas from 1994 to 2014 during the MEDITS program (Maiorano et al., 2019). The total fish assemblage and two commercially valuable species (red mullet, *Mullus barbatus* and hake, *Merluccius merluccius*) were chosen as ecosystem components for the analysis. The surface of the three areas is 200 km^2 (GCAST), 280 km^2 (GTERM) and 400 km^2 (GSANT), and their entire surface was covered by the sampling grid.

Table 1.II.1 Spatial Assessment Units (SAUs) included in the dataset for the Mediterranean biogeographic ecoregions. Abbreviated names of MPAs are reported in brackets. YEAR: Year of MPA establishment. EC: available data on Ecosystem Components (P = *P. oceanica*; C = Canopy algae; E = Erect algae; T = Turf; B = Barrens; U = Sea Urchins; F = Fish). For each SAU, in the Protected Areas, both the sampled ("Sampled") and the actual surface area ("Real") are indicated (in km^2). Other controls are represented by Non-Protected areas at a distance greater than 10 km from the MPAs. For the Non-Protected areas, in addition to the sampled surface, a buffer zone of 5 and 10 km around the MPA was considered as the counterpart of the Protected real surface (in km^2). The table also shows the ratio ("%") between the sampled surface and the real surface for Protected Areas and between the sampled surface and the buffer surface of 5 km for Non-Protected areas.

Ecoregion	SAU	YEAR	EC	Descriptor	Protected			Non-Protected			
					Sampled	Real	%	Sampled	5km	10km	%
Adriatic Sea	**Torre Guaceto (TrG)**	1991	P-C-T-F	D 1,4,5,6	0.004	22.27	0.02	0.002	92.27	234.24	0.002
	Telascica (Tel)	2013	F	D 1,4	0.004	70.00	0.01	0.002	155.27	448.39	0.001
	Brijuni (Bri)	2013	E-T-U-F	D 1,4,5,6	0.002	26.00	0.01	0.002	108.37	257.89	0.002
	Other Controls	-	P	D 1,4,6				0.0004	100.77	382.61	0.0004
Aegean Sea	**Alonissos (Alo)**	1996	C-E-T-B-U-F	D 1,4,5,6	2.25	2315.5	0.10				
	Kas (Kas)	1996	C-E-T-B-U-F	D 1,4,5,6	0.002	165.91	0.001	0.002	238.85	476.98	0.001
	Other Controls		C-E-T-B-U-F	D 1,4,5,6				0.04	2805.23	11253.97	0.001
Ionian Sea	**Zakynthos (Zak)**	1996	C-E-T-B-U-F	D 1,4,5,6	0.01	83.30	0.01	0.01	299.81	854.31	0.003
	Porto Cesareo (PtC)	1997	P-C-U	D 1,4,5,6	0.001	166.54	0.001	0.001	153.37	351.72	0.001
	Karaburun-Sazan (Kar)	2016	P	D 1,4,6	0.0004	127.21	0.0003	0.0004	406.64	912.43	0.0001
	Other Controls	-	P	D 1,4,6				0.0004	74.32	269.88	0.001
Tunisian plateau/ Gulf of Sidra	**Isole Pelagie (IPe)**	2002	C-E-U-F	D 1,4,5,6	0.002	41.00	0.01	0.002	226.87	576.33	0.001
	Other Controls	-	-								
Western Mediterranean Sea	**Cinque Terre (CiT)**	1997	P-C-F	D 1,4,5,6	0.02	45.03	0.04	0.01	111.95	290.43	0.01
	Portofino (Por)	1998	P-C-F	D 1,4,5,6	0.02	3.50	0.57	0.01	97.56	250.48	0.01
	Bergeggi (Ber)	2007	P-F	D 1,4,6	0.01	2.06	0.49	0.02	51.76	158.32	0.04
	Asinara (Asi)	2002	U-F	D 1,4,6	0.01	108.03	0.01	0.002	266.82	641.65	0.001

Tavolara (Tav)	1997	U-F	D 1,4,6	0.01	153.57	0.01	0.004	194.69	451.17	0.002
Capo Carbonara (CaC)	1998	F	D 1,4	0.01	143.00	0.004	0.002	188.82	480.06	0.001
Egadi (Ega)	1991	F	D 1,4	0.01	540.17	0.001	0.002	534.27	1127.39	0.0004
Es Freus (EsF)	2000	P-C-E-T-B-U-F	D 1,4,5,6	0.01	150.00	0.01	0.004	224.32	538.56	0.002
Menorca (Men)	2000	P-C	D 1,4,5,6	0.002	56.99	0.004	0.001	134.42	345.24	0.001
Mallorca (Mal)	2000	P-C	D 1,4,5,6	0.002	24.13	0.01				
Cabo de Palos (CdP)	1995	F	D 1,4	0.01	19.31	0.03	0.003	144.49	396.83	0.002
Medes (Med)	2001	P-E-U-F	D 1,4,5,6	0.08	5.00	1.60	0.09	139.68	454.12	0.06
Cap de Creus (CdC)	2001	P-C-F	D 1,4,5,6	0.01	30.73	0.03	0.003	102.66	377.07	0.003
Bonifacio (Bon)	2009	F	D 1,4	0.01	760.00	0.001	0.002	557.44	1123.57	0.0004
Banyuls (Ban)	1974	F	D 1,4	0.01	6.50	0.15	0.003	67.86	214.47	0.004
Cote Bleue (CoB)	2012	C-E-T-B-U-F	D 1,4,5,6	0.01	2.95	0.34	0.01	235.51	518.35	0.004
Cap Roux (CaR)	1998	F	D 1,4	0.002	4.45	0.05	0.004	87.72	310.32	0.01
Other Controls		P-F	D1,4,6				0.07	683.91	2425.8	0.01

1.II.3.2 NEAT analyses and experimental design

NEAT allows integrated assessments by assembling data from various response variables and their associated error over different spatial and temporal scales (Borja et al., 2019, 2021; Pavlidou et al., 2019; Kazanidis et al., 2020). It is based on a hierarchical, nested structure of Spatial Assessment Units (SAUs), i.e. the areas where the environmental status assessment takes place (Borja et al., 2016a; Uusitalo et al., 2016).

Central to the application of NEAT is the need of indicators that are the response variables used to measure the status of each SAU. In addition, each indicator is assigned to specific ecosystem components and to different MSFD descriptors (Table S.2). The overall assessment is an average of the SAUs, weighted by their surface areas (km^2).

Indicators are transformed into values that range from 0 (worst status) to 1 (best status) using a continuous piecewise linear interpolation (Berg et al., 2019). On this scale, the value of 0.60, identified as threshold value, corresponds to the boundary between GES and non-GES. The indicator values are translated to standardized values with four boundaries among different conditions: *high-good* (value of 0.80), *good-moderate* (value of 0.60), *moderate-poor* (value of 0.40) and *poor-bad* (value of 0.20) (Borja et al., 2016a). Though the transformation function is piecewise linear, the definition of five segments or classes allows a reasonable approximation to non-linear functions (Berg et al., 2019) (Box S.1.II.1).

The analyses provide an overall assessment of the environmental status for all SAUs (i.e., the Mediterranean Sea), and a separate assessment for each SAU (i.e., the different MPAs included in the study) or for each of the ecosystem components considered. Each NEAT value has an associated confidence level, which is the probability of being in a determinate class status (*bad, poor, moderate, good, high*). This probability is estimated using the standard error linked to the observed indicator value, which is assumed to represent the mean value of a normal distribution. The resulting assessment was obtained by performing a Monte-Carlo simulation technique with 1000 iterations and using the standard error to repeat the assessment multiple times with simulated values. In this way, each iteration led to different NEAT values, returning a quantitative estimate of confidence level for the original NEAT values, expressed as the percentage of values falling into the five different assessment classes (Borja et al., 2016b).

The nested structure considered for the NEAT assessment is synthesized in Figure S.1.II.1. Each SAU (Level 3) is represented by an MPA or control area hierarchically nested in the Condition (Level 2, protected vs. non-protected) and Ecoregion (Level 1), and includes multiple nested Sites (Level 5) exposed to different protection levels (Level 4).

1.II.3.3 Selection of indicators and ecosystem components

The ecosystem components *P. oceanica*, Canopy algae, Erect algae, Turf, Barren, Sea urchins, and Fish were selected since a sufficient amount of information regarding their spatial occurrence, current status, temporal trends, and strength of ecological interactions is available through the literature (Guidetti, 2006; Sala et al., 2012; Boada et al., 2017; Thibaut et al., 2017; de los Santos et al., 2019; Fabbrizzi et al., 2020). Each ecosystem component was represented by one or more indicators, selected among variables available from the literature (Table S.1.II.2).

Data for the NEAT calculations were provided by the authors, and were collectively organized in a unified dataset. Only data collected during the period 2015–2019 were included to depict the most recent environmental status of the Mediterranean Sea. For each indicator, mean observed values and standard errors were included in the dataset. Overall, we combined a total of 1249 records, comprising data from five Mediterranean ecoregions.

1.II.3.4 Setting thresholds

To set the threshold for each indicator, a combination of literature review and dedicated workshops with experts on different ecosystem components were carried out. We decided to interpret changes of the indicators as non-linear transitions, since there is evidence that linear changes across a gradient of human pressures and conditions rarely occur (Litzow and Hunsicker, 2016) (Box S.1.II.1, Table S.1.II2). Fig. 1.II.2 and Fig. S.1.II.3-8 show the distribution of the values of each indicator across sites (n) within each SAU, grouped by protected and non-protected areas and ecoregions. The thresholds identified for each indicator and outcomes of the NEAT analyses are also included.

1.II.3.5 Analyses performed

NEAT analyses were carried out using different spatial extensions for each SAU. More specifically, we used the actual sampled surface area within and outside the protected area vs. the total protected area and a non-protected buffer of 5 and 10 km for the controls. Buffer zones of 5 and 10 km were selected according to the literature (Zupan et al., 2018), and allowed to obtain comparable surfaces within and outside MPAs (Table 1.II.1).

1.II.4 Results

1.II.4.1 NEAT analyses

NEAT results, at basin scale, provide an overall *moderate* status assessment for the whole Mediterranean Sea, considering Descriptors 1, 4, 5, 6 (corresponding to a value of 0.49, on a scale 0–1), as detected in other studies based on different datasets and approaches (Borja et al., 2019) (Table 1.II.2). At the basin scale, MPAs reflect this condition (value of 0.47), while some unprotected areas are found unexpectedly in a *good* status. The result is mostly due to the generally healthy status of the seagrass *P. oceanica*, which is a priority habitat for protection under the Habitats Directive (Council Directive 92/43/EEC), largely represented also in Natura 2000) Sites and unprotected areas (Fig. 1.II.1, Table S.1.II.1).

At the ecoregion level, a mosaic of conditions is highlighted, confirming that basin scale analyses can capture general trends, but not the regional variability of the selected indicators (Table 1.II.2). The Western Mediterranean (value of 0.65) and the Tunisian plateau (value of 0.78) reach the GES, the Aegean and the Adriatic Seas are in a *moderate* status (0.45 and 0.55 respectively) and the Ionian Sea is in a *poor* status (value of 0.35) (Fig. 1.II.1, Table 1.II.2).The *good* status of the Tunisian plateau is scarcely representative, as the assessment of this ecoregion was based on data limited to one MPA and adjacent controls, despite the high confidence level found in this analysis (over 95%, Table 1.II.2).

Zooming to the MPA scale, most MPAs are in a *good/high* status in the Western Mediterranean, coherently with the result obtained regionally (Fig. 1.II.1, values between 0.65 and 1). Out of their sixteen control areas, six are in a *good/high* status, with three of them being Natura 2000 Sites. Very clear results were also obtained from the analyses testing if no-trawl areas can be considered a tool for the attainment of GES. The output from the NEAT assessment is strikingly clear in showing the effect of the trawl ban (Table 1.II.3). The no-trawl area ranks the highest NEAT values while the two control areas rank lowest, with GTERM ranking lower than GSANT. As regards the analyzed components, the total fish assemblage seems to suffer more than the two species studied in thetrawled gulfs, and red mullet is in worse condition than hake in GTERM (which overall is the area that ranks the lowest).

In the Adriatic Sea, most MPAs and unprotected areas show a *moderate* status, as a result of the contrasting conditions in which the different ecosystem components have been found. In the Ionian Sea, the MPAs of Porto Cesareo in Italy and Karaburun in Albania are found in a good status under both protected and unprotected conditions. In the Aegean Sea, *moderate/poor* conditions are found in both protected and unprotected locations (Fig. 1.II.1).

Noteworthy, all the above results were obtained considering the actual extension of the sampled area (from 0.0004 to 2.52 km^2) that was derived from the sum of the generally low sample effort carried out inside and outside MPAs. The consequence of weighting the analyses on the real extension of the MPAs, and including the buffer areas of 5 and 10 km radius for the controls, as allowed by NEAT, leads to a general downgrading of the detected conditions. In particular, both protected and unprotected Western Mediterranean locations (originally identified as *good*) turn into *moderate*, indicating the consequences of assuming the results obtained from limited spatial scales representative of the actual extension of the area of interest (Fig. 1.II.1; Table 1.II.2). As an example, the high condition identified in Portofino turns into *good* in the MPA and to *moderate* in the unprotected locations.

Considering the ecosystem components, *P. oceanica* is in the best status (*good/high*, corresponding to a shoot density above the thresholds defined for each depth in Table S1.II.2) across locations and independently from the protection regime and the sampling extent (Figure S1.II.3). The same consideration applies to sea urchins that show *good/high* status (corresponding to densities below 5 ind/m^2 and to biomass below 30, 50, 85 g/m^2, respectively for the Eastern Mediterranean and the Western Mediterranean at low or high nutrient concentration) across geographical areas. The overall status for the density/biomass of sea urchins at the scale of MPAs in the Western Mediterranean turns into *moderate* (Fig. 1.II.1, Figure S. 1.II.4) when the sampled area is considered, due to the greater weight of the Medes MPA, which shows a sea urchins biomass of 318 g/m^2. Medes MPA is larger than the other three MPAs of the Western Mediterranean with urchin data (Tavolara, Es Freus, Cote Bleue) taken together. As far as turfs and barrens (Figure S1.II.5 and S1.II.6) are concerned, a *moderate* status (corresponding to a percentage cover between 0 and 5%) is identified independently from the protection regime and the sample extension, indicating a scarce presence of these habitats across SAUs.

Despite the analyses carried out at the basin scale indicated that canopy and erect algae are in bad conditions (below 5% cover), especially under protected regimes, results from the Western Mediterranean show that canopies are in a better condition within MPAs, corresponding to a cover above 50% (Fig. 1.II.2 and S1.II.7). Unexpectedly, in the Adriatic Sea we found that MPAs protect more effectively erect algae, while canopies are apparently in a better condition under a non-protected regime. The same consideration applies to the Ionian Sea. In the Aegean Sea, extensive barrens (cover between 5 and 95%) have been formed by the overgrazing activity of invasive alien rabbitfish regardless of the reef protection status.

Our results stress the local effect of MPAs on the fish component (Figure S1.II.8 a,b). In addition, MPAs reach a better status compared to unprotected areas only when analyses were weighted on the sample extent. Considering the real extension of MPAs together with the control areas worsened the estimated ecological status of fish in the MPAs, possibly also driven by the very high patchiness of the seascape (at any scale) and thus also of the ecological components inside and outside MPAs.

At the ecoregion level, the fish component in MPAs is consistently in a better status in the Western Mediterranean compared to unprotected conditions. Fish are in *poor/bad* and *moderate/poor* status (corresponding to a total biomass below 4250 g/125 m^2 and to a high-level predator biomass below 3580 g/125 m^2) inside MPAs, respectively, in the Ionian and Adriatic Seas. Weighting the analyses on the real MPA extent reduced the differences between protected and unprotected conditions. In general, a worsening of the Adriatic and Ionian Seas respectively to *poor* and *bad* was detected. In the Aegean Sea, the fish component is in *good* state in protected areas and in *poor* state in unprotected areas when considering the sample extension. When weighted, the status of MPAs was reduced to *moderate* (Table 1.II.2).

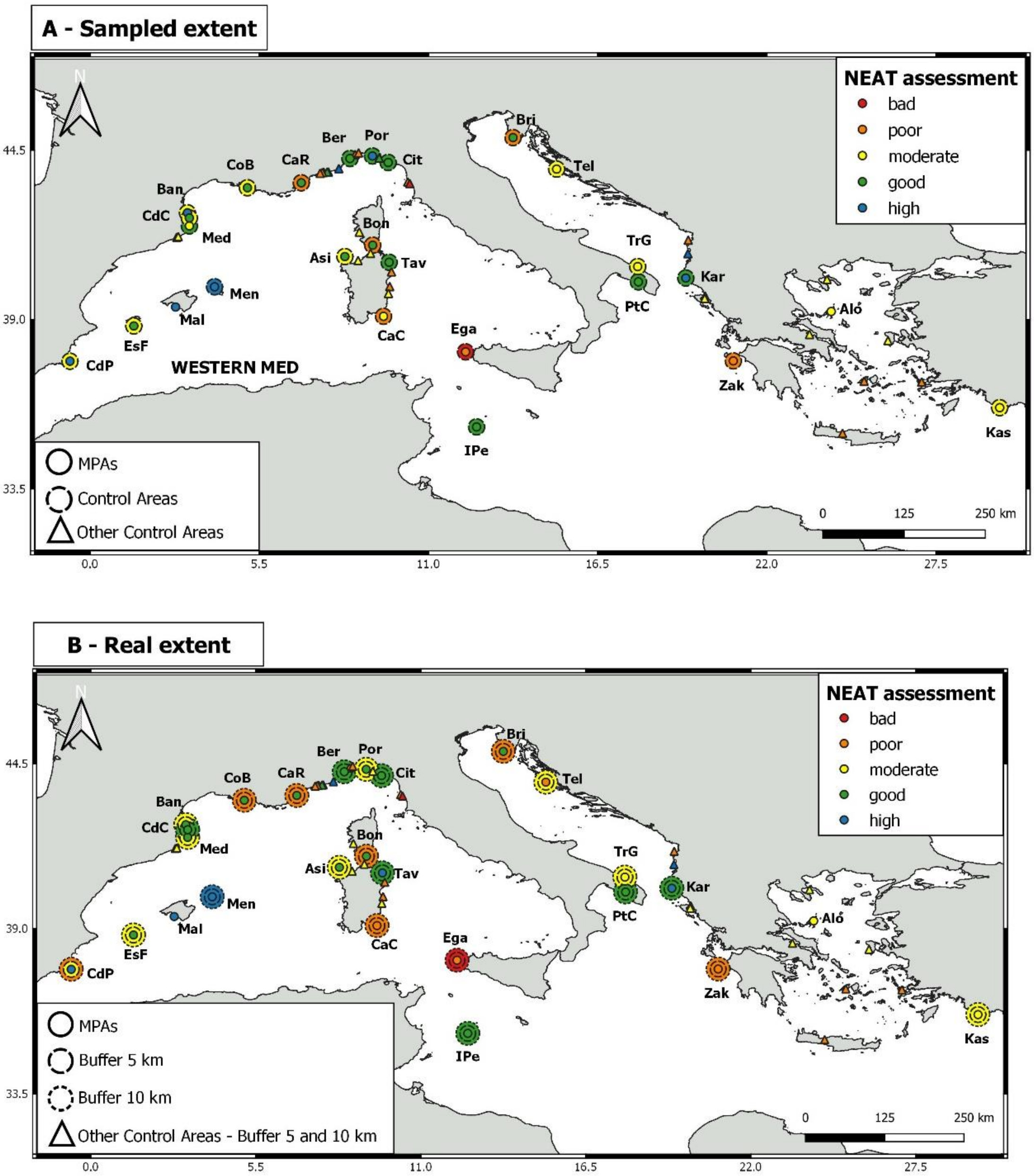


Figure 1.II.1 Distribution of the SAUs across the Mediterranean Sea with the assessment resulting from the NEAT analysis, considering the actual extension of the sampled area (Fig. 1.II.1A) and the real extension of MPAs with the control areas included with the buffer (Fig. 1.II.1B). Colors of the SAUs correspond to their estimated status: red = bad (0.0–0.2), orange = poor (0.2–0.4), yellow = moderate (0.4–0.6), green = good (0.6–0.8), blue = high (0.8–1.0). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 1.II.2 Nested Environmental status Assessment Tool (NEAT) values, considering the actual extension of the sampled area (Table 1.II.2a), the real extension of the Marine Protected Areas (MPAs) with the buffered control areas of 5 km (Table 1.II.2b) and the real extension of the MPAs with the buffered control areas of 10 km (Table 1.II.2c) SAU: Spatial Assessment Unit; PR: protected; MED: whole Mediterranean.

Table 1.II.2a	**Sampled extent**											**Table 1.II.2b**		**Real extent – buffer 5 km**								
SAU	**Area (km^2)**	**NEAT value**	**Status class**	**Confidence level (%)**	**Erect algae**	**Canopy algae**	**Fish**	***P. oceanica***	**Sea urchins**	**Turf**	**Barren**	**Area (km^2)**	**NEAT value**	**Status class**	**Confidence level (%)**	**Erect algae**	**Canopy algae**	**Fish**	***P. oceanica***	**Sea urchins**	**Turf**	**Barren**
MED	2.78	0.49	mod.	100	0.19	0.02	0.58	0.85	0.79	0.56	0.50	13558.79	0.47	mod.	100	0.23	0.16	0.38	0.77	0.87	0.55	0.53
PR	2.48	0.47	mod.	99.7	0.18	0.02	0.62	0.79	0.79	0.56	0.50	5073.14	0.53	mod.	100	0.17	0.10	0.51	0.85	0.86	0.56	0.50
Aegean	2.25	0.45	mod.	97	0.17	0.002	0.62		0.85	0.56	0.50	2481.48	0.45	mod.	98.3	0.16	0.002	0.59		0.87	0.56	0.49
Adriatic	0.01	0.55	mod.	100	0.52	0.38	0.46	0.66	1.00	0.59		118.27	0.48	mod.	99.9	0.52	0.38	0.39	0.69	1.00	0.51	
Ionian	0.01	0.35	poor	99.8	0.02	0.19	0.20	0.78	0.87	0.41	0.79	377.05	0.70	good	100	0.02	0.16	0.18	0.84	0.72	0.41	0.79
Western Med	0.21	0.65	good	98.7	0.83	0.68	0.67	0.80	0.54	0.64	0.97	2055.34	0.58	mod.	93.6	0.78	0.87	0.51	0.88	0.86	0.70	0.97
Tunisian Plateau	0.002	0.78	good	96.1	0.43	0.80	0.64		1.00			41.00	0.78	good	95.3	0.43	0.80	0.64		1.00		
Non-PR	0.30	0.64	good	100	0.45	0.17	0.39	0.87	0.78	0.54	0.58	8485.65	0.44	mod.	100	0.27	0.20	0.31	0.73	0.88	0.55	0.55
Aegean	0.04	0.41	mod.	99.9	0.16	0.03	0.23		0.94	0.53	0.54	3044.08	0.41	mod.	99.9	0.17	0.03	0.22		0.94	0.54	0.53
Adriatic	0.01	0.42	mod.	91.3	0.36	0.41	0.35	0.45	0.49	0.59		456.68	0.46	mod.	99.6	0.36	0.41	0.37	0.54	0.49	0.58	
Ionian	0.01	0.35	poor	100	0.01	0.22	0.15	0.67	0.96	0.45	0.57	934.14	0.53	mod.	100	0.01	0.41	0.16	0.67	0.96	0.45	0.57
Western Med	0.25	0.69	good	100	0.75	0.51	0.42	0.88	0.66	0.61	0.95	3823.88	0.43	mod.	99.7	0.80	0.67	0.33	0.89	0.79	0.66	0.96
Tunisian Plateau	0.002	0.76	good	96.2	1.00	0.52	0.47		0.90			226.87	0.76	good	97.7	1.00	0.52	0.47		0.90		

Table 1.II.2c	**Real extent – buffer 10 km**										
SAU	**Area (km^2)**	**NEAT value**	**Status class**	**Confidence level (%)**	**Erect algae**	**Canopy algae**	**Fish**	***P. oceanica***	**Sea urchins**	**Turf**	**Barren**
MED	**31195.72**	**0.46**	**mod.**	**100**	**0.23**	**0.15**	**0.34**	**0.75**	**0.89**	**0.55**	**0.54**
PR	**5073.14**	**0.53**	**mod.**	**100**	**0.17**	**0.10**	**0.51**	**0.85**	**0.86**	**0.56**	**0.50**
Aegean	**2481.48**	**0.45**	**mod.**	**98.7**	**0.16**	**0.002**	**0.59**		**0.87**	**0.56**	**0.49**
Adriatic	**118.27**	**0.48**	**mod.**	**100**	**0.52**	**0.38**	**0.39**	**0.69**	**1.00**	**0.51**	
Ionian	**377.05**	**0.70**	**good**	**100**	**0.02**	**0.16**	**0.18**	**0.84**	**0.72**	**0.41**	**0.79**
Western Med	**2055.34**	**0.58**	**mod.**	**94.6**	**0.78**	**0.87**	**0.51**	**0.88**	**0.86**	**0.70**	**0.97**
Tunisian Plateau	**41.00**	**0.78**	**good**	**95.9**	**0.43**	**0.80**	**0.64**		**1.00**		
Non-PR	**26122.58**	**0.44**	**mod.**	**100**	**0.24**	**0.16**	**0.31**	**0.74**	**0.89**	**0.54**	**0.54**
Aegean	**11730.95**	**0.40**	**mod.**	**93.8**	**0.16**	**0.03**	**0.22**		**0.93**	**0.54**	**0.54**
Adriatic	**1323.13**	**0.46**	**mod.**	**99.9**	**0.36**	**0.41**	**0.38**	**0.54**	**0.49**	**0.59**	
Ionian	**2388.34**	**0.51**	**mod.**	**100**	**0.01**	**0.39**	**0.16**	**0.67**	**0.96**	**0.45**	**0.57**
Western Med	**10103.83**	**0.45**	**mod.**	**100**	**0.79**	**0.68**	**0.34**	**0.89**	**0.79**	**0.67**	**0.97**
Tunisian Plateau	**576.33**	**0.76**	**good**	**96.6**	**1.00**	**0.52**	**0.47**		**0.90**		

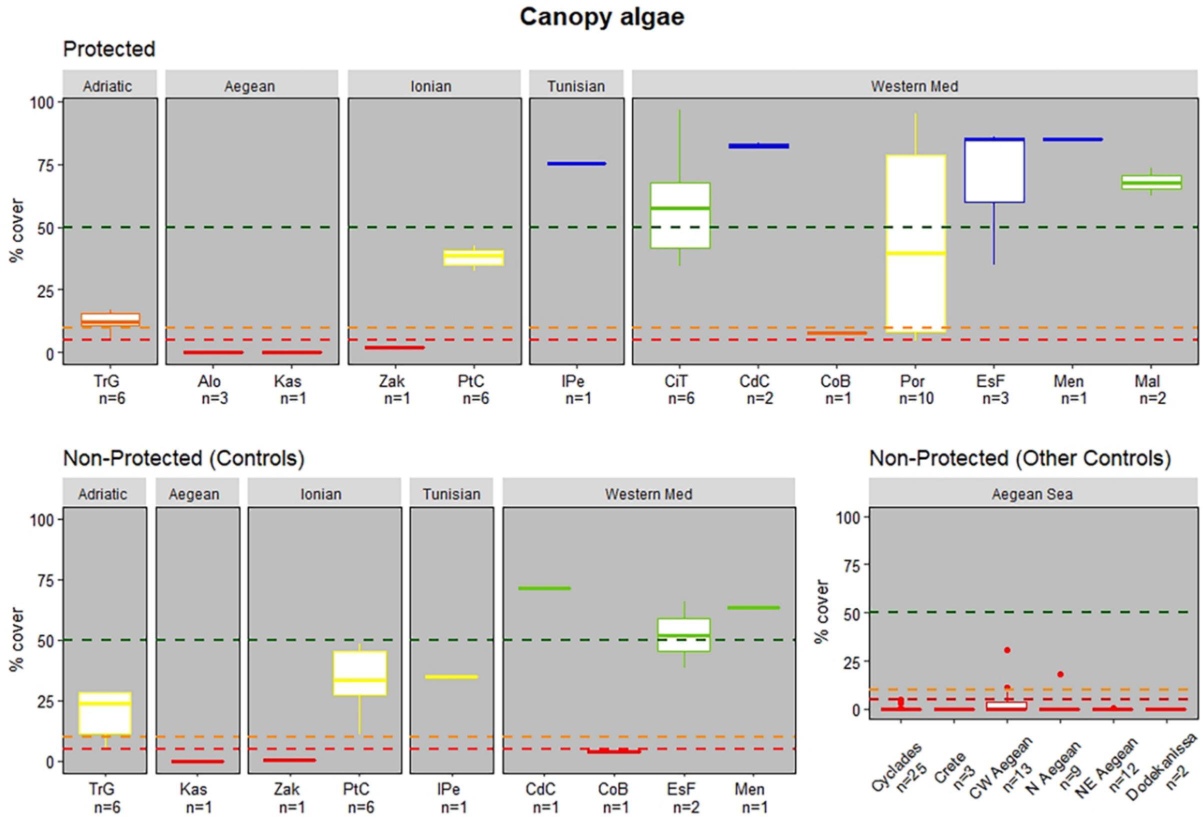


Figure 1.II.2 The figure shows the distribution of the percentage cover values across sites (“n” = number of sites in each SAU) collected for Canopy algae grouped by protected and non-protected areas and ecoregions. Selected thresholds are also included as dashed lines: red = bad/poor (5%); orange = poor/moderate (10%); green = moderate/good (50%). Colors of the boxplots corresponds to the outcomes of the NEAT analyses. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 1.II.3 NEAT output for the Sicilian no-trawl case study. GCAST: no-trawl area; GTERM, GSANT: trawled (control) areas.

SAU	NEAT value	Status class	Confidence level (%)	*Merluccius merluccius*	*Mullus barbatus*	Total teleosts
NW Sicily	**0.464**	**mod.**	**100**	**0.533**	**0.438**	**0.423**
GCAST - No trawl	**1.000**	**high**	**100**	**1.000**	**1.000**	**1.000**
GTERM - Ctrl1	**0.164**	**bad**	**99.7**	**0.264**	**0.106**	**0.120**
GSANT - Ctrl2	**0.230**	**poor**	**80.9**	**0.334**	**0.207**	**0.148**

1.II.4 Discussion

Despite the limitations in upscaling the assessments from a local condition (MPAs) to the basin-ecoregion level for information scarcity, the use of NEAT introduces some interesting insights. Available information provides evidence that the Mediterranean Sea is in a *moderate* environmental status for all MSFD Descriptors considered. However, a complex pattern of conditions was found, differing across scales and ecosystem components, reflecting the context dependency of the status of marine systems and the different management regimes in the Mediterranean Sea. Zooming at ecoregion scale, the Western Mediterranean Sea is found in GES. This result is possibly driven by the effects of synergistic management actions for biodiversity protection (MPAs, including Natura 2000 Sites) and interventions to improve water quality, documented at national and subnational scales: the increase of wastewater treatment plants from 2003 to 2010 along the Catalonia coast in Spain resulted in significant improvements of water quality, with positive effects on both macroalgal canopies and *P. oceanica* (Roca et al., 2015). These results are in agreement with Micheli et al. (2013), who detected a medium cumulative impact in the Mediterranean Sea and the lowest cumulative impact score in its Western basin, although areas of high impact exist within this ecoregion, as our NEAT analysis confirms. Most of the MPAs in the Western Mediterranean Sea are assigned to good/high status. This means that Mediterranean MPAs and FRAs contribute significantly to the achievement of GES. They are already effective tools for the fulfilment of the MSFD objectives, especially because of their generally positive effect on fish assemblages, and the local restoration of top-down control on herbivores (mostly sea urchins) by predatory fish, which, in turn, allows more structured and abundant macroalgal canopies to develop within MPAs. Our findings are consistent with what has been found in several studies considering single descriptors (mainly fish), comparing protected vs. unprotected conditions and confirm that fish, in well enforced protected areas, can reach GES, possibly affecting other ecosystem components even in “crowded” marine environments (Giakoumi et al., 2017).

From available data, the Adriatic and Ionian regions, are, respectively, found at a *moderate* and *poor* state. Fraschetti et al. (2018) and Gissi et al. (2017) recently showed the limits and uncertainties in their conservation, management and cumulative impacts assessment. These areas should be prioritized in terms of concrete management actions coordinated at transboundary levels (Gissi et al., 2018), including transparent data sharing to complement information from different research projects and fields (Cavallo et al., 2018; Pınarbas¸ı et al., 2020) and monitoring programs. In the Adriatic Sea, the GES has not been attained in most MPAs and unprotected areas, despite the effectiveness of protection shown from the literature in MPAs such as Torre Guaceto (Guidetti, 2006). The status found is still suboptimal considering the potential GES of the indicators assessed at Mediterranean scale, stressing the need of integrating more ecosystem components in the analysis to better depict the condition of an area (Borja et al., 2019; Pavlidou et al., 2019; Kazanidis et al., 2020). It is also a paradigmatic example of the need to integrate the decision about the NEAT thresholds, common across sites, with the knowledge of the ecological contingencies (e.g., the frequency and intensity of present-past disturbances, seafloor conditions and spatial context) with the consequence that each site may have thresholds that cannot be exceeded. In this respect, Torre Guaceto, most likely due to its specific environmental features (e.g., habitat types and complexity, depth, etc.), has never been reported to host wide populations of large-sized nekto-benthic predatory fishes (e.g. dusky grouper and brown meagre), independently from the effectiveness of the protection regime (Guidetti et al., 2014). Future analyses that incorporate ‘noisy’ spatial and temporal contingencies may find that system-specific thresholds are more common than universal ones (Dudney and Suding, 2020).

Considering the remaining regions, the *moderate/poor* conditions detected in the Aegean Sea are not surprising, since most MPAs in that area generally suffer from low enforcement (Sini et al., 2017), while several ecological features have been found in a relatively poor state in unprotected areas (Bevilacqua et al., 2020; Sini et al., 2019). In the Ionian Sea, Zakynthos MPA was designated for the protection of sea turtles. The present management scheme has been shown to be ineffective in protecting other ecosystem components, such as fish populations (Dimitriadis et al., 2018). Although the Tunisian Plateau was found in a good state, the lack of data regarding the status of marine ecosystems and their protection in the entire southern Mediterranean remains a limiting factor in regional assessments and planning studies (Giakoumi et al., 2013, 2017). Recent studies from the southeastern Levant basin (not included in this study) showed that the overall ecological status of the coastal zone in this ecoregion is poor. Shallow reefs are mostly dominated by turf (canopy algae are rare, seagrass is absent) and alien species, even inside the one well-enforced long-term marine reserve, although the fish community inside the reserve was in better condition than outside (Rilov et al., 2018). This region also suffers from an immense loss of native biodiversity (mostly mollusca but also sea urchins), probably due to ocean warming (Rilov, 2016; Yeruham et al., 2019; Albano et al., 2021), and the consequences of takeover by alien species on reef ecosystem functioning can be considerable (Peleg et al., 2020). Under the unfolding rapid climate change, in the expending areas where sensitive native species are being lost due to warming and tropical aliens takeover, we might need to adjust some of the criteria for GES (Rilov et al., 2020), as the local biodiversity is and will be completely reshuffled (Edelist et al., 2013).

Very clear results were obtained from the analysis from the no-trawl area. These results, although limited to Italian waters, support the use of year-round trawl bans as a tool for the fulfilment of the MSFD objectives based on Descriptor 3 (i.e., populations of all commercially exploited fish and shellfish are within safe biological limits), but their contribution to GES can actually be much wider: other ecosystem elements and functions may benefit from a healthy fish assemblage, in particular biodiversity, food webs and sea floor integrity (Descriptors 1, 4 and 6, respectively, within the MSFD). Moreover, since all other uses are permitted in the selected case study (Gulf of Castellammare), including small-scale fishing which has economically benefited from the ban applied to the competitive large-scale trawling activity (Whitmarsh et al., 2003), the trawl ban provides an effective area-based management tool for the sustainable use of the marine ecosystem in general at the basin scale (Pipitone et al., 2014).

MPA effects are local, with *P. oceanica* and fish generally in *good/high* status within them (Bevilacqua et al., 2020). Despite a declining trend indicated by global assessments of seagrasses (de los Santos et al., 2019; Marba ` et al., 2014), our findings on the health status of *P. oceanica* are aligned with those from a recent review on the ecological status of seagrass beds and other marine ecosystems at the basin scale, where more than 70% of the 700 investigated sites exhibited *good* to *high* status (Bevilacqua et al., 2020) possibly thanks to the latest conservation policies (Burgos et al., 2017). This result demonstrates that despite the intensity of human pressures in the Mediterranean, there are still opportunities for a significant recovery of marine ecosystems if human impacts are locally reduced. Algal forests formed by canopy and erect algae seem to be the most challenging components for conservation, as they were overall found in *bad* condition, both in protected and non-protected areas at the basin scale. This result is in accordance with Gubbay et al. (2016) and Bevilacqua et al. (2020), who found that about two-thirds of subtidal rocky reef sites are classified in *moderate/bad* conditions. MPAs alone cannot do much for the recovery of canopy algae (Tamburello et al., 2022). Additional conservation actions are needed, such as improvement of water quality, control of indigenous and invasive herbivores (Yeruham et al., 2019), and implementation of restoration actions (De La Fuente et al., 2019; Fraschetti et al., 2021), to stop their loss.

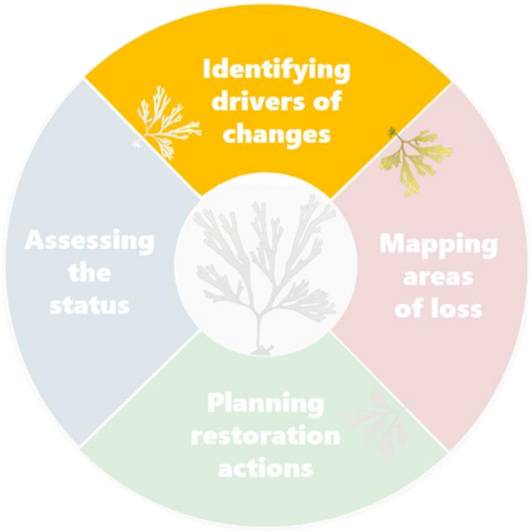
MPAs effects are local since the GES has not been found in most unprotected areas and Natura 2000 Sites, underlining that, despite the fish spillover effect of MPAs, their global effect on the environmental status of surrounding areas is limited (Di Lorenzo et al., 2020). In this respect, it is crucial to rethink and broaden the scope of Natura 2000) Sites to improve their conservation capacity and outcomes (Guidetti et al., 2019; Mazaris et al., 2019; Manea et al., 2020) since, despite being considered the largest conservation network globally, they are often found in a *poor/moderate* status (Table S1.II.1).

Central to attain these results was the challenge of setting thresholds for the ecosystem components included in the analysis. The decision about "what is good" and "what is not" is not trivial (Borja et al., 2013; Hillebrand et al., 2020), even for components like fish that have been the focus of many studies assessing the effectiveness of MPAs (Box S1.II.1). The use of available data from well enforced MPAs was suggested as a possible pathway to set up baselines for fish, but different approaches were adopted for the other ecosystem components such as *P. oceanica*, the thresholds of which were derived from Pergent et al. (1999). In addition, recent studies highlighted that regime shifts may present hysteretic behavior and are highly dependent on regional conditions (Boada et al., 2017; Rindi et al., 2017; Scheffer and Carpenter, 2003), making the identification of a single threshold value not accurate, as required by NEAT (Box S1.II.1). Rapid changes of ecosystems in the Anthropocene are further challenging the way we measure thresholds of changes. Dedicated projects should develop a framework to identify ecological thresholds across environmental conditions and gradients of human pressures, to detect the prevalence of strong nonlinearities (Rindi et al., 2017).

Despite this collaborative effort to enhance sample sizes and broaden the scale and scope of the study, we realized that the majority of ecological studies addressing the patterns of spatial-temporal variability for some of the response variables at Mediterranean scale tend to upscale the results obtained by samples covering just a few square meters to very large extensions. This asks for more investments in systematic surveys and monitoring, under protected and non-protected conditions to provide realistic GES assessments.

It is not only an issue of spatial extension. The knowledge of thresholds is also largely connected with the need for long-term data, as ranges of natural variation are identified and temporal trends emerge with prolonged observation (Gatti et al., 2015; Hughes et al., 2017). The scarcity of long-term datasets and the limited knowledge across space and time hinder our potential to tease apart the natural variability from the effects of human impacts. Our analyses clearly show that data availability is still a challenge in coastal protected and unprotected habitats, despite the effort carried out in these systems (Levin et al., 2014). We found that data availability is scattered across MPAs and systematic monitoring outside MPAs is available mainly for *P. oceanica*, stressing the need for increased monitoring efforts also on other ecosystem components, using an integrated perspective. As stressed by Micheli et al. (2020), at a time when the need for informed mitigation and adaptive action is accelerating, investment in long-term studies has perversely decreased.

Despite these limits, gaps and challenges, many areas, albeit small, show that the GES can be reached with proper management. In this respect, NEAT can facilitate the assessment process of MPAs, allowing to integrate different information and providing an overall overview (Borja et al., 2021). In addition, ensuring a better alignment between different initiatives at Mediterranean level (e.g., MSFD and Ecosystem Approach Strategy) would foster a shared vision and synergistic approaches to enhance the protection and the recovery of the Mediterranean marine environment (Cinnirella et al., 2014) The MSFD represents an opportunity to understand how species, habitats and entire ecosystems respond to environmental changes and ever-growing human pressures. As recommended by Katsanevakis et al. (2020), only a change of vision about the importance of decreasing human pressures aimed at developing a sustainable economy to support healthy socio-ecological systems will allow the achievement of GES both locally and regionally.



CHAPTER 2

EFFECTS OF NATURAL AND ANTHROPOGENIC STRESSORS ON FUCALEAN BROWN SEAWEEDS ACROSS DIFFERENT SPATIAL SCALES IN THE MEDITERRANEAN SEA

2.1 Abstract

Algal habitat-forming forests composed of fucalean brown seaweeds (*Cystoseira*, *Ericaria*, and *Gongolaria*) have severely declined along the Mediterranean coasts, endangering the maintenance of essential ecosystem services. Numerous factors determine the loss of these assemblages and operate at different spatial scales, which must be identified to plan conservation and restoration actions. To explore the critical stressors (natural and anthropogenic) that may cause habitat degradation, we investigated (a) the patterns of variability of fucalean forests in percentage cover (abundance) at three spatial scales (location, forest, transect) by visual estimates and or photographic sampling to identify relevant spatial scales of variation, (b) the correlation between semi-quantitative anthropogenic stressors, individually or cumulatively (MA-LUSI index), including natural stressors (confinement, sea urchin grazing), and percentage cover of functional groups (perennial, semi-perennial) at forest spatial scale. The results showed that impacts from mariculture and urbanization seem to be the main stressors affecting habitat-forming species. In particular, while mariculture, urbanization, and cumulative anthropogenic stress negatively correlated with the percentage cover of perennial fucalean species, the same stressors were positively correlated with the percentage cover of the semi-perennial *Cystoseira compressa* and *C. compressa* subsp. *pustulata*. Our results indicate that human impacts can determine spatial patterns in these fragmented and heterogeneous marine habitats, thus stressing the need of carefully considering scale-dependent ecological processes to support conservation and restoration.

Keywords: Macroalgal forests, life history, PERMANOVA, RDA, percentage cover

2.2 Introduction

Predicting the effects of abiotic and biotic stressors on marine vegetation changes has been a central concern of marine science in the last decades world-wide (Lotze et al., 2006; Airoldi and Beck, 2007; Coleman et al., 2008), and in the Mediterranean sea (Viaroli et al., 2008; Macreadie et al., 2014; Papathanasiou et al., 2015; Tsioli et al., 2019). Among the different algal groups, special attention has been devoted to canopy-forming fucalean brown algae (Strain et al., 2014; Coleman and Wernberg, 2017), characterized by slow growth and limited propagule dispersal and, therefore, unable to respond rapidly to anthropogenic and climatic changes (Buonomo et al., 2017; Bermejo et al., 2018). In turn, since the fucalean brown algae form some of the most productive, diverse, and valuable marine habitats (Benedetti-Cecchi et al., 2001), providing essential ecosystem services (Cheminée et al., 2013; Mineur et al., 2015; Bianchelli and Danovaro, 2020), they are recognized as a subtype of a natural habitat type (Reefs, code 1170) in need of conservation in Europe (Directive 92/43/EEC). Data from laboratory experiments with artificial seagrass showed that leaf area index (LAI), which combines the effect of leaf length and shoot density and therefore can be used as a metric of meadow abundance, was positively related to the delivery of ecosystem services such as wave attenuation (Paul et al., 2012), highlighting the need for a better understanding of the scale based abundance patterns (Townsend et al., 2018) under marine pristine and degraded ecosystems (Benedetti-Cecchi et al., 2001).

Algal forests composed by the genera *Cystoseira*, *Ericaria*, and *Gongolaria* (Fucales), with the exception of *C. compressa*, likely represent the most endangered habitat in the Mediterranean Sea (Barcelona Convention-Annex II; United Nations Environment Programme/Mediterranean Action Plan-UNEP/MAP; Verlaque et al., 2019) and have undergone a major decline in the last decades (Thibaut et al., 2005; Blanfuné et al., 2016; Rindi et al., 2020). Although the processes driving fucalean species diversity, abundance, and coexistence along environmental gradients are still unclear, the cumulative impacts of local anthropogenic stressors such as coastal development, habitat destruction, pollution, and fisheries along with climate change are considered the main causes (Sala et al., 2012; Gianni et al., 2013; Strain et al., 2014; Mineur et al., 2015; Bianchi et al., 2018; Fabbrizzi et al., 2020). In addition, human over-exploitation of top predators of sea urchins (e.g., the sparid fishes *Diplodus sargus* and *D. vulgaris* (Guidetti, 2004), has reduced the control over this component, thus leading to sea urchin overgrazing on algal forests, creating barrens (Ling et al., 2015). In addition, disruption of climatic patterns with more frequent and stronger storm events, as well as heat waves, which are becoming increasingly frequent during summer in the whole Mediterranean Sea (Darmaraki et al., 2019), are expected to have severe effects at local scales on fragmented populations of fucalean species (Verdura et al., 2021).

When fucalean populations are reduced or become locally extinct, their natural recovery is almost impossible, primarily due to their slow growth and the low dispersal abilities of their large and heavy zygotes (Orfanidis, 1991; Ballesteros et al., 2009; Buonomo et al., 2017; Bermejo et al., 2018). Although restoration by transplantation or recruitment enhancement techniques has emerged as a tool to restore and prevent species loss (Susini et al., 2007; Marzinelli et al., 2014, 2016; Shabnam et al., 2015; Falace et al., 2018; Verdura et al., 2018; Medrano et al., 2020), not all restoration initiatives have been successful (Coleman and Wernberg, 2017; Tamburello et al., 2019).

The knowledge required to disentangle the key drivers and abiotic variables that influence the distribution and abundance of fucalean seaweed species is still incomplete, mainly when the link between species decline and environmental conditions is considered (de Caralt et al., 2020). Therefore, understanding the reasons of the persistence and loss of benthic forests is crucial to evaluate the role of ecological and evolutionary processes and to manage direct and indirect anthropogenic stressors.

The determination of the most appropriate spatial scales for investigating species/environment relationships, directly related to the scale of ecological processes, is another crucial issue (O'Neill et al., 1989). For example, to explain differences in fucalean seaweed abundance and distribution, an understanding of the underlying ecological processes is necessary (Rindi and Guiry, 2004; Orfanidis et al., 2008), and therefore, quantification of their spatial patterns is needed (Wiens et al., 1993; Underwood, 1996; Benedetti-Cecchi et al., 2001; Fraschetti et al., 2005). Despite this interest in scale-specific patterns, effective implementation of multiscale approaches in theoretical and empirical research is still limited (Benedetti-Cecchi et al., 2001; Fraschetti et al., 2005; Mancuso et al., 2018).

To explore the stressors (natural and anthropogenic) estimated semi-quantitatively that may cause fucalean habitat degradation, we studied the patterns of variability of 11 species and subspecific taxa in abundance (expressed as percentage cover) to identify relevant scale of spatial variation. Indeed, five species or subspecies of the genus *Cystoseira* [*C. compressa* (Esper) Gerloff and Nizamuddin, *C. compressa* subsp. *pustulata* (Ercegovic) Verlaque, *C. corniculata* (Turner) Zanardini, *C. foeniculacea* (Linnaeus) Greville, *C. foeniculacea* f. *tenuiramosa* (Ercegovic) A. Gómez Garreta, M.C. Barceló, M.A. Ribera, and J. Rull Lluch], four of the genus *Ericaria* [*E. amentacea* (C. Agardh) Molinari and Guiry, *E. barbatula* (Kützing) Molinari, and Guiry, *E. crinita* (Duby) Molinari, and Guiry, and *E. mediterranea* (Sauvageau) Molinari and Guiry], and two of the genus *Gongolaria* [*G. barbata* (Stackhouse) Kuntze, *G. elegans* (Sauvageau) Molinari, and Guiry] were studied. The study was performed at three spatial scales (ranging from a few meters to 10 s of kilometers) in five countries and eight different locations across the Mediterranean Sea (2 in Greece; 1 in Albania; 2 in Italy; 2 in Spain; 1 in Turkey). In terms of functional morphology, *C. compressa* and *C. compressa* subsp. *pustulata* are semi-perennial or pseudo-perennial (i.e., algae in which most of the thallus is lost every year and the species persists in unfavorable seasons in the form of a small holdfast), whereas the others are perennial (i.e., algae in which a more or less large part of the thallus persists continuously for many years). We also explored the potential correlation between stressors individually or cumulatively (MA-LUSI index), including natural stressors (confinement, sea urchin grazing), and the functional groups (perennial, semi-perennial) percentage cover at forest spatial scale.

2.3 Material and Methods

2.3.1 Sampling locations

Sampling was conducted selecting randomly from a number of locations of the Mediterranean Sea where the presence and long-term persistence of the fucalean genera *Cystoseira*, *Ericaria*, and *Gongolaria* was documented by previous data or observations (Figure 2.1). A description of the locations is provided in Supplementary Material.

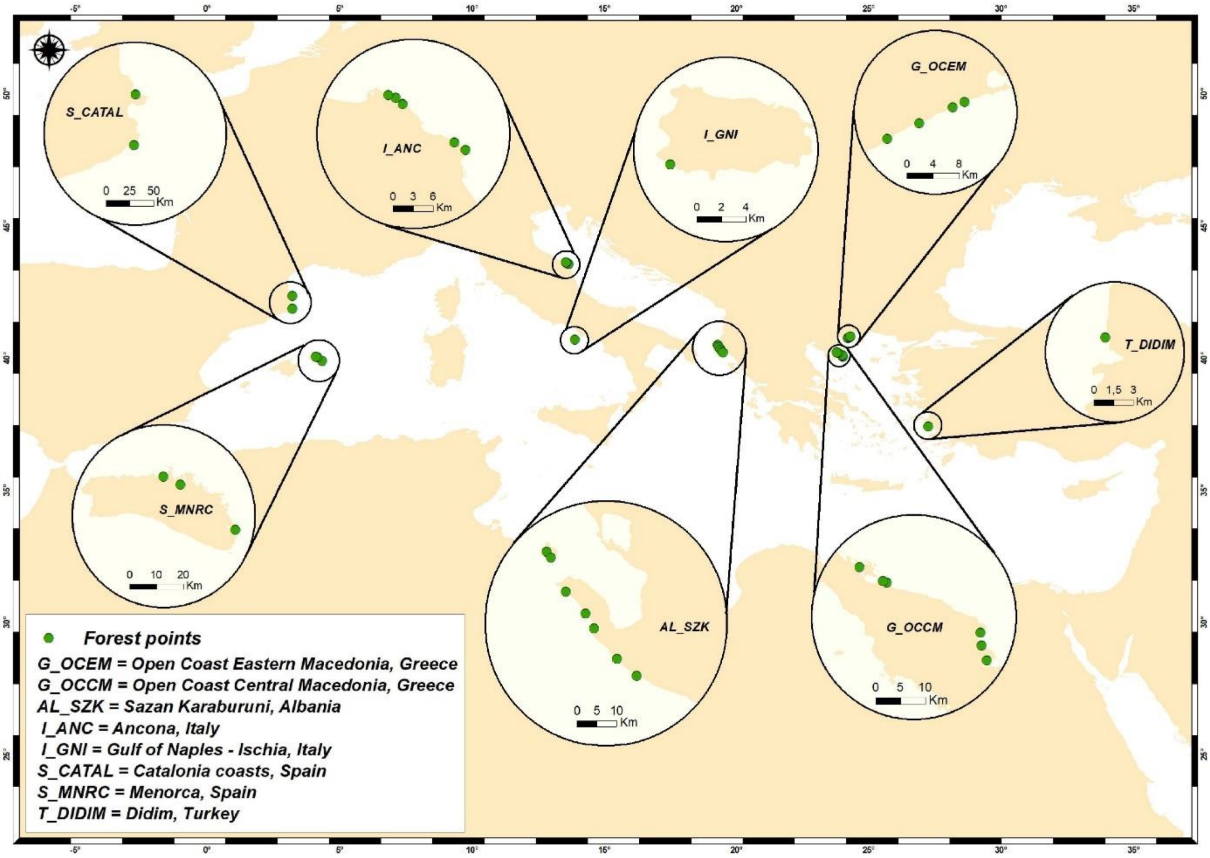


Figure 2.1 A map of the sampled fucalean seaweed forests in the Mediterranean Sea.

2.3.2 Sampling design

Cystoseira, *Ericaria*, and *Gongolaria* forests were sampled following a random nested design with a hierarchy of three spatial scales: transect, forest, location (Figure 2.2). An exception was the locations I_GNI and T_DIDIM, where the sampling was implemented at two spatial scales, i.e. transect and forest, due to the limited extent of the only forest present. Sampling was undertaken during the warm time of the year (corresponding in the Mediterranean to late spring and summer), from May to October 2019 (and in July 2020 for T_DIDIM). In this period, sea surface temperature ranges between 20 and 25°C in most of the Mediterranean. This is the most suitable time of the year to survey fucalean algae and evaluate their percentage cover, because both perennial and semi-perennial species occur in the field in their fully developed habit (consisting of a crust- or disk-like holdfast, a stipe and many branched fronds). During this period, in each location, one or more forests were selected, representing a spatial scale that equals the site's spatial scale. Several transects parallel to the shore were selected randomly within each forest, depending on the forest's extent (Figure 2.2). A georeferenced 10- m graduated transect line was placed in a randomly selected spot within a forest down to 1–1.5 m depth. Three to twelve (n = 3– 12) metallic or PVC quadrats (20 × 20, 25 × 25, 30 × 30, or 50 × 50 cm) were placed randomly in parallel to each transect at 0.5 m distance from the line. The fucalean species within the quadrats were recorded, and their percentage cover was estimated by visual census or photographs that were subsequently analyzed by means of imaging software (PhotoQuad; Trygonis and Sini, 2012). While the two forests sampled in each location were at least 1,000 m apart, the distance between transects was higher than 10 and less than 900 m. The numbers of locations, forests, transects, and quadrats sampled for each location are summarized in Table 2.1.

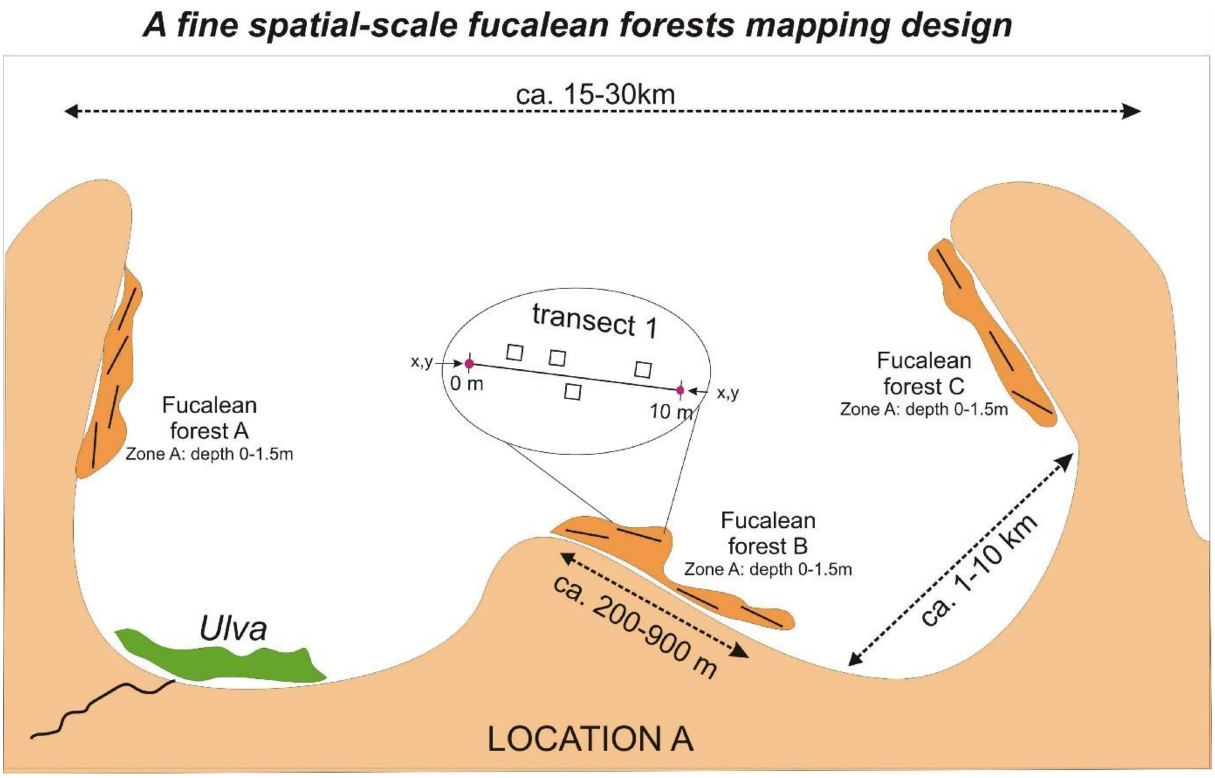


Figure 2.2 Scheme illustrating the random nested sampling design on a hierarchy of three spatial scales (transect, meadow, and location) used to study fucalean forests percentage cover and frequency.

Table 2.1 Sampling design used for fucalean forests abundance and frequency estimations. G_OCEM: Open coasts Eastern Macedonia, Greece; G_OCCM: Open coasts Central Macedonia, Greece; I_ANC: Conero Riviera—Ancona, Italy; I_GNI: Gulf of Naples, Italy; AL_SZK: Sazan Karaburuni MPA—Sazan, Albania; S_MNRC: Menorca, Spain; S_CATAL: Catalonia coasts, Spain; T_DIDIM: Didim, Turkey

Country	**Greece**		**Italy**		**Spain**		**Albania**	**Turkey**
Location	**G_OCEM**	**G-OCCM**	**I_ANC**	**I_GNI**	**S_MNRC**	**S_CATAL**	**AL_SZK**	**T_DIDIM**
No. forest	5	4	5	1	3	2	7	1
No. transects	14	16	27	4	9	5	28	6
No. quadrats/transects	12	12	6	3	12	12	3	10
No. quadrats	168	192	162	12	108	60	84	60
Quadrat size (cm)	25 x 25		50 x 50	50 x 50	20 x 20		50 x 50	25 x 25
Cover (%) estimation method	Visual estimates		Photographic analysis (ANC) Visual estimates (GNI)		Visual estimates		Visual estimates	Visual estimates

2.3.3 Anthropogenic and biological stress Index

MA-LUSI is a cumulative index of anthropogenic stress (GIG, 2013; Papathanasiou and Orfanidis, 2018) specific for coastal water benthic macrophytes and inspired by the LUSI index (Flo et al., 2011, 2019). To calculate the quantitative index, information of key indirect and direct stressors in a 3-km buffer zone around the sampling forests is needed. The ESRI GIS software was used to create the buffer zones on the Corine Land Cover database 2018[1] and to assess the extension of indirect stressors such as urban, agricultural (irrigated land), and industrial pressures in terms of percentage of land cover accounted for by the respective activities. The semi-quantitative information on stressors is classified and assigned in a score (Supplementary Table 2.1). Direct stressor categories A and B and their corresponding scores are assigned to describe Sewage outfall, Mariculture, Sediment that releases nutrients, Freshwater inputs, Harbor-related impacts (Supplementary Tables 2.2, 2.3). All the scores were summed together and multiplied by correction factors related to hydrology and coastline

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[1] https://land.copernicus.eu/pan-european/corine-land-cover

confinement (Supplementary Table 2.4) to obtain a numerical value. Herbivory on fucalean forests was estimated by the number of sea urchins found close to the sampled transects.

2.3.4 Statistical analysis

Frequency indicates the number of times a species was present within a given number of sampled quadrates. It was measured by noting the presence of a species in randomly sampled locations, which are distributed as widely as possible throughout the study area. Since frequency is highly influenced by the size of the quadrats used, we calculated it at a transect scale for which the chosen size was 10 m across the Mediterranean locations sampled. Frequency (%) = (Number of sampling transects in which the species occurs)/(Total number of sampling transects employed for the study)* 100. The percent (%) area of the quadrat covered by a species was used as a measure of abundance as follows: The mean percentage cover = (Total percentage cover of a species in transects in which the species occurs)/(Total number of transects in which the species occur)* 100. The mean percentage cover of each species at each transect was calculated as the mean percentage cover of the species at quadrates sampled in the transect. To test whether the number of quadrats sampled in each location was sufficient to represent the existing fucalean species variance, the T-S species accumulation curves of Ugland et al. (2003) were used. The species accumulation curve describes the accumulation rates of new species over the sampled area and depends on species identity. Differences in patterns of distribution across spatial scales were tested using Permutational Multivariate Analysis of Variance (PERMANOVA) of non-transformed percentage cover data based on Bray–Curtis dissimilarities, using location (6 levels as random factor), forests (3–7 levels, nested in location, random) and transect (2–12 levels, nested in forests, random) as random variables, n = 3–12 for each transect. Due to the unbalanced design, results were interpreted with a more conservative significance level of $\alpha = 0.01$ (Underwood, 1996). The locations I_GNI and T_DIDIM weren't included in these analyses since each location hosted a single forest. Multivariate analyses (nMDS, SIMPROF, and SIMPER) were plotted to visualize and explain patterns of dissimilarities at the scale of locations, forests, transects, and spatial relationships among the fucalean species. For all the above analyses, Primer version 7 was used with the add-on package PERMANOVA+. Relationships between percentage cover and anthropogenic (urbanization, agriculture, mariculture, sewage discharge, harbor-related pollution) and natural (confinement, sea urchin grazing) stressors of the MA-LUSI were explored with Redundancy Analysis (RDA) using the CANOCO 5 software. Collinearity of natural and anthropogenic stressors was tested by Spearman rank correlations analysis using R 3.5.0 environment (R Core Team, 2020). All plots were designed using the "ggplot2" package (Wickham, 2016).

2.4 Results

2.4.1 Species frequency and percentage cover

At the transect scale, *Cystoseira compressa* was by far the most frequent species (57.8%) and, therefore, the species with broader distribution in the locations studied (Supplementary Table 2.5). While other frequent fucalean algae were *E. crinita* (37.61%) and *E. amentacea* (26.61%), *C. compressa* subsp. *pustulata* (2.75%), and *G. elegans* (0.92%) were the rarest recorded. At the transect scale, *E. barbatula* (38.84%), *C. compressa* (30.15%), and *C. corniculata* (29.39%) were the most abundant species in the studied locations (Supplementary Table 3.5). The least abundant were *G. elegans* (3.33%) and *C. compressa* subsp. *pustulata* (2.12%).

2.4.2 Spatial variability of species percentage cover across scales

PERMANOVA of fucalean species percentage cover showed statistically significant differences at all spatial scales ($p < 0.001$; Table 2.2). Figure 2.3 shows the mean percentage cover values of the fucalean species at different spatial scales. Based on the components of variance (Table 2.2B), the highest variability in percentage cover (51%) was observed at the scale of location, followed by a high variance component associated with residuals (25%). Lower components of variance were observed at forest (15%) and transect (9%) scales. The test of homogeneity of dispersion (PERMDISP) also confirmed that there was a significant difference in within location variance [F(5, 680) = 62.283; P(perm): 0.001; Supplementary Table 2.6].

Although locations were originally chosen randomly and so treated as a random factor in the analysis, it was nevertheless of interest to describe and discuss differences among locations. There were differences in species composition of the fucalean assemblages between the locations sampled (Supplementary Table 2.7). While *C. barbatula* and *C. corniculata* were found to be the species with the highest percentage cover in the Greek forests, they were absent in all other locations. In both Italian locations, *C. compressa* had the highest percentage cover, while *C. crinita* dominated in Sazan, Albania. The open coasts of Eastern Macedonia and Menorca were the two locations with the highest number (4) of coexisting species.

Table 2.2 Permutational Analysis of Variance (PERMANOVA) of fucalean species percentage cover between locations (random), forests (random, nested within locations), and transects (random, nested within forests and locations) (A), and the components of variance (B).

(A)	**df**	**SS**	**MS**	**Pseudo-F**	**P(per)**
Location	5	1.252E+06	2.504E+05	11.280	<0.001
Forest:location	20	4.1956E+05	20,978	5.1576	<0.001
Transect:forest:location	73	2.9261E+05	4008.3	3.569	<0.001
Residuals	587	6.5914E+05	1122.9		
Total	685	2.8558E+06			

(B)	**Component of variance**	**Sq. root**
Location	2310.5	48.068
Forest:location	696.86	26.398
Transect:forest:location	429.28	20.719
Residuals	1122.9	33.510

nMDS analyses on the Bray-Curtis similarity index indicated differences among locations, forests and transects (Figure 2.4). SIMPROF analysis identified statistically distinct subsets (groups) of species at forest and transect scales. At forestscale seven groups were identified (A-F; Figure 2.5A). SIMPER analysis results showed species contribution to the dissimilarity between significant SIMPROF groupings based on species-level percentage cover. The species principally responsible for the forests groups were *C. corniculata* (Group A), *G. barbata* and *C. foeniculacea* f. *tenuiramosa* (B), *E. barbatula* and *C. compressa* (C), *C. compressa* (D), *E. amentacea* and *E. crinita* (E), *E. crinita* (F). At transect scale eight groups were identified (A-H; Figure 2.5B). The species principally responsible for the transects groups were C. corniculata (Group A), *E. mediterranea* (B), *G. barbata* and *C. foeniculacea* f. *tenuiramosa* (C), *G. barbata* and *C. compressa* subsp. *pustulata* (D), *E. barbatula* (E), *C. compressa* (F), *E. amentacea* and *E. crinita* (G), and *E. crinita* (H).

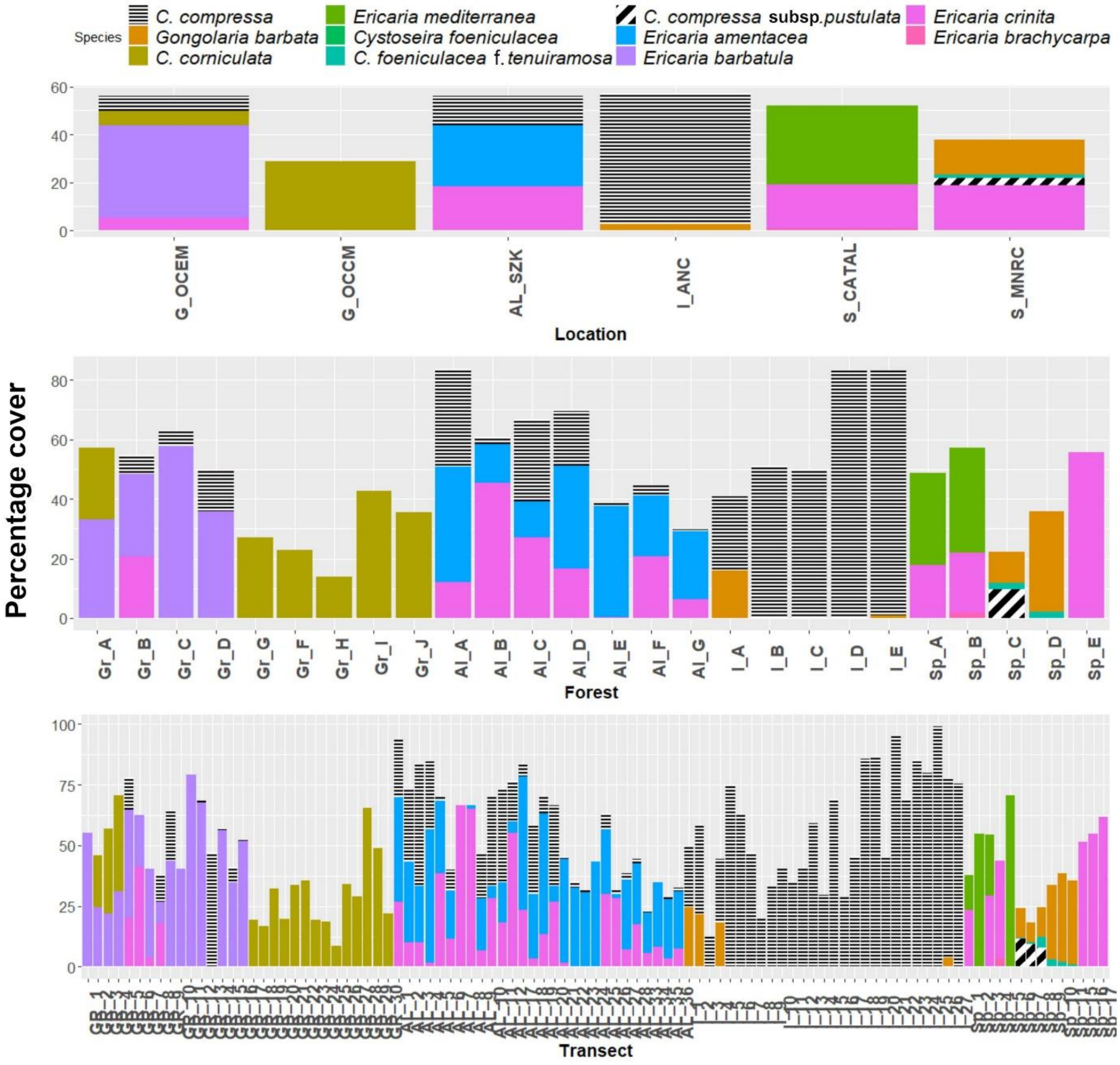


Figure 2.3 Mean percentage cover of eleven fucalean brown seaweed species at the three different spatial scales (transect, meadow, and location) studied. In comparison, the perennial fucalean are indicated by colour columns, the semi-perennials by black and white.

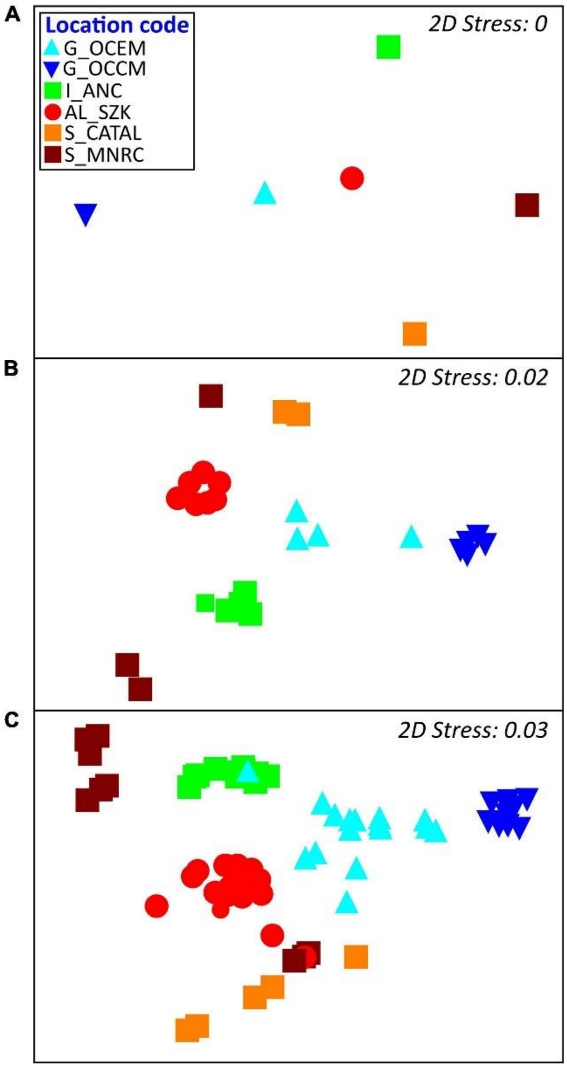


Figure 2.4 Non-metric multidimensional scaling (nMDS) analysis plot based on Bray-Curtis Index values showing the spatial similarities across locations (A), forests (B), and transects (C). For further information for the location codes see Figure 3.1. For the forest and transect codes see Supplementary Table 2.8.

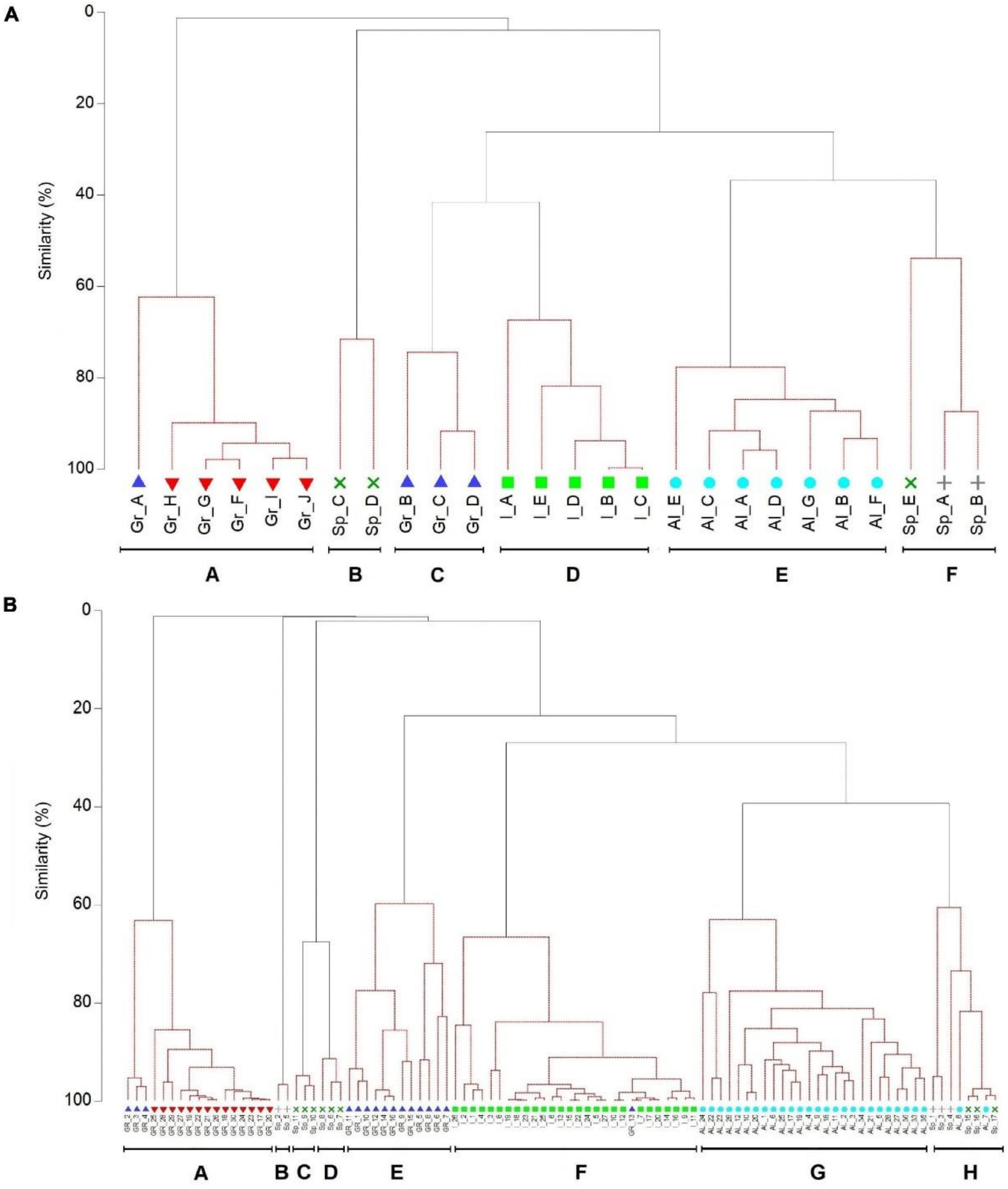


Figure 2.5 Dendrogram from group-average clustering of the fucalean species percentage cover at forest (A), transect (B) spatial scales. Continuous lines indicate the groups which were significantly differentiated by SIMPROF tests (at the 5% level). Within each of these groups, the null hypothesis that all pairs of species have the same association to each other cannot be rejected, the subgroup structure identified by cluster analysis thus having no statistical support (dashed lines).

2.4.3 Spatial relationship between the fucalean species

nMDS analysis based on fucalean percentage cover at the scale of 10 m transect revealed low species relationship in different transects across the Mediterranean Sea and therefore being heterogeneous, except *E. crinita* and *E. amentacea* (Figure 2.6).

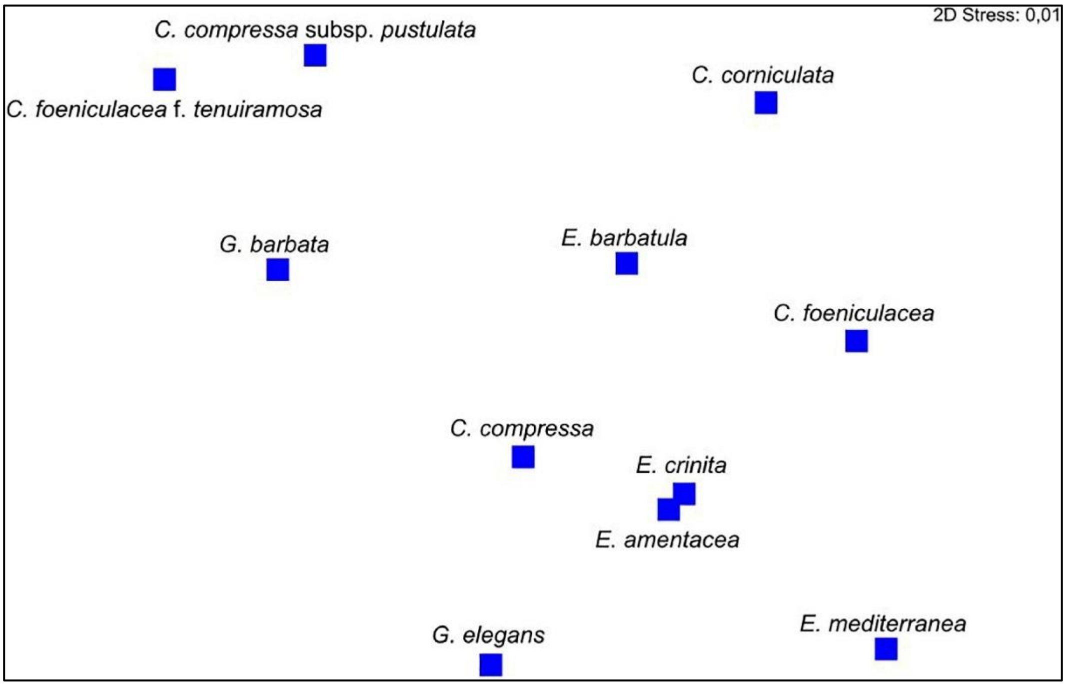


Figure 2.6 Non-metric multidimensional scaling (nMDS) analysis plot based on Bray-Curtis Index values showing the relationship at 10 m transects between the eleven fucalean species and subspecific taxa studied. C, *Cystoseira*; E, *Ericaria*; G, *Gongolaria*.

2.4.4 Anthropogenic and natural stressors metric relationship

The collinearity between the anthropogenic (urbanization, agriculture, mariculture, sewage discharge, harbor- related pollution) and natural (confinement, sea urchin grazing) stressors was weak ($\rho < 0.7$, Supplementary Table 2.10). The anthropogenic stressors identified in the studied forests in a decreasing rank were changes in coastline caused by the construction of harbors, by agriculture and by mariculture (Supplementary Table 2.9). According to MA-LUSI, three forests from the Conero Riviera in Italy (Grotta Azzurra, Passetto, and Passetto-Scalaccia) are featured by the highest anthropogenic stress (MA-LUSI > 4.5). The lowest anthropogenic stress was estimated for the forests of the Albanian coasts (0) and for two Open Eastern Macedonia coasts, Greece (1). The primary natural stress identified for the studied forests was sea urchins grazing, with its highest value estimated at the Grame forest in Albania (3) and the lowest in the Italian, Spanish and Turkish forests (0). Confinement values ranged from 0.75 (convex coastline) to 1.25 (concave coastline).

The RDA full model results showed that 72.5% of the response data variance were explained in the first two axes, while 93.5% in the first three axes. As shown in the ordination graph

(Figure 2.7A) the percentage cover of perennial fucalean species was negatively correlated mainly with mariculture, urbanization and harbors, as well as to MA-LUSI index. The same activities and stress showed a positive correlation with the percentage cover of *C. compressa*. Mariculture, urbanization, MA-LUSI and harbors also explained statistically ($p < 0.05$) the fucalean percentage cover variability (%, Table 2.3).

The RDA forward selection model results showed that only two of the explanatory variables added significantly to the explanatory power of the analysis (Figure 2.7B and Table 2.3). These were mariculture (73.6% to the explanatory power, $p = 0.001$) and urbanization (that added 11.1%, $p = 0.01$). The two ordination axes accounted for 61.4% of the total variation in the response data. A very clear pattern emerged in this analysis, with the percentage cover of semi-perennial fucalean species being positively correlated to both explanatory variables, while the perennial fucalean species showed a negative correlation.

Cystoseira compressa showed a different pattern in relation to anthropogenic stress compared to perennial species. This is illustrated in Figure 2.8, where the polynomial regressions between perennial and semi-perennial fucalean species and MA-LUSI are shown. While the perennial fucalean percentage cover was negatively affected by the increase of MA-LUSI ($R^2 = 0.61$), the percentage cover of *C. compressa* and *C. compressa* var. *pustulata* increased ($R^2 = 0.73$). In both cases, changes in percentage cover wasn't linear, with thresholds at values of MA-LUSI greater than about 1.5 and 2.5, respectively.

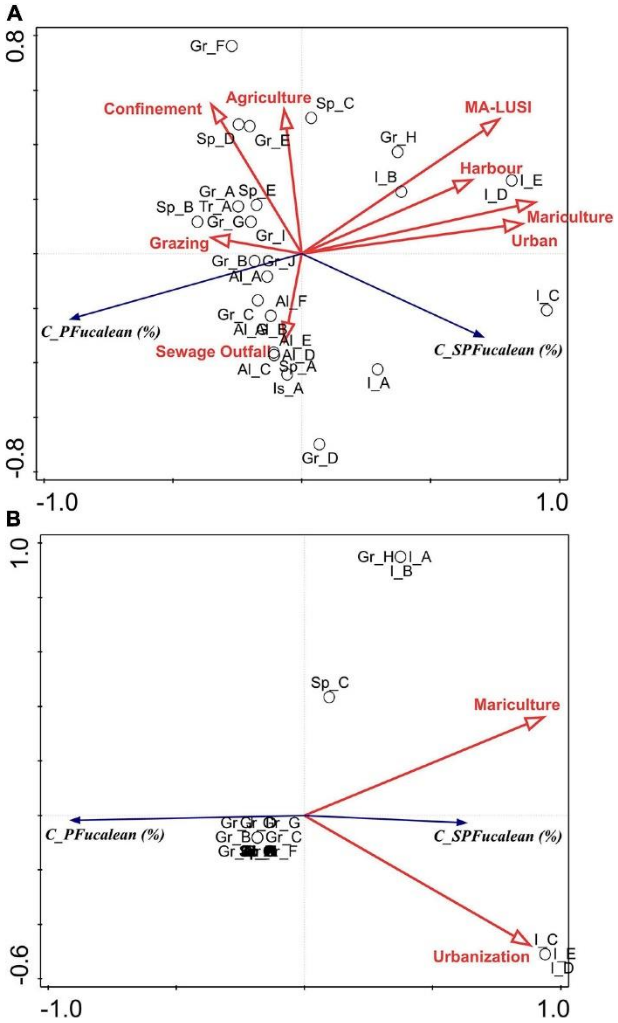


Figure 2.7 Graph plots of the full (A) and forward selection (B) models of redundancy analysis (RDA) between percentage cover of perennial fucalean species (C_PFucalean) and semi-perennial *C. compressa*, and *C. compressa* var. *pustulata* (C_SPFucalean) found in the sampled locations, and the main anthropogenic and biological (grazing) stress

Table 2.3 Simple term effect of explanatory variables of RDA full and forward selection models between the percentage cover of perennial fucalean seaweeds and semi-perennial *C. compressa* and *C. compressa* var. *pustulata* found in the sampled locations, and the main anthropogenic (urbanization, agriculture, mariculture, sewage discharge, harbour- related pollution) and natural (confinement, sea urchin grazing) stressors (for more information see Figure 2.6).

	Full model			**Forward selection model**			
Name	**Explains%**	**Pseudo-F**	***P***	**Explains%**	**Contribution%**	**Pseudo-F**	***P***
Mariculture	53.4	30.9	0.001	53.4	73.6	30.9	0.001
Urbanization	47.5	24.4	0.002	8	11.1	5.4	0.01
MA-LUSI	39.8	17.8	0.001				
Harbor	28.8	10.9	0.002				
Confinement	10.2	3.1	0.064				
Grazing	8	2.3	0.117				
Agriculture	2.4	0.7	0.483				
Sewage discharge	1.1	0.3	0.793				

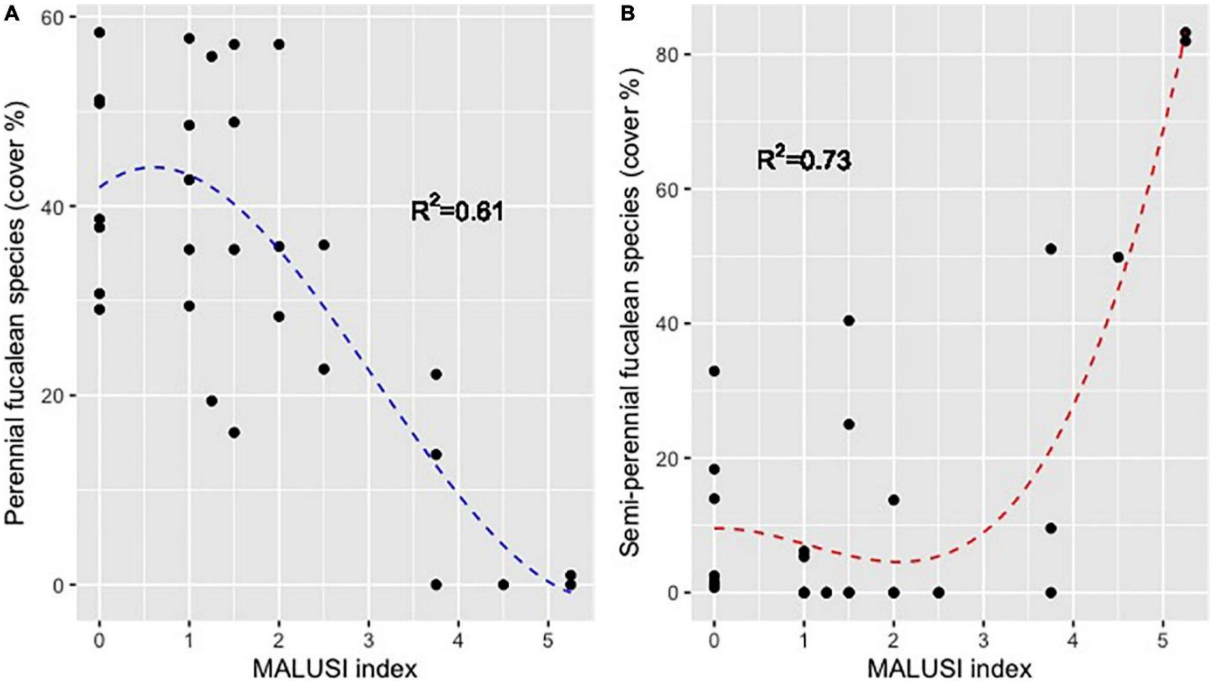


Figure 2.8 Polynomial regression between MALUSI and (A) percentage cover of perennial fucalean species (y = 0.73x3 − 6.81x2 + 7.39x + 41.96) and (B) the semi-perennial species *Cystoseira compressa*, *C. compressa* var. *pustulata* (y = 1.24x3 − 3.93x2 + 0.4x + 9.52).

2.5 Discussion

Even though this is not a manipulative experiment able to disentangle cause effects relationships, this large scale approach allowed to better identify the potential effects of anthropogenic and natural stressors on fucalean species percentage cover in the Mediterranean Sea. Sampling was carried out at eight locations, characterized by extensive rocky shores and more or less accessible from the coast. The forests were selected (see T–S species accumulation curves; Supplementary Figure 2.1; Ugland et al., 2003) along Mediterranean coasts, from Catalonian coasts in the West to the Aegean Sea in the East (approximately between 4 and 24◦E), and to cover a less extensive latitudinal gradient in the central Mediterranean (approximately between 44 and 40◦N). This approach enabled us to assess distinct spatial patterns in fucalean assemblages and correlate the percentage cover of *Cystoseira*, *Ericaria*, and *Gongolaria* species with key stressors considered the most important for the decline of fucalean forests in the Mediterranean (e.g., Fabbrizzi et al., 2020), either singly or in combination.

2.5.1 Relationship between species percentage cover and anthropogenic and natural stress

Spatial-scale-based mapping allowed to understand better the effects of anthropogenic and natural stressors on fucalean species percentage cover in the Mediterranean Sea. By examining the effects of spatial scales and main stressors singly or cumulatively on the percentage cover of *Cystoseira*, *Ericaria*, and *Gongolaria*, we could explore by correlative means the stressors that may cause species degradation.

Indeed, the results of this study indicated significant differences in the percentage cover of fucalean species at all spatial scales, with the highest variance detected at the scale of location, i.e., 10 s of kilometers (Figure 2.4 and Tables 2.2A,B). Variance at such a broad spatial scale is likely to be related to environmental factors such as local geomorphology, or chronic pollution, e.g., eutrophication (Benedetti-Cecchi et al., 2001; Sales and Ballesteros, 2009; Cefalì et al., 2016). This explanation is also supported by the multivariate analyses results (nMDS, SIMPROF, and SIMPER), where the sampled forests and transects, but not the locations, were sub-grouped based on different species percentage cover (Figures 2.5A,B). The criteria for this grouping were ecological, e.g., the group of open high hydrodynamic coasts inhabited by *C. corniculata* or of anthropogenic stressed coasts inhabited by *C. compressa*, as well as geographical, e.g., groups of a broadly distributed species, *G. barbata*, and groups of *C. foeniculacea* f. *tenuiramosa* or *C. compressa* subsp. *pustulata*, with a more restricted distribution. Similar patterns have also been observed in studies of other benthic macrophytes, where the functional metrics like species percentage cover reduced the spatial complexity and showed similarities of habitats with similar ecological conditions (Orfanidis et al., 2008, 2010). In *Posidonia oceanica* meadows, the highest variations of the standing crop (g dry biomass m^{-2}), a comparable metric with percentage cover used in this study, have also been observed at the largest spatial scale (10's kilometers apart). Such a result might reflect differences in the features of habitats in different localities, such as wave exposure, substrate type (rocky vs. pebble), sediment characteristics, and grazing pressure (Balestri et al., 2003). Therefore, the location spatial scale-specific management, also adopted by the European Water Framework Directive (2000/60/EC), i.e., the spatial scale on which the most significant variation at a habitat exists, is relevant to a wide range of present and future conservation and restoration actions.

Fucalean species exhibiting different functional traits, i.e., perennial vs. semi-perennial species, responded differently to anthropogenic and natural stressors. While the existence of mariculture, harbors, and the cumulative anthropogenic stress (MA-LUSI index) negatively correlated with the percentage cover of perennial species, the same stressors seem to produce favorable conditions, at least up to a certain intensity, i.e., MA-LUSI values between 2 and 5, for the growth of *C. compressa* and *C. compressa* subsp. *pustulata*. This result agrees with the well-known pattern of replacement of perennial fucalean species by the more tolerant, relatively fast-growing *C. compressa* under stressing conditions (Panayotidis et al., 2004; Airoldi and Beck, 2007; Devescovi and Iveša, 2007; Falace et al., 2010; Giakoumi et al., 2012; Kletou et al., 2018). Competitive release is likely to be a critical determinant of fucalean diversity and abundance when certain anthropogenic stressors limit the growth of perennial species, allowing the semi-perennials to expand and dominate in the community (Segre et al., 2016). However, although the specific mechanisms behind these changes have not been fully understood yet, habitat destruction and decrease in water quality are likely to play a major role in the decline of perennial species (Tsiamis et al., 2013; Thibaut et al., 2015; Iveša et al., 2016; Rindi et al., 2020). These processes are indicators of nutrient enhancement, water turbidity, and high eutrophication levels, which are invoked in several studies as the main causes for the regression of fucalean species (Sala et al., 2012; Mancuso et al., 2018) and seagrasses (Danovaro et al., 2020) in the Mediterranean Sea. The input of nutrients and changes in water transparency are considered among the processes affecting the growth of macrophyte communities (De Jonge et al., 2002; Viaroli et al., 2008). In a recent paper (Fabbrizzi et al., 2020), geomorphological features were recognized among the most relevant drivers predicting presence of fucalean seaweeds, followed by anthropogenic variables such as distance from ports and urbanization.

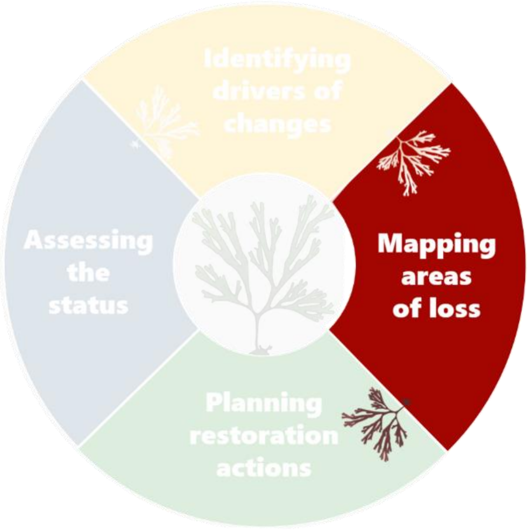
2.5.2 Species frequency and percentage cover

At the transect scale, *Cystoseira compressa* was by far the most frequent species. Variation in distribution may be caused by several factors like growth pattern, amount and dispersal of zygotes, and grazing (Falace et al., 2005; Mangialajo et al., 2012). In terms of growth pattern, fucalean seaweeds in the Mediterranean generally undergo a morphological shift from the period of main growth in late winter-early summer to dormancy in late summer-autumn, when many species shed a large part of their fronds (Orfanidis et al., 2017). Indeed, fronds of *C. compressa* undergo senescence and get detached in summer, after the alga has released the gametes (Sauvageau, 1912). In late summer and autumn this species usually consists of only a small, perennial holdfast and a few short flattened branches (the so-called rosetta form, Cormaci et al., 2012), which will issue new fronds in the subsequent winter. Zygotes of *C. compressa* tend to adhere to parental receptacles, remaining entrapped in a layer of mucilage formed on the surface of the alga (Sauvageau, 1912; personal observation). We suggest that this may be a strategy of *C. compressa* to resist strong hydrodynamic and unsuitable habitat conditions or to expand distribution by fragments of fronds floating far away from parental populations as reported for *Sargassum muticum* (Yendo) Fensholt (Deysher and Norton, 1981).

Knowing the abundance patterns of different species can provide insight into how a community or ecosystem functions and how the processes link the local abundance of a species and its regional distribution (Brown, 1984). The present study indicated a large variation locally in the mean percentage cover of fucalean seaweeds, confirming that assemblages formed by these algae are fragmented and heterogeneous across the Mediterranean Sea. Differences in species composition among the coasts have been suggested as a key biological feature of Mediterranean biogeography (Coll et al., 2010; Sales et al., 2012). However, since fucalean seaweeds unequivocally dominated communities characterized by good water quality with low anthropogenic stress, we argue that a severe local decline may be caused mainly by habitat destruction, decrease in water quality, and overgrazing by herbivores (Tsiamis et al., 2013; Thibaut et al., 2015; Iveša et al., 2016; de Caralt et al., 2020). This leads to replacement by relatively fast-growing species (e.g., *C. compressa*) or a shift to less-structured assemblages formed by morphologically simple algae (i.e., turf-forming, or other ephemeral, opportunistic seaweeds), mussels, or barren grounds (Airoldi and Beck, 2007; Devescovi and Iveša, 2007; Falace et al., 2010; Giakoumi et al., 2012; Sala et al., 2012; Kletou et al., 2018).

2.6 Conclusion

The results of this study on the genera of *Cystoseira*, *Ericaria*, and *Gongolaria* in the Mediterranean Sea provided new insights into: (a) the role of different anthropogenic and natural stressors, which can individually or cumulatively affect these algal forests, (b) the differential responses of species belonging in different trait (functional) groups, i.e., perennial vs. semi-perennial species. However, experimental studies are additionally required to mechanistically identify the drivers for the observed replacement of perennial fucalean species by relatively fast-growing semi-perennial species, e.g., *C. compressa*, or the general seaweed regression along the Mediterranean Sea. Understanding the effect of multiple stressors is particularly challenging because their potential cumulative effects on these habitats cannot be predicted in a single-stressor framework.



CHAPTER 3

CAN WE PRESERVE AND RESTORE OVERLOOKED MACROALGAL FORESTS?

CHAPTER 3

Can we preserve and restore overlooked macroalgal forests?

3.1 Abstract

Habitat degradation and loss are severely affecting macroalgal forests worldwide, and their successful mitigation depends on the identification of the drivers of loss and the implementation of effective conservation and restoration actions.

We made an extensive literature review 1- to document the historical (1789–1999) and recent (2000−2020) occurrence of the genus *Cystoseira*, *Ericaria* and *Gongolaria* reported in the literature along the 8000 km of the coasts of Italy, 2- to assess their decline and patterns of extinction, 3- to ascertain the drivers responsible for these changes, 4- to highlight the existence of success stories in their conservation and natural recovery. In the last twenty years, overall information on the distribution of *Cystoseira s.l.* exponentially increased, although research focused almost exclusively on intertidal reefs. Despite the lack of systematic monitoring programs, the local extinction of 371 populations of 19 different species of *Cystoseira s.l.* was documented across several regions, since 2000. Coastal engineering and poor quality of waters due to urban, agricultural or industrial activities were often documented as leading causes of habitat loss. However, the drivers of extinction were actually unknown for the majority of the populations and cause-effects relationships are scarcely documented. Although the proportion of protected populations increased to 77.8%, Marine Protected Areas are unlikely to guarantee adequate conservation efficacy, possibly also for the widespread lack of management and monitoring plans dealing specifically with *Cystoseira s.l.* species, and few evidences of natural recovery were observed.

Our review shows the dramatic lack of baseline information for macroalgal forests, highlighting the urgent need for the monitoring of less accessible habitats, the collection of long-term data to unveil drivers of loss, and an up-dated reporting about the conservation status of the species of interest to plan future interventions.

Keywords: *Cystoseira*, Spatial distribution, Conservation, Restoration, Marine Protected Areas

3.2 Introduction

Habitat degradation and destruction are recognized among the most serious threats to biodiversity and functioning of ecosystems, both in terrestrial (Hoekstra et al., 2005) and marine realms (Crain et al., 2009). Coastal marine ecosystems are particularly affected by this global phenomenon, due to increasing population density and multitude of human activities disturbing these systems (Micheli et al., 2013; Airoldi et al., 2020). Some habitats are entirely lost as a direct effect of coastal engineering, while others are exposed to the compound effect of multiple stressors, driving the shift from complex, diverse habitats to simpler and less productive ones (Airoldi et al., 2008; Claudet and Fraschetti, 2010).

Despite the efforts to contrast habitat destruction through the mitigation of human pressures or the implementation of conservation measures, the natural recovery of coastal habitats is rare, even when the proximate drivers of loss are removed (Lotze et al., 2011; Colletti et al., 2020). The onset of feedback mechanisms favors the persistence of degraded habitats and impairs the recovery ability of the system (Scheffer et al., 2001). In addition, recovery potential may be compromised by a scarce connectivity among degraded and healthy ecosystems, acting as vital sources of propagules (Duarte et al., 2013).

In this scenario, restoration is growingly acknowledged as a convenient strategy to actively trigger or accelerate the recovery of degraded coastal habitats (Abelson et al., 2020), as also recognized by the recently announced UN Decade on Ecosystem Restoration (2021−2030). Although restoration is less advanced in marine compared to terrestrial ecosystems, significant progress has been made for several coastal habitats, including seagrasses, saltmarshes, oyster reefs, mangroves, kelp forests, and coral reefs (Bayraktarov et al., 2016).

Yet, to make restoration interventions consistently successful, a detailed knowledge of present and past distribution of lost habitats, the individuation of donor populations, and the identification of the stressors that caused their decline or disappearance together with the evaluation of their mitigation state, should be considered prerequisites to select putative restoration sites (Gann et al., 2019). Habitat formers are common targets of marine ecosystem restoration, intended not only to reverse species' local decline, but also to improve and provide habitat for other species of commercial value. However, recent analyses show that only a minority of the restored marine species for conservation purposes are actually of international conservation concern (Swan et al., 2016), and baseline knowledge is generally very limited to plan cost effective restoration interventions (Bayraktarov et al., 2016). Macroalgal forests formed by fucalean algae (*Cystoseira* sensu latu, including the genera *Cystoseira*, *Ericaria* and *Gongolaria*; Molinari and Guiry, 2020) are critical habitats of intertidal and subtidal reefs in the Mediterranean Sea (Sala et al., 2012) and an excellent case study to show gaps strongly limiting restoration interventions. They host highly diverse assemblages, providing food and shelter for associated organisms and representing privileged nursery habitats for fish assemblages of commercial interest (Cheminée et al., 2013, 2017). Their presence enhances coastal primary productivity, and their primary role in maintaining high biodiversity and the functioning of rocky habitats have long been recognized. As a consequence, several species (*C. sedoides*, *E. amentacea* var. *stricta*, *E. mediterranea*, *E. zosteroides*, *G. montagnei*) have been protected since 1982, with the enforcement of the Bern Convention (1979). In 2009, an amendment of the Mediterranean Action Plan (Annex IV, SPA/BD Protocol - United Nations Environment Programme) adopted within the framework of the Barcelona Convention (1976), identified the conservation of all but one (*C. compressa*) Mediterranean *Cystoseira s.l.* species as a priority. Despite the robust legislative framework, specific conservation measures for the protection of these habitat-forming species have never been implemented (Fraschetti et al., 2011). For example, the selection of marine sites deserving protection under the Natura 2000 Sites network[1] in the Mediterranean is generally based on the presence of *Posidonia oceanica* meadows, while the presence of *Cystoseira s.l.* is only incidental, as brown macroalgal forests are not listed in the Habitat Directive annexes (Directive 92/43 EEC).

Cystoseira s.l. are highly vulnerable to several human disturbances, which caused their regression in many regions of the Mediterranean Sea (Thibaut et al., 2005, 2014; Airoldi and Beck, 2007), and their natural recovery has rarely been observed (Perkol-Finkel and Airoldi, 2010). Restoration has been proposed as a promising approach to halt their decline (Gianni et al., 2013), and several European projects (e.g., MERCES[2], and AFRIMED[3]) developed and

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[1] https://ec.europa.eu/environment/nature/natura2000/index_en.htm
[2] http://www.merces-project.eu/
[3] http://afrimed-project. eu/

tested new techniques to reintroduce *Cystoseira s. l.* species after local extinction, regenerating self-sustaining populations (Verdura et al., 2018).

At present, despite the interest of the European Community in the conservation and restoration of macroalgal forests and the growing attention of the scientific community, most of these questions risk to be hardly addressed for most Mediterranean regions. Here, we assess knowledge and gaps challenging the restoration of macroalgal forests with an extensive literature review that dates back to 1789, using Italian shores as a case study. The aims are i) documenting the historical and recent occurrence of *Cystoseira s.l.* species reported in the literature along the coasts of Italy; ii) assessing the current knowledge on the extent of decline and extinction of *Cystoseira s.l.* populations and on the drivers responsible for these changes; iii) exploring whether the present network of protected areas is efficiently contributing to the conservation of brown macroalgal forests.

3.3 Methods

Data on the historical and current distribution of brown fucalean forests in Italian coastal waters were collected from: 1) published literature, 2) grey literature, 3) monitoring programs, 4) unpublished data from experts and ongoing projects (e.g., AFRIMED). Herbarium specimens were included only when reported in published and grey literature.

The research of published literature was conducted using two databases (ISI Web of Science and Scopus) for the 1985–2020 time-frame. The systematic literature screening was carried out by searching in the “Title,” “Abstract,” and “Keyword” fields the following combination of terms: (“*Cystoseira*” OR “*Cystoseira* canopies” OR “Fucales” OR “brown algae” OR “macroalgal forest*” OR “habitat form*”) AND (“distribution” OR “occurrence” OR “presence” OR “shift” OR “habitat loss” OR “decline”) AND “Mediterranean”. Grey literature, dating back to 1883, included publications on national journals edited by national associations or institutions (e.g., Italian Society of Marine Biology, Italian Botanical Society, Gioenia Academy of Catania), books, unpublished Ph.D. theses and conference proceedings. We also searched the citation lists of the selected articles for further publications of interest. In addition, nearly 670 records of *Cystoseira* spp. were acquired from the monitoring program CARLIT (CARtography of LITtoral and upper-sublittoral benthic communities, Ballesteros et al., 2007).

The full list of publications included in the analysis is reported in the Supplementary material. In terms of algal classification and nomenclature, we followed the current taxonomic arrangement of AlgaeBase (Guiry and Guiry, 2021).

We collected all the information about the georeferenced occurrence of *Cystoseira s.l.* populations along Italian coasts and classified it according to the region and basin (Supplementary Fig. S3.1). The geographic location of brown fucalean populations was digitized as shapefile points or polylines in order to be associated with a map, using the Open Source QGIS software (QGIS Development Team, 2018). In addition, for each geographical record, we noted the identity and number of *Cystoseira s. l.* species observed, the sampling method adopted (e.g., visual estimate, destructive sample, herbarium specimens), the year, season, and depth of observation, the extent of the population (when available, expressed as linear coastal extent or mapped area), the eventual certification of the disappearance or decline of a population and the drivers advocated as causes of decline. The localization of each population (encoded by single or multiple points or polylines, according to the source data) inside areas characterized by different protection regimes (i.e., national parks, natural marine reserves, underwater parks, Marine Protected Areas - MPA, Natura 2000 Sites of Community Importance - SCI or Specially protected areas - SPA, established on the Habitats Directive, Specially Protected Areas of Mediterranean Importance - SPAMI, defined in the Barcelona Convention) at the time of the biological sampling was assessed by using a modified shapefile from MAPAMED (2017)[4], a database on sites of interest for the conservation of marine environment in the Mediterranean Sea.

Results of the review are reported in Figs. 3.1–3.7, Supplementary Figs. S3.1–S3.8, and summarized in Tables 3.1, 3.2. We selected the 2000 as the break year between historical and recent assessments, as the decline and loss of brown fucalean forests has been reported in the literature since the 2000s, and local extinctions were mostly documented around 2000 (27.6% and 65% respectively in the '90s and 2000s).

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[4] https://medpan.org/main_activities/mapamed/

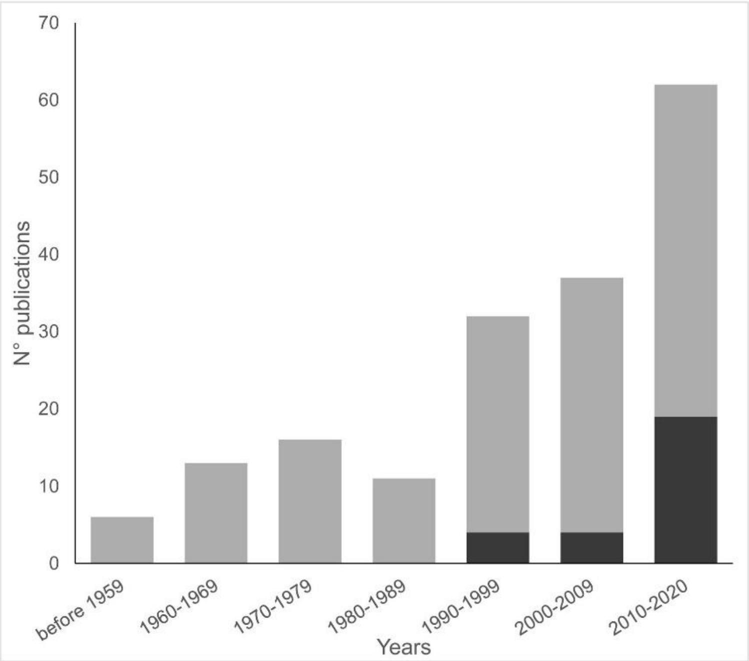


Figure 3.1 Temporal trend of publications including georeferenced data on *Cystoseira s.l.* species distribution. Grey bars represent descriptive studies, black bars show experimental studies.

3.4 Results

3.4.1 Research of temporal trends and focus of interest

The review included 169 articles, books, and PhD theses. The number of publications reporting information on the distribution of *Cystoseira s.l.* increased during the '60s, showing an exponential growth in the last three decades, when experimental studies added to descriptive ones (Fig. 3.1). Before 1990, all published articles were represented by floristic/phytosociological studies reporting the description of algal assemblages in different regions, or taxonomic studies including morphological descriptions of the species (De Toni, 1895; Gerloff and Nizamuddin, 1976; Battiato et al., 1979; Giaccone, 1985; Cormaci and Furnari, 1988). Few studies focused on the description of assemblages associated to *Cystoseira s.l.* (Campisi et al., 1973; Pastore, 1981), and only two studies related the local regression of *Cystoseira s.l.* to anthropogenic impacts (i.e., eutrophication, water turbidity, industrial pollution, urbanization; Giaccone, 1974; Sfriso, 1987).

During the ‘90s, the literature was still dominated by phytosociological and taxonomic studies (e.g., Alongi et al., 1999a), although a growing interest was dedicated to phenological studies (e.g., Benedetti-Cecchi and Cinelli, 1993; Alongi et al., 1999b; Verlaque et al., 1999) and to the investigation of ecological interactions with the extant assemblage (Benedetti-Cecchi and Cinelli, 1992a,b, 1995; Benedetti-Cecchi et al., 1996). In recent years, *Cystoseira s.l.* was also the focus of new research fields, including ecotoxicology (Conti et al., 2010; Renzi et al., 2011; Conti et al., 2015), genetics (Buonomo, 2017; Buonomo et al., 2017), and microbiology of associated bacterial communities (Mancuso et al., 2016; Buonomo, 2017).

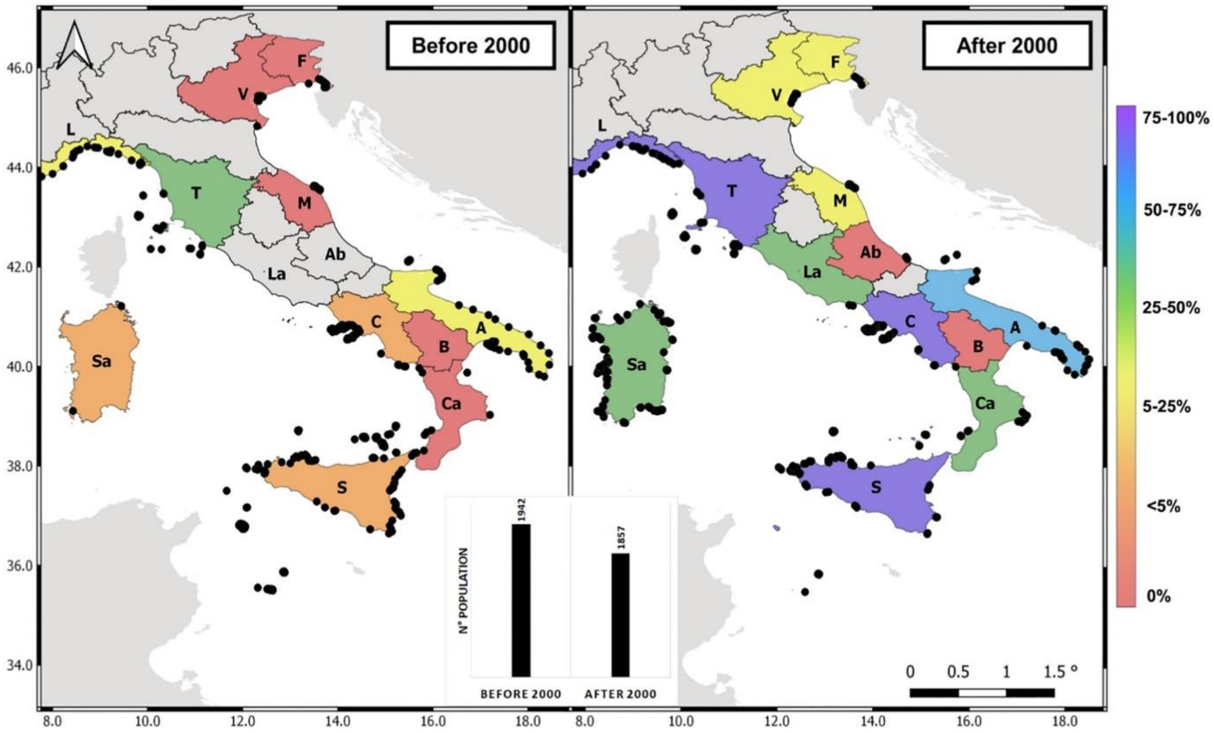


Figure 3.2 Historical and recent distribution of *Cystoseira s.l.* along Italian coasts. Black dots and lines indicate the presence of the species. The color of each region represents the percentage of populations protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region, grey regions indicate no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.

3.4.2 Human threats on macroalgal forests

Since the 2000s, brown algal forests have been perceived as habitats threatened by humans: researchers investigated patterns of local extinction (Curiel et al., 2001; Catra et al., 2006; Serio et al., 2006) and the major drivers of decline (e.g., urbanization, Benedetti-Cecchi et al., 2001; Mangialajo et al., 2008; human trampling, Milazzo et al., 2002, 2004; climate change, Schiaparelli et al., 2007; water pollution, Drago et al., 2004). In the last decade, efforts concentrated on investigating global and local drivers of forests decline (e.g., Porzio et al., 2011; Baggini, 2014; Mancuso, 2016; Buosi and Sfriso, 2017; Mancuso et al., 2018), among which overgrazing by herbivores emerged as a relevant threat (e.g., Agnetta et al., 2015; Gianni, 2016; Ferrario et al., 2016; Piazzi and Ceccherelli, 2019; Tamburello et al., 2019).

3.4.3 Protection and restoration of macroalgal forests

The efficacy of MPAs in preserving forests (Mangialajo et al., 2004; Ceccherelli et al., 2005; Cecere et al., 2005) and the effect of their loss for associated assemblages (Benedetti-Cecchi et al., 2001; Maggi et al., 2009) have been also studied since the 2000s. The role of *Cystoseira s.l.* as habitat formers was examined for several faunal groups (e.g., hydroids, Fraschetti et al., 2006; molluscs, Milazzo et al., 2000; Chemello and Milazzo, 2002; Gianguzza et al., 2005; Chiarore et al., 2019; fish assemblages, Fiorin et al., 2008; Riccato et al., 2008; Cheminée et al., 2013). In addition to habitat protection inside MPAs (Fraschetti et al., 2012; Sala et al., 2012; Gianni, 2016; Guarnieri et al., 2016) restoration was presented as a new, promising approach to contrast the loss of macroalgal forests (PerkolFinkel and Airoldi, 2010; Perkol-Finkel et al., 2012; Ferrario, 2013; Gianni, 2016; Gianni and Mangialajo, 2016).

3.4.4 A zoom at species level: historical and recent distribution of *Cystoseira s.l.* species along the coasts of Italy

The historical (1789–1999) and recent (2000–2020) distribution of 11 species of the genus *Cystoseira*, 8 species of the genus *Ericaria*, and 7 species of the genus *Gongolaria* are reported in Table 3.1 and Figs. 3.3–3.9, S3.2–S3.8. The subdivision of Italian seas into sectors adopted in the present study is reported in the Supplementary material (Fig. S3.1). Overall, 674 records have been listed for the historical period and 3238 for recent years. Populations occurring in intertidal and subtidal habitats were almost equally frequent in historical data (respectively 487 and 468 observations), while the majority of information was on the intertidal habitat after 2000 (respectively 1301 and 353 observations). Since the 2000s, the local extinction of 371 populations of 19 different species of *Cystoseira s.l.* was documented across 8 Italian regions. Yet, drivers of extinctions were largely unknown for many populations (108 out of 371, Fig. 3.10). Fig. 3.2 reports the percentage of *Cystoseira s.l.* populations protected by MPAs, National and Underwater Parks, Natura 2000 Sites and SPAMIs in each region. Only 51 out of the 1942 populations reported by historical data were protected, while populations protected after 2000 were 1445 out of 1857, concurrently with an increase of protected coastline from 3840 km to 5025 km in the last two decades. In Table 3.2, the presence of *Cystoseira s.l.* inside MPAs, National and Underwater Parks is reported, specifying whether it has been documented before or after the institution of a protected area.

3.4.4.1 *Cystoseira compressa* (Esper) Gerloff & Nizamuddin - intertidal, subtidal

Historical data reported 303 populations in 9 Italian regions across all seas (Fig. 3.3, Table 3.1). At the time of sampling, 5.3% of the populations were protected by MPAs, National Parks and Natura 2000 SCIs (Tables 3.1, 3.2).

After 2000, 721 populations were reported across all seas in 12 regions. The majority of the populations were located inside protected areas under different regulations: 49.9% were protected by MPAs or national parks, while 32.9% were located inside Natura 2000 Sites and SPAMIs. The local extinction of 39 populations has been documented in 1979–1984 in Veneto (Lido island) as a consequence of eutrophication (Sfriso, 1987) and in 2013–2016 in Campania (Gulf of Naples, Procida, Ischia and Capri). A reduction in nutrient loads of the Venice lagoon favored the natural recovery of populations located in Lido island by 2006–2008 (Sfriso and Facca, 2011).

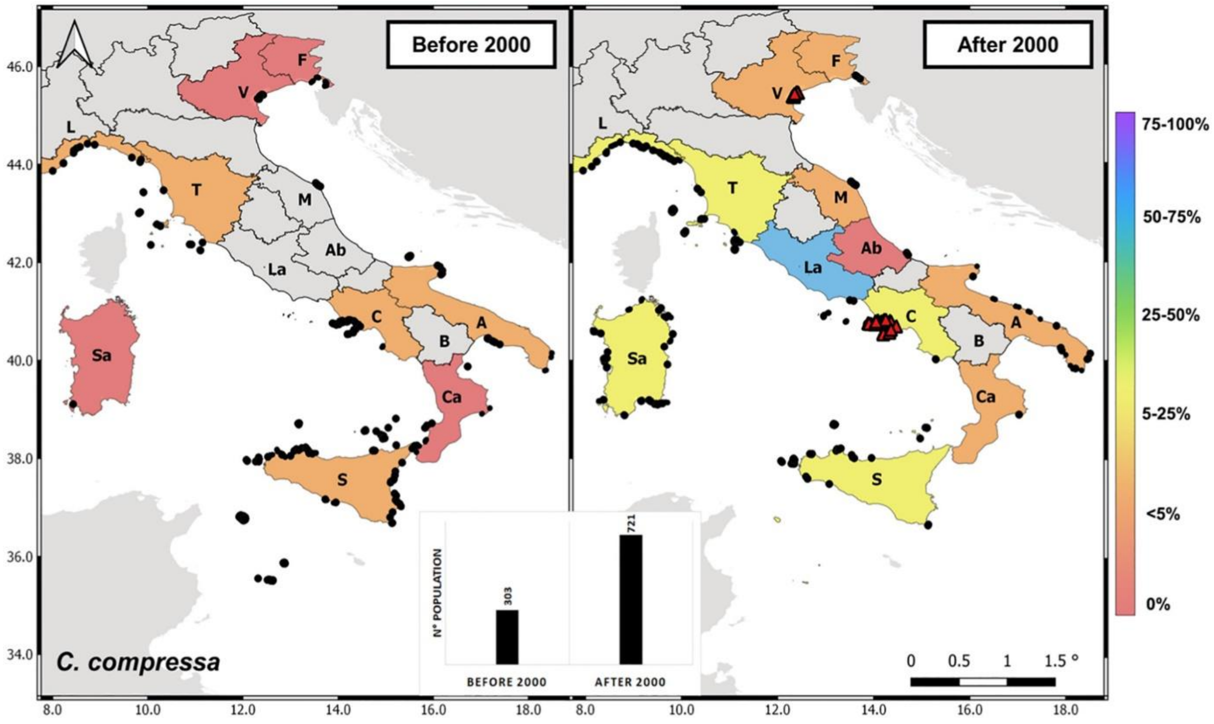


Figure 3.3 Historical and recent distribution of *Cystoseira compressa* along Italian coasts. Black dots and lines indicate the presence of the species, red triangles show documented local extinctions. The color of each region represents the percentage of populations protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region, grey regions indicate no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.

3.4.4.2 *Cystoseira corniculata* (Turner) Zanardini – subtidal

The species was observed for the first time in 1889 in Sardinia. During the '60s and '70, 17 populations were recorded along the coasts of the Northern Ionian Sea (Capo Rizzuto in Calabria and Santa Maria di Leuca in Apulia) and in the Central Adriatic Sea (Gargano Promontory, Tremiti islands) (Fig. S3.2, Table 3.1). None of them was included inside protected areas. After 2000, no populations of the species have been observed, and its local extinction has been documented in the Gargano Promontory (Apulia) in 1997, although no specific cause of disappearance has been identified (Cecere et al., 2000).

3.4.4.3 *Cystoseira crinitophylla* Ercegovic – intertidal, subtidal

Between 1958 and 1997, 16 populations were recorded in several locations of the Sicilian region, and in the central Adriatic Sea at the Tremiti islands (Apulia) (Fig. S3.2, Table 3.1). Only the population censused in 1997 was protected by the MPA of the Tremiti islands (Table 3.2). No populations of the species have been censused after 2000, and its local extinction in Linosa was documented in 1999. The disappearance of *C. crinitophylla* and of several other *Cystoseira s.l.* species (*C. foeniculacea*, *C. humilis*, *E. brachycarpa*, *E. zosteroides*, *G. elegans*, and *G. sauvageauana*) from the island was attributed to sea water warming, in the absence of other local stressors, such as water turbidity, eutrophication, overgrazing by sea urchins, fishing activities or anchoring (Serio et al., 2006).

3.4.4.4 *Cystoseira dubia* Valiante – subtidal

Early records of *C. dubia* along Italian coasts date back to 1880–1930, when seven populations were censused in the Gulf of Naples (Campania) and on the Amendolara sea-mount (Calabria) (Fig. S3.2, Table 3.1). Between the '70s and '80s, ten further populations were recorded in Campania, Sicily, and the presence of the species on the Amendolara sea-mount (Calabria) was confirmed in 1982. None of the 17 populations recorded was protected. Two populations of the species were individuated after 2000 in the Sicilian MPA of Capo Gallo – Isola delle Femmine (Table 3.2), while the re-examination of algal assemblages in the Gulf of Naples (2013–2016) and on Favignana island in the Aegadian Archipelago (2001) documented the local disappearance of 5 populations in the last twenty years.

3.4.4.5 *Cystoseira foeniculacea* (Linnaeus) Greville - intertidal, subtidal

Historical data reported 197 populations in 10 Italian regions across all seas except the Northern Tyrrhenian (Fig. 3.4, Table 3.1). No populations were protected at the time of observation.

After 2000, only 32 populations were recorded in Tuscany, Sardinia, Campania, and Sicily. 46.9% of them were protected by MPAs (Capo Caccia – Isola Piana, Regno di Nettuno, Isole Egadi, Capo Gallo – Isola delle Femmine) and Natura 2000 Sites and SPAs (Table 3.2). The local extinction of 38 populations has been documented in 1999–2001 in Sicily (Linosa island and several locations in the Aegadian islands), in 2013–2016 in Campania (Gulf of Naples, Procida), and in 2020 along the Conero Riviera (Marche).

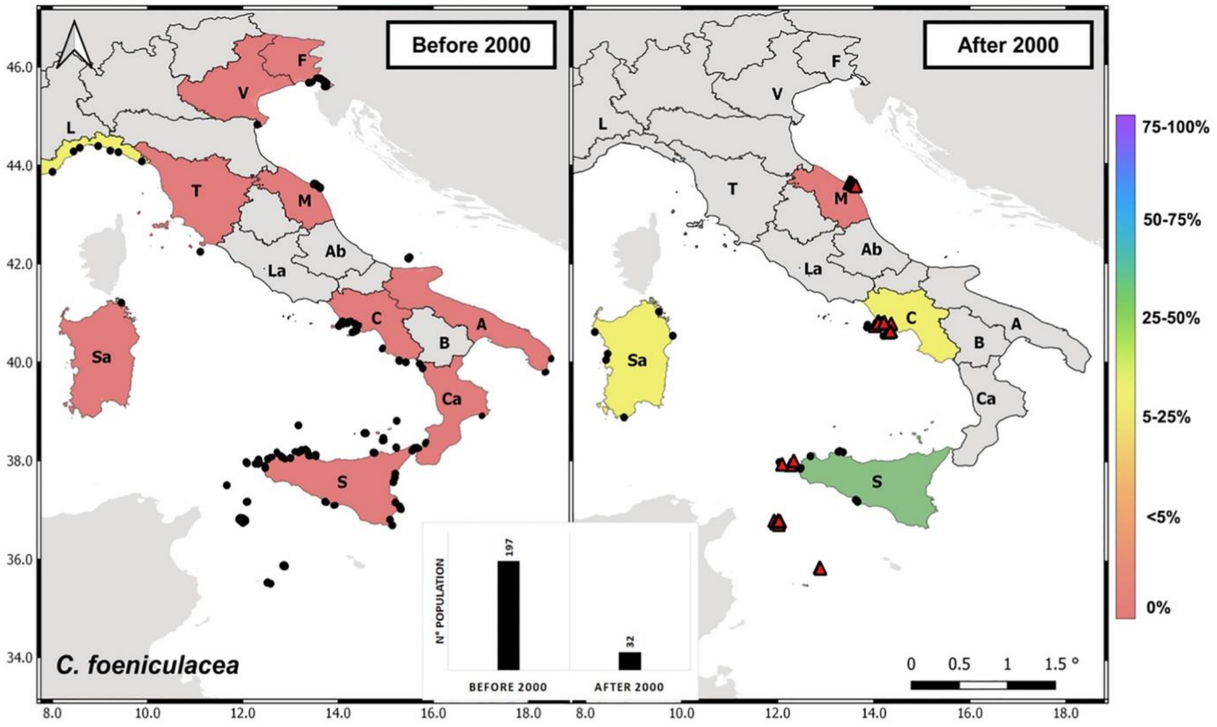


Figure 3.4 Historical and recent distribution of *Cystoseira foeniculacea* along Italian coasts. Black dots and lines indicate the presence of the species, red triangles show documented local extinctions. The color of each region represents the percentage of populations protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region, grey regions indicate no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.

3.4.4.6 *Cystoseira humilis* Schousboe ex Kutzing - intertidal, subtidal

Between the '40s and the '90s, 52 populations of the species were recorded along the Conero Riviera (Marche) and in all seas surrounding Sicily and its archipelagos, including Pantelleria, Aeolian, Aegadian and Pelagian islands (Fig. S3.3, Table 3.1). Between 1994 and 1997, one population of *C. humilis* was recorded in Tuscany. None of the populations was located within protected zones.

After 2000, only five populations have been described inside the MPAs of Capo Gallo and in the Aegadian islands, and the presence of the species in Tuscany was confirmed in 2018 (Table 3.2). The local disappearance of the species from Pelagian islands was documented in 1999.

3.4.4.7 *Cystoseira hyblaea* Giaccone – intertidal

The species was described only in 1985 at Punta d'Aliga (Ragusa), in southern Sicily (Giaccone, 1985; Giaccone et al., 1985) (Fig. S3.3, Table 3.1). *C. hyblaea* was not protected, nor was its presence documented in recent years.

3.4.4.8 *Cystoseira pelagosae* Ercegovic – subtidal

Only two populations of the species were described in 1985 in Sicily and Ustica island (Giaccone et al., 1985) (Fig. S3.3, Table 3.1). The species was not protected, nor was its presence documented in recent years.

3.4.4.9 *Cystoseira platyclada* Sauvageau – subtidal

Between 1971 and 1985, 17 populations of the species were recorded in the Strait of Sicily in Pantelleria, Aegadian and Pelagian islands and on shallow rocky banks (Pantelleria, Talbot) (Fig. S3.4, Table 3.1). None of the populations was protected, and their presence has never been reassessed after 2000. In 2001, the local disappearance of the species from Favignana island was documented.

3.4.4.10 *Cystoseira schiffneri* Hamel – subtidal

Between 1923 and 1999, 27 populations were recorded in the Sicilian islands of the Southern Tyrrhenian Sea (Ustica and all Aeolian islands) and in Apulia (Cheradi islands, Tremiti islands and the Gargano promontory) (Fig. S3.4, Table 3.1). A single population in the MPA of Ustica was protected at the time of observation (Table 3.2). The local extinction of the populations of Filicudi and Cheradi islands was documented respectively in 1991 and 1992, while the species was not censused any longer along the Gargano promontory or in Tremiti islands in 1997. In the Cheradi islands, the degradation of algal assemblages and the local disappearance of *C. schiffneri* and several other *Cystoseira s.l.* species (*E. amentacea*, *E. crinita*, *G. montagnei*, *G. sauvageauana*) were attributed to pollution and illegal fishing of the date mussel Litophaga litophaga (Cecere et al., 1996; Colletti et al., 2020). In the Tremiti islands, the disappearance of the species and of *E. crinita* and *G. montagnei* was attributed to water turbidity due to terrigenous sediment, possibly associated to the presence of pollutants (Cormaci et al., 2001). No records of the species were provided after 2000.

3.4.4.11 *Cystoseira sedoides* (Desfontaines) C. Agardh - intertidal, subtidal

The species was recorded along the coasts of Pantelleria island (Sicily) in 1970–1971, and its distribution along the island coasts was confirmed by studies in 1985 and 1999 (Fig. S3.4, Table 3.1). None of the 19 populations of the species was protected, and no information on its distribution in recent years is available.

3.4.4.12 *Ericaria amentacea* (C. Agardh) Molinari & Guiry – intertidal

Historical data reported the distribution of 271 populations in 9 regions and across all Italian seas (Fig. 3.5, Table 3.1). 8.9% of the populations were located inside protected areas (the MPAs of Portofino, Punta Campanella, Ciclopi islands, Tremiti islands, and Porto Cesareo, the national park of the Tuscan Archipelago, and several Natura 2000 SCIs) (Table 3.2).

After 2000, 616 populations were recorded in 8 regions across all seas except the Northern Adriatic. Among them, 75% of the populations were protected: 49.8% were located inside MPAs, national and underwater parks, while 25.2% were located inside SPAMIs, and Natura 2000 – SCIs. The local disappearance of 27 populations has been reported by the end of the 19th century in Liguria, in 1992 in Apulia (Cheradi islands), and in 2013–2016 in Campania.

3.4.4.13 *Ericaria barbatula* (Kutzing) Molinari & Guiry - intertidal, subtidal

During the ‘90s, 5 populations of the species were recorded in the Strait of Sicily, on the islands of Pantelleria and Lampedusa (Fig. S3.5, Table 3.1). In 2002, the presence of the species was confirmed in a single location of Lampedusa island. None of the populations was protected at the time of observation.

3.4.4.14 *Ericaria brachycarpa* (J. Agardh) Molinari & Guiry - intertidal, subtidal

Historical data reported 107 populations of *E. brachycarpa* distributed in 6 regions along the Ligurian, Tyrrhenian, Sardinian, Ionian Sea and the Strait of Sicily (Fig. 3.6, Table 3.1). At the time of observation, only 5 populations were protected by the MPAs of Ustica and Ciclopi islands, in Sicily (Table 3.2).

After 2000, 163 populations were recorded in 4 regions along the Ligurian, Tyrrhenian, Sardinian, Southern Ionian Sea and the Strait of Sicily. The majority of them (88.3%) were protected by 11 MPAs, National Parks (Tuscan Archipelago, Maddalena Archipelago), Natura 2000 sites, and SPAMIs. Between 1994 and 2016, the local disappearance of 17 populations was documented in Campania (Gulf of Naples, Ischia) and Sicily (mainland and Linosa). The extinction of *E. brachycarpa* forests from the eastern coasts of Sicily was attributed to increase in water turbidity, sediment deposition and overgrazing by sea urchins (Catra et al., 2019).

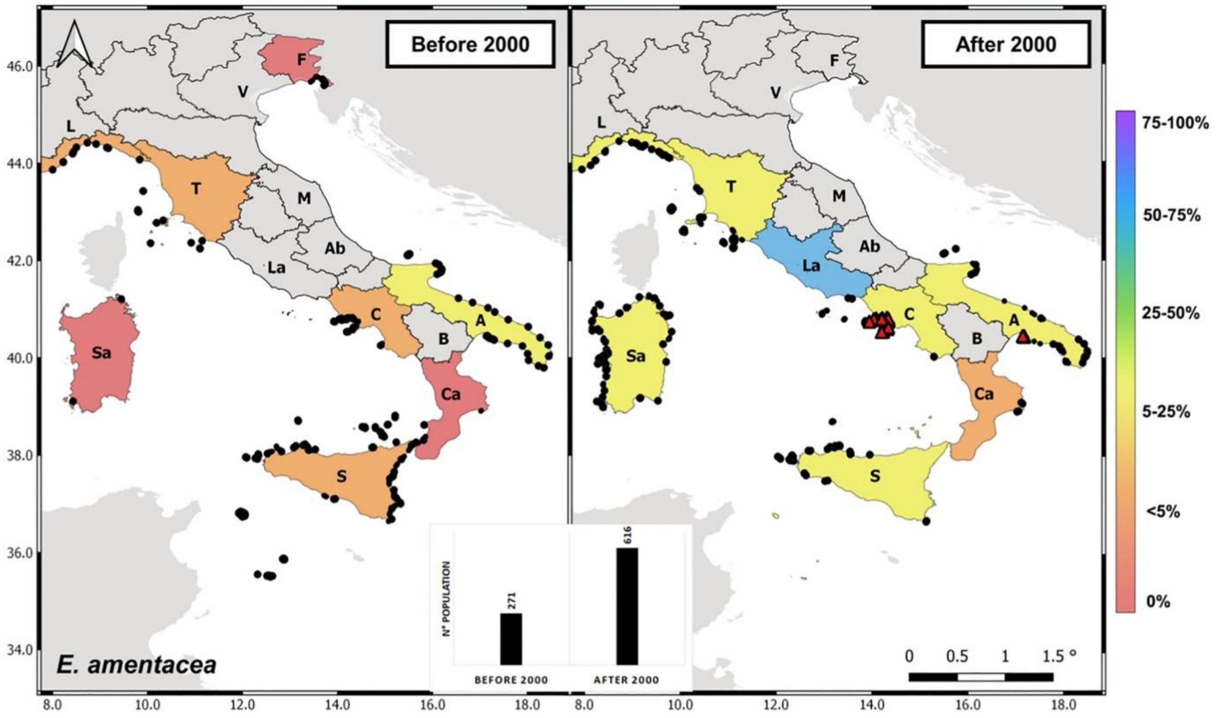


Figure 3.5 Historical and recent distribution of *Ericaria amentacea* along Italian coasts. Black dots and lines indicate the presence of the species, red triangles show documented local extinctions. The color of each region represents the percentage of populations protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region, grey regions indicate no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.

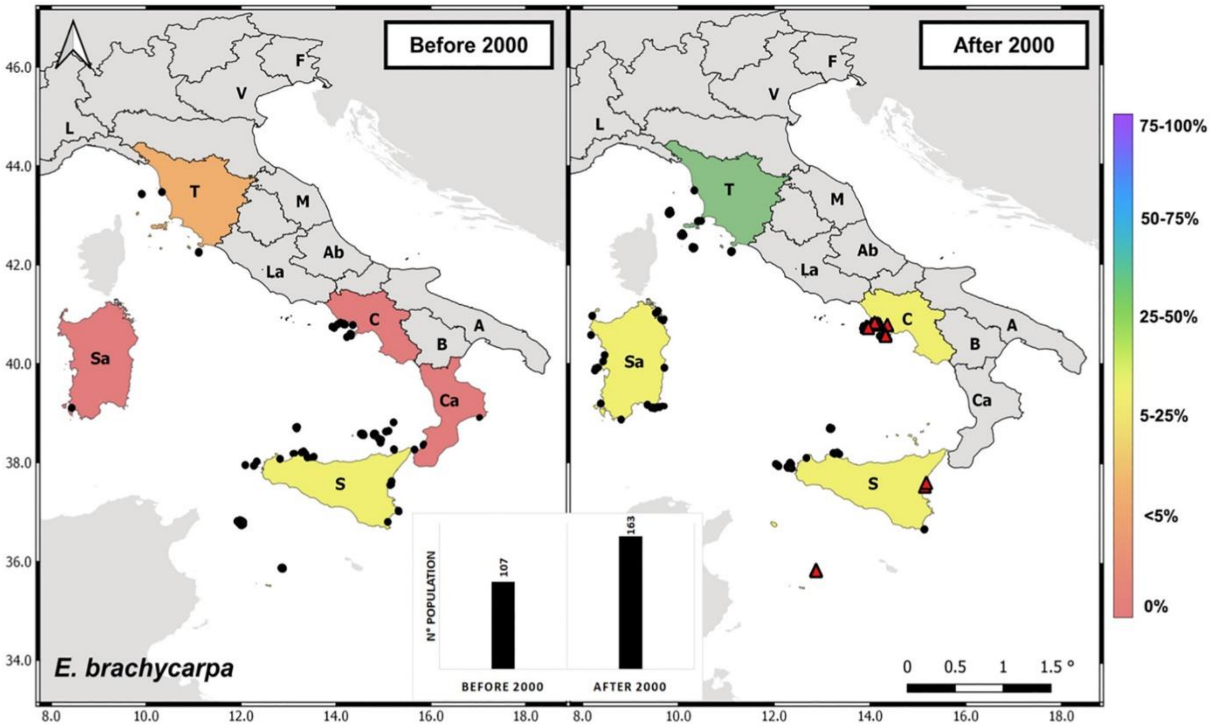


Figure 3.6 Historical and recent distribution of *Ericaria brachycarpa* along Italian coasts. Black dots indicate the presence of the species, red triangles show documented local extinctions. The color of each region represents the percentage of populations protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region, grey regions indicate no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.

3.4.4.15 *Ericaria crinita* (Duby) Molinari & Guiry - intertidal, subtidal

Historical data recorded 119 populations in 8 Italian regions across all seas except the Northern Tyrrhenian and the Southern Adriatic Sea (Fig. 3.7, Table 3.1). None of the populations was protected.

After 2000, 83 populations were reported in 6 Italian regions along the Tyrrhenian, Sardinian, Northern and Southern Adriatic Sea, and the Strait of Sicily. 69.9% of the populations were protected, being located inside 8 MPAs, National Parks (Tuscan and Maddalena Archipelagos), Natura 2000 sites, and SPAMIs (Tables 3.1, 3.2). Between 1992 and 2016, the local extinction of 36 populations was documented in Campania, Sicily (Filicudi, Aegadian islands) and Apulia (mainland, Cheradi and Tremiti islands).

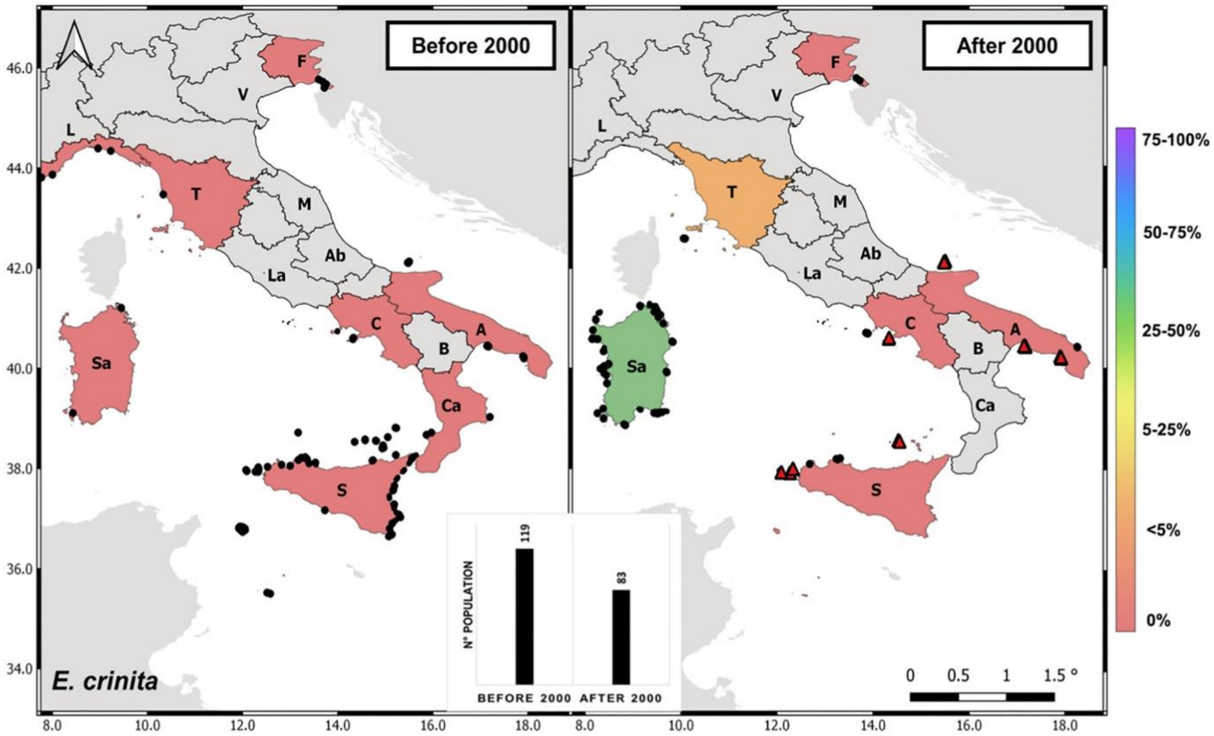


Figure 3.7 Historical and recent distribution of *Ericaria crinita* along Italian coasts. Black dots indicate the presence of the species, red triangles show documented local extinctions. The color of each region represents the percentage of populations protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region, grey regions indicate no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.

3.4.4.16 *Ericaria funkii* (Schiffner ex Gerloff & Nizamuddin) Molinari & Guiry – subtidal

Between 1909 and 1959, three populations of the species were documented on the Sorrento Peninsula in the Gulf of Naples (Campania) (Fig. S3.5, Table 3.1). Eleven further populations were recorded along Ionian Sicilian coasts, Pantelleria, Aeolian and Aegadian islands between 1974 and 1991. The presence of the species in the MPA of the Aegadian islands (instituted in 1991) was confirmed in 2001 (Table 3.2). The local extinction of the species was documented in 1999 in Pantelleria (Sicily) and in 2013–2016 in the Gulf of Naples (Campania).

3.4.4.17 *Ericaria mediterranea* (Sauvageau) Molinari & Guiry – intertidal

Historical data reported 48 populations of *E. mediterranea* in Southern Italy across Campania, Calabria, and Sicily (Fig. S3.6, Table 3.1). No population was protected at the time of observation. In recent years, only 7 populations located in Campania and Sicily were described, and 6 of them were protected inside the MPAs Regno di Nettuno and Capo Gallo, and in the Natura 2000 - SCI & SPA of Punta Campanella - Capri island (Table 3.2). The local extinction of 10 populations across Campania and Sicily (Marettimo) was documented between 2001 and 2016.

3.4.4.18 *Ericaria selaginoides* (Linnaeus) Molinari & Guiry – subtidal

Between 1883 and 1999, 41 populations were recorded across Campania, Calabria, Sicily, and Apulia (Fig. S3.6, Table 3.1). None of them was protected. No populations of the species have been censused after 2000, and the local extinction of all the populations in Campania was verified in 2013–2016.

3.4.4.19 *Ericaria zosteroides* (C. Agardh) Molinari & Guiry – subtidal

99 populations of the species were recorded between 1883 and 1999 in 6 regions across the Ligurian, Central and Southern Tyrrhenian, Sardinian, Ionian Sea and Strait of Sicily (Fig. S3.6, Table 3.1). None of them was protected at the time of observation.

After 2000, only 10 populations have been recorded across Liguria, Tuscany, Sardinia, Campania, and Sicily. All of them were located inside MPAs or Natura 2000 sites (Table 3.2). The local extinction of 21 populations has been documented between 1991 and 1999 in Sicilian islands (Filicudi, Marettimo, Pantelleria, Linosa), in 2013–2016 in Campania (Capri and Ischia, Gulf of Naples), and between 2009 and 2016 in the Natura 2000 SCI of Gallinara island (Liguria).

3.4.4.20 *Gongolaria barbata* (Stackhouse) Kuntze - intertidal, subtidal

Between 1879 and 1999, 187 populations of the species were censused in 10 regions across all seas (Fig. 3.8, Table 3.1). None of the populations was protected at the time of observation.

107 populations have been recorded in recent years in 6 Italian regions across Central and Southern Tyrrhenian, Sardinian, Adriatic and Northern Ionian Sea. 53.3% of them were located inside different types of protected areas (7 MPAs, the National Park of the Maddalena Archipelago, Natura 2000 SCIs and SPAs, and SPAMIs) (Table 3.2). Local extinctions of the species have been documented between 1979 and 2013–2016 for 40 populations across several

regions, including Campania, Apulia, Sicily, Marche and Veneto. Yet, populations in Venice and Lido island, which were considered as extinct respectively in 1996 and 1979–1984, naturally recovered by 2000–2008 due to reduced nutrient load and improved water quality (Curiel et al., 2001, 2002; Marzocchi et al., 2003; Sfriso and Facca, 2011). A regression of the species along the Conero Riviera (Marche) was reported by Perkol-Finkel and Airoldi (2010), who invoked as main cause a combination of local disturbances (rock mining, beach nourishment) along with extreme storm events. In the practice, the species is still well represented and abundant at some sites (Rindi et al., 2020; Rindi, personal observation), although its distribution is patchy due to the particular nature of the substrate in this area. Trampling and swimming due to summer tourism is a major threat for the species at several sites of the Conero Riviera (Rindi, personal observation).

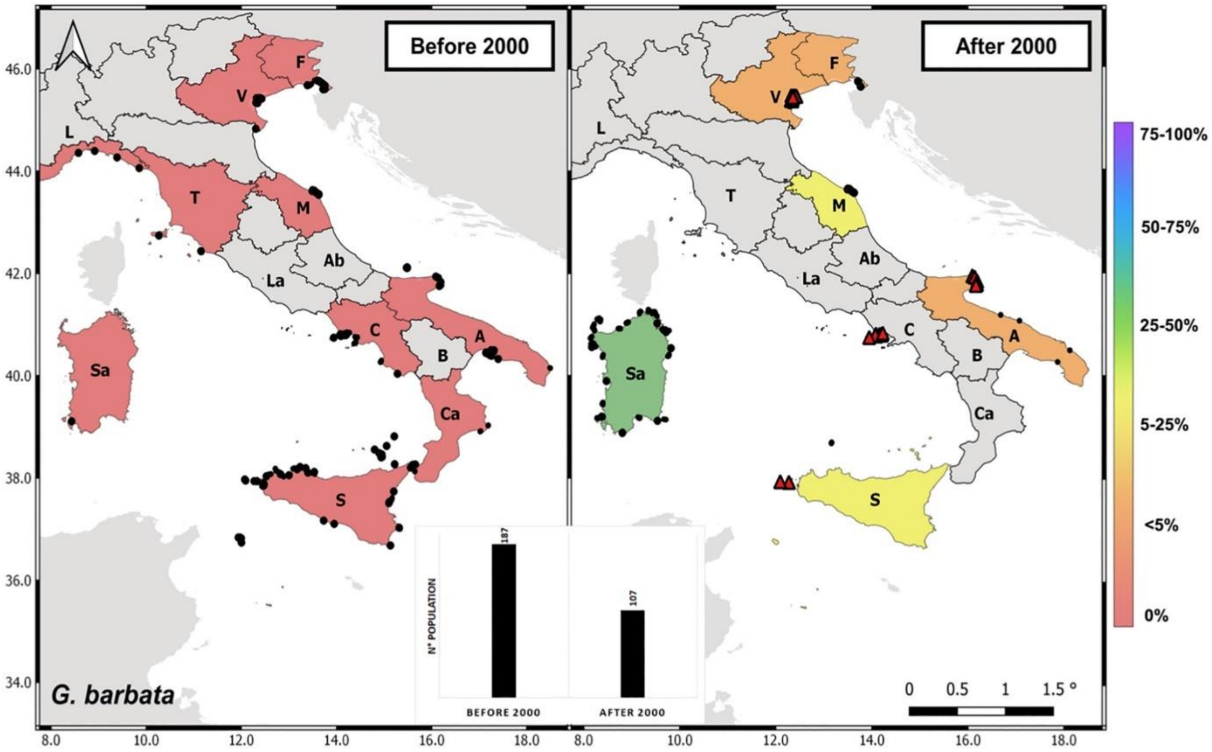


Figure 3.8 Historical and recent distribution of *Gongolaria barbata* along Italian coasts. Black dots and lines indicate the presence of the species, red triangles show documented local extinctions. The color of each region represents the percentage of populations protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region, grey regions indicate no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.

3.4.4.21 *Gongolaria elegans* (Sauvageau) Molinari & Guiry – intertidal, subtidal

Between 1961 and 1985, 50 populations of the species were recorded in Tuscany, Marche, the Ionian coasts of Apulia, and Sicily (Fig. S3.7, Table 3.1). The distribution in Sicily comprised several islands and archipelagos, in addition to vast extensions of the coasts in the Southern Ionian Sea, Southern Tyrrhenian Sea and Strait of Sicily. The population of Capo Passero island was located inside a Natura 2000 SCI.

After 2000, only 7 populations were censused in Sardinia and Liguria. The Ligurian population was located in the Natura 2000 SCI of Gallinara island, and one of the Sardinian populations was protected by the MPA of Capo Caccia - Isola Piana (Table 3.2). In 2000–2001, the local extinction of the species was documented in three locations of Apulia and three Sicilian islands (Marettimo, Linosa and Pantelleria).

3.4.4.22 *Gongolaria montagnei* (J. Agardh) Kuntze – subtidal

Historical studies reported the presence of the species in all sectors of Italian seas, describing 213 populations across 10 regions (Fig. 3.9, Table 3.1). Two populations were protected by the MPA of Ustica and the Natura 2000 - SCIs of Gallinara island (Tables 3.1, 3.2).

After 2000, 74 populations were mapped across 5 regions in the Ligurian, Central and Southern Tyrrhenian, Sardinian, Southern Ionian Sea and Strait of Sicily. Among them, 79.7% of the populations were protected, being located inside 10 MPAs, the National Park of La Maddalena Archipelago, Natura 2000 sites, and SPAMIs. Between 1992 and 1999, the local disappearance of the species was certified for 27 populations across the Gargano promontory and Tremiti islands (Apulia), Pantelleria (Sicily), and in Tuscany. The extinction of further 24 populations was documented in 2009–2016 in the Natura 2000 SCI of Gallinara island (Liguria), and in 2013–2016 in Campania due to sewage outfalls.

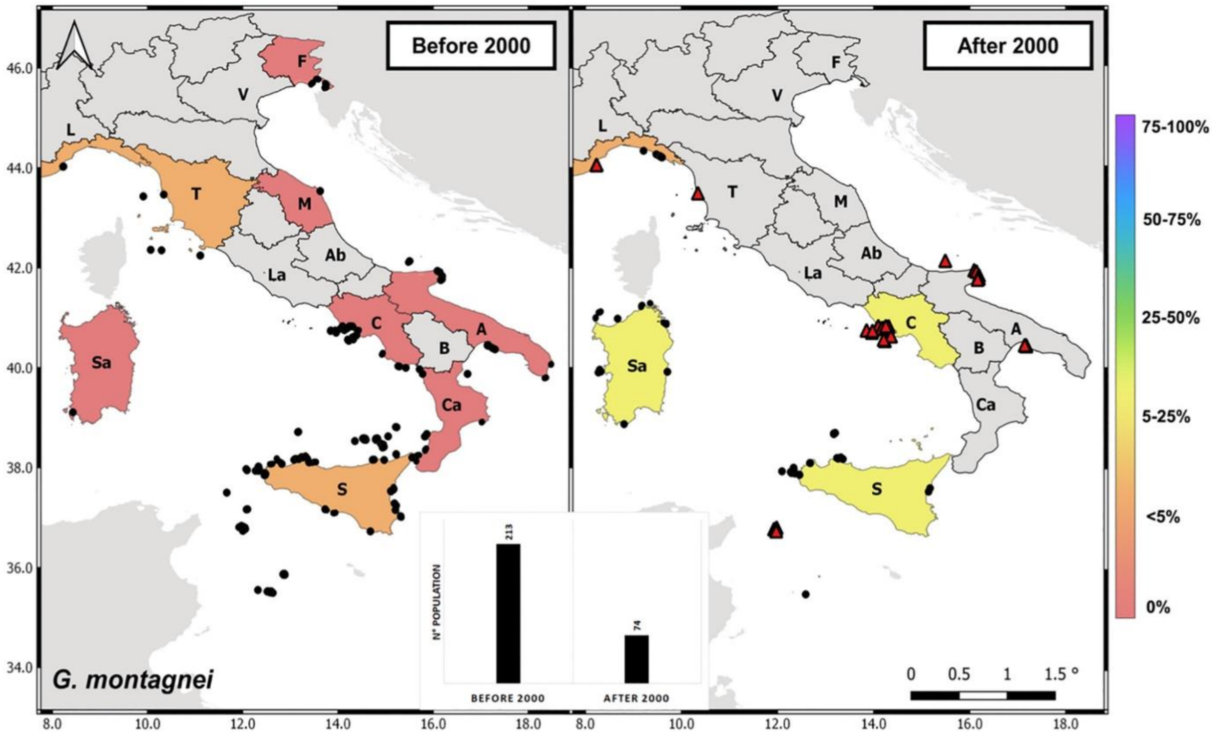


Figure 3.9 Historical and recent distribution of *Gongolaria montagnei* along Italian coasts. Black dots and lines indicate the presence of the species, red triangles show documented local extinctions. The color of each region represents the percentage of populations protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region, grey regions indicate no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.

3.4.4.23 *Gongolaria sauvageauana* (Hamel) Molinari & Guiry - intertidal, subtidal

Between 1879 and 1999, 106 populations of the species were censused in Southern Italy along the coasts of 5 regions (Fig. S3.7, Table 3.1). Two populations located inside the MPA of Ustica were protected (Table 3.2).

In recent years, 15 populations were recorded in Sicily, one population was censused in Tuscany, and three populations were still documented in the Gulf of Naples (Campania). 16 of them were protected, being located inside the MPAs of Ustica, Plemmirio, Punta Campanella, Regno di Nettuno, Tuscan Archipelago, and in Natura 2000 - SCIs. Between 1992 and 2016, the local extinction of the species has been documented for 31 populations across the Gulf of Naples (Campania), Amendolara Sea mount (Calabria), Cheradi islands (Apulia), Linosa, Pantelleria, Favignana and Marettimo islands in Sicily.

3.4.4.24 *Gongolaria squarrosa* (De Notaris) Kuntze – intertidal

In Linosa (Sicily), nine populations of the species were recorded in 1973 (Fig. S3.8, Table 3.1). At the time of observation, the island was not protected. After 2000, 7 populations of the species were censused in Sardinia, Sicily, and Apulia. Sardinian populations were located inside the Pelagos Sanctuary for the Conservation of Marine Mammals, which was instituted in 2001, while the Apulian and Sicilian populations were respectively protected by the Natura 2000 SCI Alimini and the Capo Gallo MPA (Tables 3.1, 3.2).

3.4.4.25 *Gongolaria susanensis* (Nizamuddin) Molinari & Guiry – intertidal

Between 1991 and 1995, a single population of the species was sampled in a non-protected location in eastern Sicily (Alongi et al., 1999a) (Fig. S3.8, Table 3.1). No further populations of the species have been censused.

3.4.4.26 *Gongolaria usneoides* (Linnaeus) Molinari & Guiry – subtidal

A single population of the species was described in 1985 in eastern Sicily (Giaccone et al., 1985) (Fig. S3.8, Table 3.1). The species was not protected, nor was its presence documented in recent years.

Table 3.1 Populations of *Cystoseira s.l.* recorded across Italian seas before and after 2000. For each species, the overall number of populations, the number of populations protected (divided in two categories: MPAs, National and Underwater Parks; or Natura 2000 SICs and SPAs, SPAMIs), the number of extinct populations, and the regional distribution in different seas, are reported. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.

	Before 2000													
	n° population	Ligurian	N Tyrrhenian	C Tyrrhenian	S Tyrrhenian	Sardinian	N Adriatic	C Adriatic	S Adriatic	N Ionian	S Ionian	Strai of Sicily	MPAs, National Parks	Natura 2000, SPAMI
Cystoseira														
compressa	303	L T	T	C	Cl S	Sa	V F	A	A	A Cl	S	S	13	2
corniculata	18					Sa		A		A Cl			-	-
crinitophylla	16				S			A				S	1	-
dubia	17			C	S					Cl	S	S	-	-
foeniculacea	197	L		C B	Cl S	Sa	V F	M A	A	A	Cl S	S	-	-
humilis	53	T			S			M			S	S	-	-
hyblaea	2											S	-	-
pelagosae	2				S								-	-
platyclada	17											S	-	-
schiffneri	27				S			A		A			1	-
sedoides	19											S	-	-
Ericaria														
amentacea	271	L T	T	C	Cl S	Sa	F	M A	A	A	S	S	24	-
barbatula	5											S	-	-
brachycarpa	107	T	T	C	Cl S	Sa				Cl	S	S	5	-
crinita	119	L T		C	Cl S	Sa	F	A		A Cl	S	S	-	-
funkii	14			C	S						S	S	-	-
mediterranea	48			C	Cl S						Cl	S	-	-
selaginoides	41			C	Cl S			A			S	S	-	-
zosteroides	99	L T		C	Cl S					A	Cl S	S	-	-
Gongolaria														
barbata	187	L	T	C	S	Sa	V F	M A	A	A Cl	Cl S	S	-	-
elegans	50	T			S			M		A	S	S	-	1
montagnei	213	L T	T	C B	Cl S	Sa	F	M A	A	A Cl	Cl S	S	1	1

sauvageauana	106			C	Cl S	Sa				A Cl	Cl S	S	2	-
squarrosa	9											S	-	-
susanensis	1										S		-	-
usneoides	1										S		-	-

After 2000

n° population	Ligurian	N Tyrrhenian	C Tyrrhenian	S Tyrrhenian	Sardinian	N Adriatic	C Adriatic	S Adriatic	N Ionian	S Ionian	Strai of Sicily	MPAs, National Parks	Natura 2000, SPAMI	Local extinctions
Cystoseira														
721	L T	T	Sa C La	Sa S	Sa	V F	A Ab M	A	A Cl	S	S	360	237	39
0												-	-	5
0												-	-	1
2				S								2	-	5
32		T	Sa C	S	Sa						S	15	6	38
6	T										S	4	1	11
0												-	-	0
0												-	-	0
0												-	-	1
0												-	-	22
0												-	-	0
Ericaria														
616	L T	T	Sa C La	S	Sa		A	A	A Cl	S	S	307	155	27
1											S	-	-	0
163	T	T	Sa C	Sa S	Sa					S	S	132	12	17
83		T	Sa C	Sa S	Sa	F		A			S	45	13	36
2											S	2	-	3
7			C	S								4	2	10
0												-	-	7
10	L	T	C		Sa						S	8	2	21
Gongolaria														
107			Sa	Sa S	Sa	V F	M	A	A			36	21	40
7	L		Sa		Sa							1	1	6
74	L	T	Sa C	S	Sa					S	S	48	11	51
19		T	C	S						S	S	14	2	31
7			Sa	S	Sa			A				1	3	0
0												-	-	0

0 - - 0

Table 3.2 Populations of the different species *of Cystoseira s.l.* located inside Marine Protected Areas, Underwater, and National Parks. The presence of each species before and after the institution of protection is indicated. b = before, a = after, e = extinct.

Designation name	Year of institution	*C. compressa*	*C. corniculata*	*C. crinitophylla*	*C. dubia*	*C. foeniculacea*	*C. hyblaea*	*C. humilis*	*C. pelagosae*	*C. platyclada*	*C. schiffneri*	*C. sedoides*	*E. amentacea*	*E. barbatula*	*E. brachycarpa*
Isola di Bergeggi MPA	2007	b a											b a		
Portofino MPA	1998	a				b							b a		
Cinque Terre MPA	1997	b a											b		
Arcipelago Toscano National Park	1996	b a				a							b a		b a
Tavolara- Punta Coda Cavallo MPA	1997	a											a		a
Arcipelago della Maddalena National Park	1991	a	b			b							b a		a
Capo Testa – Punta Falcone MPA	2018	b											a		
Isola dell'Asinara MPA	2002												a		
Capo Caccia – Isola Piana MPA	2002	a				a							a		a
Penisola del Sinis – Isola Mal di Ventre MPA	1997	a											a		a
Capo Carbonara MPA	1999	a											a		a
Gaiola Underwater Park	2002	b a e				b e							b a e		b e

Baia Underwater Park	2002	a				b e							b e
Regno di Nettuno MPA	2007	b a e			b e	b a e					b a e		b a e
Punta Campanella MPA	1997	b a e				b e					b a e		b a e
Santa Maria di Castellabate MPA	2009	b				b					b		
Costa della Masseta e degli Infreschi MPA	2009					b							
Capo Rizzuto MPA	1991	b a	b			b					b a		b
Capo Gallo – Isola della Femmine MPA	2002	b a			b a	b a	b a				b a		b a
Isola di Ustica MPA	1986	b a				b		b		a	b a		b a
Isole Egadi MPA	1991	b a			b e	b a e	b a		b e		b a		b a
Isole Pelagie MPA	2002	b		b e		b e	b e		b		b	b	b e
Plemmirio MPA	2004	b			b	b	b				b		b
Isole Ciclopi MPA	1989	b a			b	b					b a		b a e
Porto Cesareo MPA	1997	a									b a		
Torre Guaceto MPA	1991	a											
Isole Tremiti MPA	1989	b a	b	a		b				b e	b a		
Miramare MPA	1986	a				b							

Designation name	Year of institution	*E. crinita*	*E. funkii*	*E. mediterranea*	*E. selaginoides*	*E. zosteroides*	*G. barbata*	**G.** *elegans*	*G. montagnei*	*G. sauvageauana*	*G. squarrosa*	*G. susanensis*	*G. usneoides*	Year of documented extinction
Isola di Bergeggi MPA	2007													
Portofino MPA	1998					a			a					
Cinque Terre MPA	1997									a				
Arcipelago Toscano National Park	1996	a				a	b	b	b a					

Tavolara- Punta Coda Cavallo MPA	1997	a					a		a			
Arcipelago della Maddalena National Park	1991	b a					a		a			
Capo Testa – Punta Falcone MPA	2018	b					b		b			
Isola dell’Asinara MPA	2002	a					a		a			
Capo Caccia – Isola Piana MPA	2002	a					a	a				
Penisola del Sinis – Isola Mal di Ventre MPA	1997	a				a	a		a			
Capo Carbonara MPA	1999	a					a					
Gaiola Underwater Park	2002			b e	b e	b e	b e		b e	b e		2013-2016
Baia Underwater Park	2002								b e			2013-2016
Regno di Nettuno MPA	2007	a e		b a e	b	b a	b e		b a e	a		2013-2016
Punta Campanella MPA	1997	b e	b e	b e		b a			b a	a		2013-2016
Santa Maria di Castellabate MPA	2009					b	b		b			
Costa della Masseta e degli Infreschi MPA	2009		b						b			
Capo Rizzuto MPA	1991	b					b		b	b		
Capo Gallo – Isola della Femmine MPA	2002	b a	b			b	b	b	b a	b	a	
Isola di Ustica MPA	1986	b				b	a	b	b a	b a		
Isole Egadi MPA	1991	b a e	b a	b e		b a e	b e	b e	b a	b e		2001
Isole Pelagie MPA	2002	b				b e		b e	b	b e	b	1999

Plemmirio MPA	2004	b			b	b		b		
Isole Ciclopi MPA	1989	b	b	b	b	b	b	b a	b a	2015-2016
Porto Cesareo MPA	1997	b e				a	b e			2000
Torre Guaceto MPA	1991									
Isole Tremiti MPA	1989	b e				b		b e		1997
Miramare MPA	1986	a				b a				

3.5 Discussion

3.5.1 Historical and recent distribution of *Cystoseira s.l.*: intertidal vs subtidal species

In the last twenty years, overall information on the distribution of *Cystoseira s.l.* exponentially increased, as a result of growing attention to the conservation and restoration of these priority habitat-forming species. Yet, due to easiness of sampling and monitoring, research has focused almost exclusively on intertidal reefs, improving the available information only for a few target species (mostly *C. compressa* and *E. amentacea*). Although the reassessment of their distribution by comparing historical and recent data revealed several local extinctions (Cecere et al., 1996; Grech, 2017; Rindi et al., 2020), the number of populations described in the last two decades has more than doubled, and these species appear still widespread across most of the Italian rocky shoreline. A contribution to the knowledge of their distribution has been provided by the implementation of the CARLIT monitoring (Ballesteros et al., 2007), regularly applied by Regional Agencies since 2007 to assess the Environmental Status according to the EU Water Framework Directive (WFD, 2000/60/EC) (De La Fuente et al., 2018). The CARLIT index indicates *Cystoseira s.l.* assemblages as highly sensitive to environmental stresses, and associates their presence and healthy status (in terms of continuous distribution) to good environmental conditions. In addition, experimental studies, which generally focused on single species, privileged *C. compressa* and *E. amentacea* as target species (16 studies out of 27), thanks to their wide distribution (Crowe et al., 2013; Mancuso, 2016) and/or easiness of access for manipulative purposes (Maggi et al., 2009; Gianni, 2016; Rindi et al., 2017).

The scenario drawn by the available information is more heterogeneous for species inhabiting subtidal reefs. A relevant amount of recent information is concentrated on few species (i.e., *E. brachycarpa*, *E. crinita*, *G. barbata*, *G. montagnei*) inhabiting shallow waters. These species, once distributed across most Italian waters, showed a more jeopardized distribution in recent years. The description of numerous new populations of *E. brachycarpa* and *E. crinita* across Sardinia and the Tuscan Archipelago was counterbalanced by a loss of information across Calabrian and Sicilian waters. *G. barbata* and *G. montagnei*, once distributed across all basins, have drastically contracted their range of distribution, virtually disappearing from several regions, although new populations have been described in Liguria and Sardinia, which has been intensively sampled only in recent years (2 out of 829 records refer to samplings before 2000).

Inferences of extinctions based on comparison of historical and recent data must be drawn with great caution, as they are reliable only for well-studied areas that have been monitored continuously for a long time. In cases of temporally discontinuous investigations, such differences are likely to reflect a decrease in the information available on their distribution, rather than an actual decline. Even considering this, a declining trend for these species across Italian waters is strongly suggested by many local extinctions that have been reliably documented.

A second group of species (i.e., C. *foeniculacea*, *E. mediterranea*, *E. selaginoides*, *E. zosteroides*, *G. elegans*, *G. sauvageauana*), once widely distributed (3–9 regions) and frequently recorded across Italian waters, has rarely or ever been reported in recent studies. Cases of local extinction have been suggested for all these species, indicating that a declining trend combines with the contraction in the range of distribution observed by recent data. Lastly, a conspicuous group of rare species (i.e., *C. corniculata*, *C. crinitophylla*, *C. dubia*, *C. hyblaea*, *C. humilis*, *C. pelagosae*, *C. platyclada*, *C. schiffneri*, *C. sedoides*, *E. barbatula*, *E. funkii*, *G. squarrosa*, *G. susanensis*, *G. usneoides*), which were historically described in few locations (1–3 regions), appears to have virtually disappeared according to recent data. Local extinctions have been documented for most of them (i.e., *C. corniculata*, *C. crinitophylla*, *C. dubia*, *C. humilis*, *C. platyclada*, *C. schiffneri*, *E. funkii*), and very few populations have been censused in recent years (only for *C. humilis*, *E. barbatula*, *E. funkii*, *G. squarrosa*).

3.5.2 Decline, extinction of *Cystoseira s.l.* populations, and drivers responsible for these changes

Despite the overall increase in recent information (i.e., 3238 records after 2000 versus 674 before 2000), the comparison of historical and recent data highlighted a severe loss of information for most *Cystoseira s.l.* species, which may correspond to a declining trend for most species. Few studies were explicitly devoted to verify the persistence of species, by integrating historical and new data. Whenever this approach was adopted, a dramatic situation emerged: for instance, in different areas of the Gulf of Naples the loss of *Cystoseira s.l.* populations was estimated between 60 and 100%, with the disappearance of 7 out of 15 *Cystoseira s. l.* species previously reported in the area and a severe decline of the remaining species (Grech, 2017). A re-evaluation of algal assemblages of the Cheradi islands after 30 years highlighted the extinction of 5 out of 7 *Cystoseira s.l.* species (Cecere et al., 1996), while in Linosa 7 out of 8 *Cystoseira s.l.* species disappeared (Serio et al., 2006), and in the Aegadian islands 9 out of 12 *Cystoseira s.l.* species locally extinguished (Catra et al., 2006). At least two species of *Cystoseira s.l.* have disappeared from the Conero Riviera (possibly more, due to taxonomic uncertainty of some unverifiable literature records; Rindi et al., 2020). This trend suggests that the phenomenon of local extinction of *Cystoseira s.l.* might be underestimated, and the lack of information on the distribution of the majority of *Cystoseira s.l.* species in recent years could mask a dramatic decline across Italian seas.

In addition, the decline in taxonomic expertise might have substantially compromised the ability of researchers to identify several *Cystoseira s.l.* species.

Despite the scientific interest for the decline of *Cystoseira s.l.* forests, the identification of the local drivers of loss is still a daunting task. Very few studies (13 out of 169) clearly identified the stressors affecting the canopies. When the information was provided, urbanization (e.g., sewage outfall, coastal development, harbor proximity; Mangialajo et al., 2008; Grech, 2017), agricultural or industrial activities (e.g., pollutants, eutrophication, water turbidity due to terrigenous sediments; Sfriso, 1987; Cecere et al., 1996; Cormaci et al., 2001; Grech, 2017; Catra et al., 2019), aquaculture or fisheries (Cecere et al., 1996; Grech, 2017) were identified (Fig. 3.10). In addition, overgrazing by sea urchins (Catra et al., 2019), sea water warming (Serio et al., 2006), and the combination of local stressors (i.e., rock mining and beach nourishments) and extreme storminess (Perkol-Finkel and Airoldi, 2010), were advocated as drivers of decline at the scale of populations. Other experimental studies investigated mechanisms that might compromise the resilience of brown algal forests, likely contributing to their fragmentation (e.g., human trampling, Milazzo et al., 2002, 2004; competition with invasive species, Bulleri et al., 2017). Hence, although the scientific community has agreed on the major threats for Mediterranean macroalgal forests (i.e., habitat loss due to coastal development, Airoldi and Beck, 2007; pollution, Munda, 1982; Soltan et al., 2001; heavy metals, Sales et al., 2011; de Caralt et al., 2020; eutrophication, Arevalo et al., 2007; outbreaks of grazer populations including sea urchins, salema fish and rabbitfish Sala et al., 1998, 2011; climate change, Bevilacqua et al., 2019; Verdura et al., 2021), a case-by-case identification of stressors is far from being reached (Fig. 3.10), although this step represents a necessary requisite to plan local conservation interventions, or to identify putative sites for restoration purposes (Gann et al., 2019).

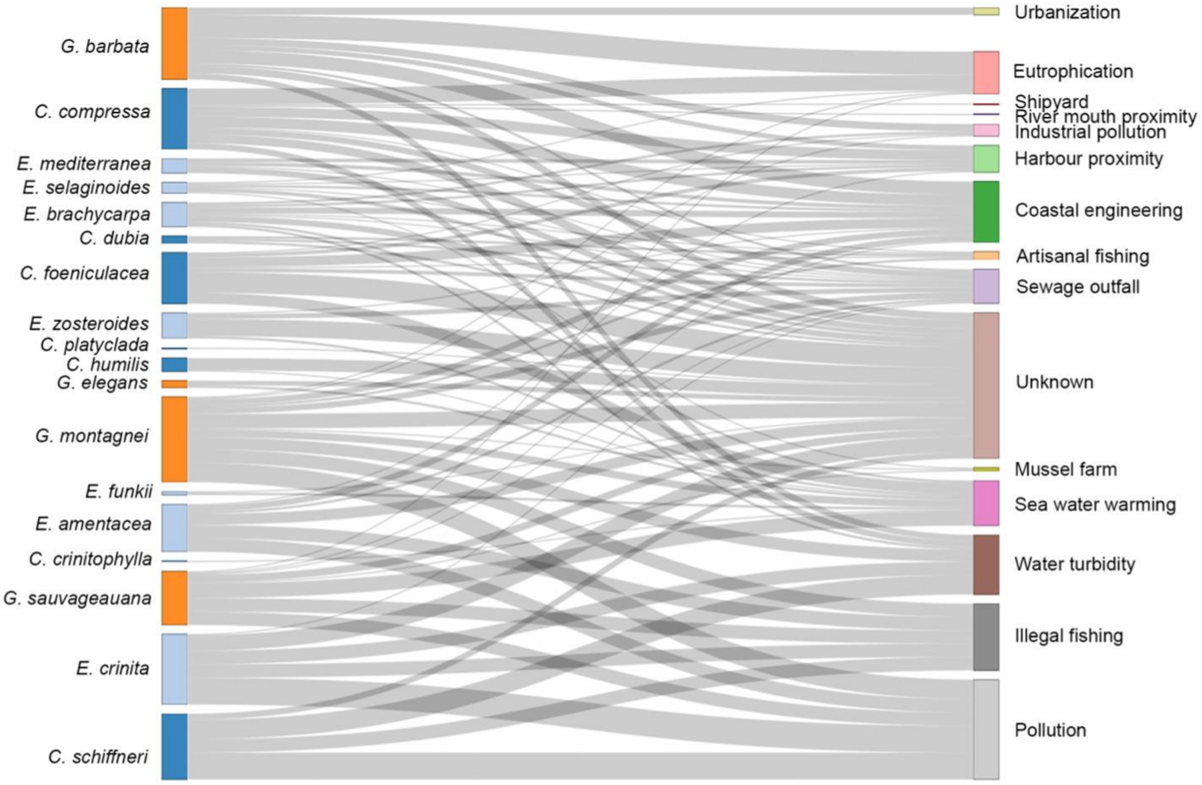


Figure 3.10 Sankey diagram representing the putative drivers affecting populations of different *Cystoseira s.l.* species described in the literature. The width of the nodes and lines is proportional to the number of extinct populations attributed to each stressor. Multiple drivers might have been attributed to a single extinction event.

3.5.3 Success stories of natural recovery

Similar to the cases observed along the Catalan and Istrian coasts (Roca et al., 2015; Ivesa et al., 2016), the partial recovery of fucalean forests has been reported in Italian waters as a result of the mitigation of local stressors and amelioration of water quality due to the reduction of nutrient loads and water turbidity in the Venice lagoon (Marzocchi et al., 2003; Sfriso and Facca, 2011). Yet, the two species populating the lagoon, namely *G. barbata* and *C. compressa*, being provided with aerocysts that allow medium-distance dispersal of vegetative fragments, present a higher potential to naturally recolonize sites after local extinction compared to the majority of *Cystoseira s.l.* species, which are generally characterized by very low dispersion distance of the zygotes (Thibaut et al., 2014). For these species of *Cystoseira s.l.*, with virtually no connectivity with other populations, restoration emerges as the solely opportunity to contrast their decline. Yet, one of the key pieces of information for the planning of restoration interventions is the historical extent of the fucalean forests and an eventual estimation of the fragmentation or contraction of their distribution. Attention to this aspect would allow to operate prompt reforestation interventions before the populations are totally lost, eventually preventing the settlement and spread of organisms that might inhibit their future recovery (e.g., the invasive *Sargassum muticum*, Marzocchi et al., 2003; algal turfs or mussels, Perkol-Finkel and Airoldi, 2010). Yet, except for a few studies including mapping of *Cystoseira s.l.* populations to establish a baseline for selected locations (e.g., Calvo et al., 1980; Gianni, 2016; Grech, 2017), only the study by Perkol-Finkel and Airoldi (2010) provided an accurate estimate of the historical extent, severity of decline, and degree of fragmentation of the declining populations of *G. barbata* along the Conero promontory.

3.5.4 How much are censused *Cystoseira s.l.* populations protected?

Before 2000, a negligible proportion of *Cystoseira s.l.* populations were protected (2.6%). The proportion of protected populations has increased in recent years to 77.8%, with nearly two thirds of the populations included in MPAs, National and Underwater Parks, and one third protected by Natura 2000 sites or SPAMIs. The regulations and enforcement of MPAs and National Parks guarantee protection to *Cystoseira s.l.* by limiting coastal urbanization and overfishing, including the illegal and destructive practice of date mussel harvesting, which can foster the permanent shift from forested habitats to barrens (Colletti et al., 2020). In addition, limitations of the fishery pressure may indirectly safeguard *Cystoseira s.l.*, by allowing the recovery of high-level predators, which control herbivore populations avoiding overgrazing (Sala et al., 1998, 2012), or directly limiting the discard of abandoned fishing gears (Capdevila et al., 2016). Instead, Natura 2000 Sites and SPAMIs are characterized by more limited regulations, as they cannot forbid several human activities such as fishery or shipping, and lack no-take zones. In addition, limited enforcement and the lack of management and monitoring plans generally make their regulations poorly applied (Olsen et al., 2013). Hence, their efficacy in protecting brown macroalgal forests might be scarce. Yet, recent studies highlighted that even MPAs might have limited protection efficacy, since they cannot constrain regional- or basin-scale stressors such as eutrophication or water turbidity, nor mitigate global change effects (Gianni et al., 2013; Fraschetti et al., submitted). This is further supported by the extinction of several *Cystoseira s.l.* species (*C. compressa*, *C. dubia*, *C. foeniculacea*, *C.*

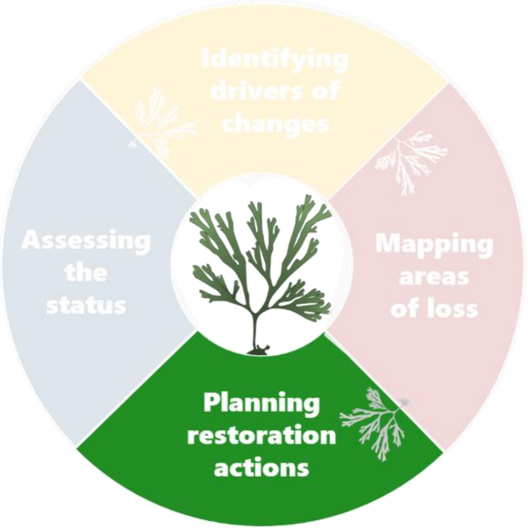
platyclada, *E. amentacea*, *E. brachycarpa*, *E. crinita*, *E. mediterranea*, *E. zosteroides*, *G. barbata*, *G. elegans*, *G. sauvageauana*) located in the MPAs of Punta Campanella, Regno di Nettuno, Porto Cesareo, Aegadian, Pelagian, Ciclopi and Tremiti islands, documented in our study.

77.9% of the populations currently protected inhabit intertidal reefs, including the non-protected species *C. compressa* (41.3%), *E. amentacea* and *E. mediterranea*. Among species populating subtidal reefs, *E. brachycarpa* is the most frequently censused in protected areas, followed by *G. montagnei* and *E. crinita* (respectively 10%, 4.1%, and 4% of the protected populations). All the *Cystoseira s.l.* species for which the presence has been confirmed during the last twenty years are partially or totally protected, except for *E. barbatula*. However, due to the lack of monitoring programs dealing specifically with brown fucalean species, half of the MPAs did not collect any information after their institution, to confirm the presence or disappearance of species previously censused within their borders. Only about one third of the MPAs have an updated overview of the situation, generally corresponding to the disappearance of most *Cystoseira s.l.* species. A positive exception is represented by the MPAs located in the Sardinian region, for which the scarcity of historical data is compensated by a recent and widespread census activity.

3.6 Conclusions

With increasing recognition of the need to adopt restoration actions, several analyses have been carried out showing the challenges that restoration should tackle to be effective (Abelson et al., 2020; Fraschetti et al. submitted). The development of effective methods for restoration upscaling, the incorporation of innovative tools to promote the consideration of climate changes, and the integration of social and ecological restoration priorities, are among the most frequent issues emerging from these perspectives. Our review on the iconic macroalgal forests shows the dramatic lack of baseline information for a group of species which are presently the focus of many restoration interventions. In particular, we found three topics deserving specific attentions in the next future: the undervaluation of habitats less accessible to monitor, the lack of long-term data to unveil drivers of loss, and an updated reporting about the conservation status of the species of interest to plan future interventions.

Macroalgal forests are a paradigmatic example to document several limits that can compromise the recent effort from scientists, policy makers and stakeholders such as MPA managers to limit the observed shifts or plan restoration actions. Together with the challenges imposed by climate changes, data about historical presence, present occurrence and the conservation status of the target species/groups are critical to succeed in restoration actions. Our study documents a dramatic lack of information for most *Cystoseira s.l.* species, together with the state of their conservation, requiring timely interventions. Above all, there is an urgent need to implement regular monitoring plans to update the knowledge on the distribution and status of *Cystoseira s.l.* populations within protected areas, representing preferential areas for restoration activities, given recovered environmental conditions and increased chances of restoration success (Medrano et al., 2020).



CHAPTER 4

THE CHALLENGE OF SETTING RESTORATION TARGETS FOR MACROALGAL FORESTS UNDER CLIMATE CHANGES

CHAPTER 4

The challenge of setting restoration targets

4.1 Abstract

The process of site selection and spatial planning has received scarce attention in the scientific literature dealing with marine restoration, suggesting the need to better address how spatial planning tools could guide restoration interventions.

In this study, for the first time, the consequences of adopting different restoration targets and criteria on spatial restoration prioritization have been assessed at a regional scale, including the consideration of climate changes. We applied the decision-support tool Marxan, widely used in systematic conservation planning on Mediterranean macroalgal forests. The loss of this habitat has been largely documented, with limited evidences of natural recovery. Spatial priorities were identified under six planning scenarios, considering three main restoration targets to reflect the objectives of the EU Biodiversity Strategy for 2030.

Results show that the number of suitable sites for restoration is very limited at basin scale, and targets are only achieved when the recovery of 10% of regressing and extinct macroalgal forests is planned. Increasing targets translates into including unsuitable areas for restoration in Marxan solutions, amplifying the risk of ineffective interventions.

Our analysis supports macroalgal forests restoration and provides guiding principles and criteria to strengthen the effectiveness of restoration actions across habitats. The constraints in finding suitable areas for restoration are discussed, and recommendations to guide planning to support future restoration interventions are also included.

Keywords: Marine Spatial Planning, Site Selection, Marxan, Restoration, Macroalgal Forests, *Cystoseira sensu latu*

4.2 Introduction

In marine and coastal areas, species populations, habitats and ecosystems are constantly modified under multiple anthropogenic stressors with severe consequences on marine biodiversity and ecosystem services (Chefaoui et al., 2017; Colletti et al., 2020; Gissi et al., 2021; Bevilacqua et al., 2021, Tamburello et al., 2022). The rate of changes that these ecosystems are experiencing calls for adopting new strategies to complement the traditional approaches of ecosystem conservation (Lester et al., 2020). Among these, marine ecosystem restoration, by the implementation of intentional activities (e.g., environmental remediation, ecological engineering, reconstruction, creation/re-creation or ecological rehabilitation), is increasingly considered as a prominent tool to promote and assist the recovery of degraded ecosystems (Society for Ecological Restoration International Science Policy Working Group, 2004). Restoring ecosystems means bringing back biodiversity and ecosystem services, representing thus a key motivation for funding and implementing restoration projects (Matzek, 2018; CBD, 2020). As a part of the EU Biodiversity Strategy for 2030, specific and binding restoration targets have been proposed in 2021 (EC, 2020). However, while criteria for reaching conservation targets have been largely discussed (Zhao et al., 2020), setting targets for restoration still needs a framework to guide the process of restoration prioritization.

One way to foster restoration targets is adopting Marine/Maritime Spatial Planning (MSP) principles, i.e., planning the spatial allocation of restoration efforts based on ecological knowledge and socio-economic constraints (Lester et al., 2020). MSP represents an effective approach in the challenge of balancing conflicting human demands of the maritime space, protecting the environment in a spatially explicit way and implementing ecosystem-based management to simultaneously fulfil environmental, biological and economic requirements (Leslie et al., 2003; Klein et al., 2008; Ehler and Douvere, 2009; Tuda et al., 2014; Stelzenmüller et al., 2021). MSP can be critical to achieve the targets of the current development and environmental policies, such as the United Nations Sustainable Development Goal 14 (UN SDG 14) (UN, 2015; Böhnke-Henrichs et al. 2013, Frazão Santos et al. 2020, Kirkfeldt and Frazão Santos 2021) and the EU Biodiversity Strategy for 2030 (Katsanevakis et al., 2020), increasing the effectiveness of restoration practices. Considering the high costs required for restoring marine habitats at large spatial scales (Bekkby et al., 2020), the selection of sites where restoration is more likely to be effective can largely contribute to the achievement of restoration objectives with a high return on investment (Bayraktarov et al., 2016). However, in the marine environment, the process of spatial planning is still scarcely considered for the attainment of environmental goals mostly focusing on economic demands (Katsanevakis et al., 2020; Trouillet, 2020). Yet, considering where restoration activities are undertaken can result more important than how they are carried out (Fraschetti et al., 2021).

Marxan software (Ball et al., 2009) is the most widely used open-source decision-support tool (Watts et al., 2017) in conservation. Initially conceived for the design of protected areas network meeting several ecological, social and economic criteria at once (Ball et al., 2009, Christensen et al., 2009), Marxan integrates cutting-edge conservation science alongside human uses shaping dialogue between scientists and decision-makers. The application of Marxan in a restoration perspective is still very limited and has mainly been implemented in terrestrial and freshwater realms (see Adame et al., 2015; Renwick et al., 2014; Yoshioka et al., 2014; Jellinek, 2017; Hermoso et al., 2021). Nolan et al. (2021) recently introduced the predictions of coral cover in a spatial prioritization analysis with Marxan to distinguish between protection and restoration areas, targeting the most degraded areas for restoration, and avoiding low-quality areas for protection.

In this study, for the first time (to the best of our knowledge), the consequences of adopting different restoration targets and criteria on spatial restoration prioritization have been assessed at a regional scale. We focused on Mediterranean macroalgal forests since, in the last 20 years, forests loss has been largely documented across the whole basin for local and global cumulative impacts (Sales and Ballesteros, 2009, Fulton et al., 2019, de Caralt et al., 2020, Verdura et al., 2021, Tamburello et al., 2022), with limited evidences of natural recovery (Riquet et al., 2021), even within protected conditions (Sala et al., 2012, Tamburello et al., 2022).

Spatial priorities were identified by Marxan under six planning scenarios considering three main restoration targets, conceived to reflect the objectives of the EU Biodiversity Strategy for 2030. We combined fine-scale data about their present and past distribution across the Mediterranean Sea with data about their environmental requirements gathered using the Habitat Suitability Model (HSM) outputs provided by Fabbrizzi et al. (2020). The use of HSMs in supporting environmental management is critical, since they provide relevant insights about potential drivers of habitat loss (Catucci et al., 2022). In addition, since the distribution of fucalean forests is strongly constrained by warming temperatures (Verdura et al., 2021), we included in the spatial planning the distribution of Sea Surface Thermal Anomalies (SSTA) hotspots across the Mediterranean Sea. Finally, the aim of this study is also to provide recommendations to guide the spatial planning of future marine restoration actions.

4.3 Materials and Methods

4.3.1 Study area and Planning Units

Considering the whole Mediterranean coastline as our Planning Region (i.e., our study area), we defined Planning Units (PUs) as the set of potential sites from which to select restoration areas. We used square PUs, superimposing a regular grid with a resolution of 0.004166 decimal degrees (i.e., about 400 m^2) to the entire coastline, obtaining 112,539 PUs. The adopted resolution matches the ones of the HSM developed in Fabbrizzi et al. (2020), as the outcomes of that model, expressing the suitability of each area for fucalean forests occurrence (with values ranging in the [0,1] interval), were used to identify areas suitable for restoration.

In our analysis, in fact, we locked out from the potential restoration areas to be selected PUs corresponding to the distribution of cells classified as unsuitable by the HSM, i.e., cells with HSM values less than 0.61. This value corresponds to the cut-off which allowed to optimize the accuracy of the HSM predictions by reaching the best compromise between the sensitivity and the specificity of the model (Fabbrizzi et al., 2020). This exclusion ensured that the analysis only retained those sites in which restoration efforts are more likely to be effective, indicating the presence of suitable conditions.

Areas with high frequency of Sea Surface Temperature Anomalies (SSTA) were locked out from the analysis as well. Considering the distribution of SSTA hotspots into the spatial planning was crucial to exclude areas where the high frequency of extreme climatic events can compromise the effectiveness of restoration actions. SSTA data were retrieved from the NOAA’s Environmental Modeling Center database[1]: monthly values over the past five years (2015–2020) were taken into account, only including spring and summer seasons (months between March and August), since temperatures of these periods are considered the most

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[1] https://www.emc.ncep.noaa.gov/emc_new.php

critical for recruitment and survival of Mediterranean fucaleans (Sauvageau, 1912; Orfanidis et al., 2021). Thus, areas where temperatures exceed the long-term average from 1981 to 2010 by at least 1 °C (Chollett et al., 2022) above the 75° percentile were considered as unsuitable for restoration actions (hereafter referred to as “thermal anomalies hot spots”).

Finally, areas where forests are already present (i.e., existing forests that are in good state and do not need restoration) and those for which no occurrence data were available were locked out too. More specifically, we locked out from the Marxan analysis 112,219 PUs, out of which 70,410 were classified as unsuitable according to the HSM and the SSTA layer (Fig. 4.1a), 36,814 corresponded to areas where forests occurrence is documented, while for the remaining 4995 no occurrence data are available (Fig. 4.1b).

4.3.2 Restoration features

Restoration features are intended to represent the entities (e.g., species, habitats, ecosystems) to be restored. In this study, we determined restoration features in three steps. Firstly, we considered the following types of fucalean forests: i) Regressing forests (Rf), i.e., areas where a pattern of regression from a previous healthy status of the canopy was documented by literature analyses and expert knowledge; ii) Extinct forests (Ef), i.e., areas where fucalean forests were historically documented but are currently absent.

To map these forests, we used an existing dataset that assembled data about the current and historical distribution of fucalean forests across the Mediterranean Sea (Fabbrizzi et al., 2020), refined by conducting a literature review and data collection. For the literature review, which involved both peer-reviewed and grey literature, different databases were used: ISI Web of Science (WOS), Scopus, AlgaeBase[2] and GBIF[3]. The search of pertinent articles was conducted for the whole Mediterranean Sea, setting 2020 as the only temporal cut-off. A total of 1236 studies including the keywords “*Cystoseira*” and “Mediterranean” were evaluated, retaining only those reporting geographical information about the distribution (presence/absence) and, when available, the status (coverage, trend, and present-past conditions) of any *Cystoseira* species. The screening of the literature was completed before the recent split of the genus

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[2] https://www. algaebase.org/
[3] https://www.gbif.org/

Cystoseira in the three separate genera *Cystoseira*, *Ericaria* and *Gongolaria* (Molinari-Novoa and Guiry, 2020). All studies without georeferenced data were no further examined and were excluded from the review. The information obtained from the literature was also combined with new data collected in the field within the framework of the AFRIMED project[4] (see Orfanidis et al., 2021). Additional data were provided by AFRIMED partners as personal information.

The assembled dataset comprises a total of 39,293 occurrence records (25,145 digitized as a vector shapefile of points and 14,148 as a vector shapefile of polylines) covering a large span of time from 1789 to 2020. Table S4.1 shows the literature used for data collection, composed of 335 articles, including both peer-reviewed and grey literature. Each article is labelled with an ID number which corresponds to the ID of the related records in the georeferenced dataset. The contribution to the dataset by AFRIMED partners through personal data is also listed in Table S4.2.

The dataset expresses the number of Regressing forests and Extinct forests occurring in each PU. The species considered for the identification of the restoration features were: *C. compressa* (Esper) Gerloff & Nizamuddin, *C. foeniculacea* (Linnaeus) Greville, *C. humilis* Schousboe ex Kützing, *E. amentacea* (C. Agardh) Molinari & Guiry, *E. brachycarpa* (J. Agardh) Molinari & Guiry, *E. crinita* (Duby) Molinari & Guiry, *E. mediterranea* (Sauvageau) Molinari & Guiry, *G. barbata* (Stackhouse) Kuntze and *G. elegans* (Sauvageau) Molinari & Guiry. These species were assumed as a rather uniform ecological entity pertaining all to the shallow rocky shores (see Fabbrizzi et al., 2020).

In the second step, each type of forest was split in two restoration features according to the level of Habitat Richness (HR) surrounding the forest. The HR data layer was assembled using the model data on the distribution of the following Mediterranean species/habitat: *Posidonia oceanica* meadows, bioconstructions (coralligenous formations and maërl beds), essential fish habitats (nursery and spawning grounds), and deep-sea habitats (Martin et al., 2014; Boero et al., 2016). We preferred data derived from models to the raw ones since spatial information on marine species and habitats are largely incomplete. HR data were combined in a polygon shapefile which displayed the number of different habitats for each PU across the

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[4] http://afrimed-project.eu/

Mediterranean Sea. Given the distribution of HR values across PUs, we considered the value corresponding to the third quartile as a threshold discriminating between low and high HR.

HR was used to incorporate into the planning process the evidence that positive species interactions can enhance restoration success (Eger et al., 2020). Facilitation between primary producers and indirect trophic effects have the potential to mitigate the effects of warming on the distribution of species, expanding the range of physical conditions under which species can persist (Silliman et al., 2015; Bulleri et al., 2018; Eger et al., 2020). Under these criteria, we defined 4 restoration features: 1) Rf in high HR; 2) Rf in low HR; 3) Ef in high HR; 4) Ef in low HR (Fig. 4.1). Finally, a further class of restoration features were assessed using the distribution of the Habitat Suitability Model false positive cases (hereafter referred to as "HSMf"). These features correspond to the areas where fucalean forests have never been documents but are suitable for their growth according to the HSM provided by Fabbrizzi et al. (2020), and hence are also suitable for restoration. In this study, HSMf were treated as equivalent to Ef in low HR under the assumption that they had macroalgal forests, now extinct due to environmental or human pressures, leading to a great uncertainty in restoration outcomes, since in these areas the presence of forests was only predicted.

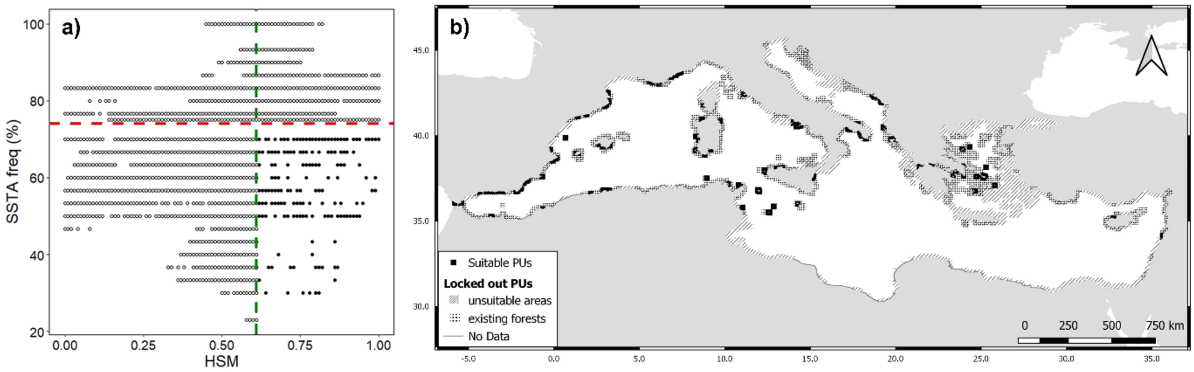

Figure 4.1 Planning region and identification of PUs. (a). The scatterplot with the distribution of suitable (black points) and unsuitable (white points) areas according to the HSM outputs and the SSTA frequency. The vertical green dotted line represents the threshold assessing suitability according to the HSM (values > 0.61). The horizontal red dotted line represents the threshold assessing suitability according to the SSTA (values < 75%). Areas where forests occurrence is documented and those for which no occurrence data are available are not represented in the scatterplot. (b). Map of the distribution of suitable and locked out PUs.

4.3.3 Restoration targets and scenarios

Restoration targets express the minimum proportion of the restoration features to be included in the planning solutions. We set six restoration scenarios with two sets of targets each: a) restoring 10% of fucalean forests in high HR and 5% of those in low HR, without considering HSMf; b) restoring 10% of fucalean forests in high HR and 5% of those in low HR, including HSMf; c) restoring 20% of fucalean forests in high HR and 10% of those in low HR, without considering HSMf; d) restoring 20% of fucalean forests in high HR and 10% of those in low HR, including HSMf; e) restoring 30% of fucalean forests in high HR and 20% of those in low HR, without HSMf; f) restoring 30% of fucalean forests in high HR and 20% of those in low HR, including HSMf (Fig. 4.1).

4.3.4 Costs of restoration

Costs data reflect the effort to be allocated in including a PU among priority areas for restoration. They pertain to the socio-economic implications of conducting restoration activities. We estimated costs of restoration of each PU from Verdura et al. (2018) where costs for restoring 25 m^2 of a forest has been assessed. These costs represent an average between 1,092 €/25 m^2 (costs of in situ restoration) and 2,665 €/25 m^2 (costs of ex situ restoration). The obtained value was then calibrated on the basis of facilities distribution which affects the costs linked to the transport for both the in situ and the ex situ techniques. Thus, we considered 0.40 €/km (see Verdura et al. 2018) to assess the cost of covering the distance between the restoration site and the nearest facility. The cost of a PU decreases in relation to its proximity to the following facilities: i) International, National and Regional MPAs. Information about their distribution across the Mediterranean Sea were retrieved from MAPAMED database[5]; ii) Ports (World Port Index, 2014[6]: this dataset was derived from the 23rd Edition of the World Port Index prepared and published by the United States National Imagery and Mapping Agency); iii) Diving facilities[7]; iv) Marine Stations (MARS network[8]), Marine Institutes (CIESM[9]) and

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[5] https://medpan.org/main_activities/mapamed/

[6] https://maps.princeton.edu/catalog/sde-columbia-worldports2014

[7] https://www.google.com/maps/d/viewer?hl=en&gl=US&ved=0CFMQjwU&ei=SOzqTIfnCJGQyQWB7KXcDg&ie=UTF8&oe=UTF8&msa=0&mid=1WLySIlMSbCBLJ0TtucPh43BOgcw&ll=36.648535204425414%2C13.872461034103644&z=5

[8] https://www.marinestations.org/members/mars-members-map/

[9] http://ciesm.org/online/institutes/CIESM_InstitutesFullIndex.php

Specially Protected Areas Regional Activity Centres (SPA/RAC[10]); v) Locations of previous experiences on restoration activities, in terms of scientific background documented with published studies (data collected in the framework of MERCES project[11]. All data for these layers were processed and converted into the same raster format of the HSM to integrate the information in the PUs grid.

Hence, costs for each PU were defined as:

$$PUc = Rc + (\overline{PUf} \times 0.40\ )$$

where PUc is the cost estimated for a PU, Rc is the average between in situ and ex situ restoration costs for a surface area of 25 m^2, $\overline{PUf}$ is the distance between the PU and the nearest facility for restoration in km and 0.40 is the cost of transports per km (€/km).

4.3.5 Marxan parameters

The three above mentioned scenarios, with their respective restoration targets (see section 4.3.3), were adopted to run Marxan. Based on a heuristic algorithm, specifically the "simulated annealing", Marxan finds multiple near-optimal solutions to maximize conservation (here restoration) interests while minimizing costs with the constraint of meeting the set of conservation (here restoration) targets. For each scenario, Marxan was run 100 times using 1,000,000 iterations, resulting in two main outputs: the best planning solution and the selection frequency of PUs, i.e., the number of times a PU is selected over the 100 runs as a measure of its relative priority (Fig. 4.2). The Boundary Length Modifier (BLM) value, used to improve the spatial compactness of individual solutions, was set to 0 since it was not critical in our analysis to have clumped solutions. The Feature Penalty Factor (FPF), a multiplier that determines the size of the penalty that will be added to the objective function if the target for a feature is not met, was calibrated to optimize Marxan performance in finding solutions. Too small FPF values mean achieving the "lowest cost" solution but, at the same time, missing several targets, since costs of selecting additional PUs is greater than the small penalties for missing the targets. Conversely, too large FPF values reduce Marxan potential for exploring

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[10] https://www.rac-spa.org/map_structure
[11] http://www.merces-project.eu/

different options resulting in higher cost solutions (Fischer et al. 2010). We iteratively increased the FPF, starting from 1, until finding the value that allows minimizing both the amount of features by which the targets are not met (namely "shortfall") and the costs for solutions in each scenario. In this analysis, Marxan solutions, supporting the decisions which underpin the spatial planning process, were used to identify priority areas for fucalean forests restoration in the Mediterranean basin.

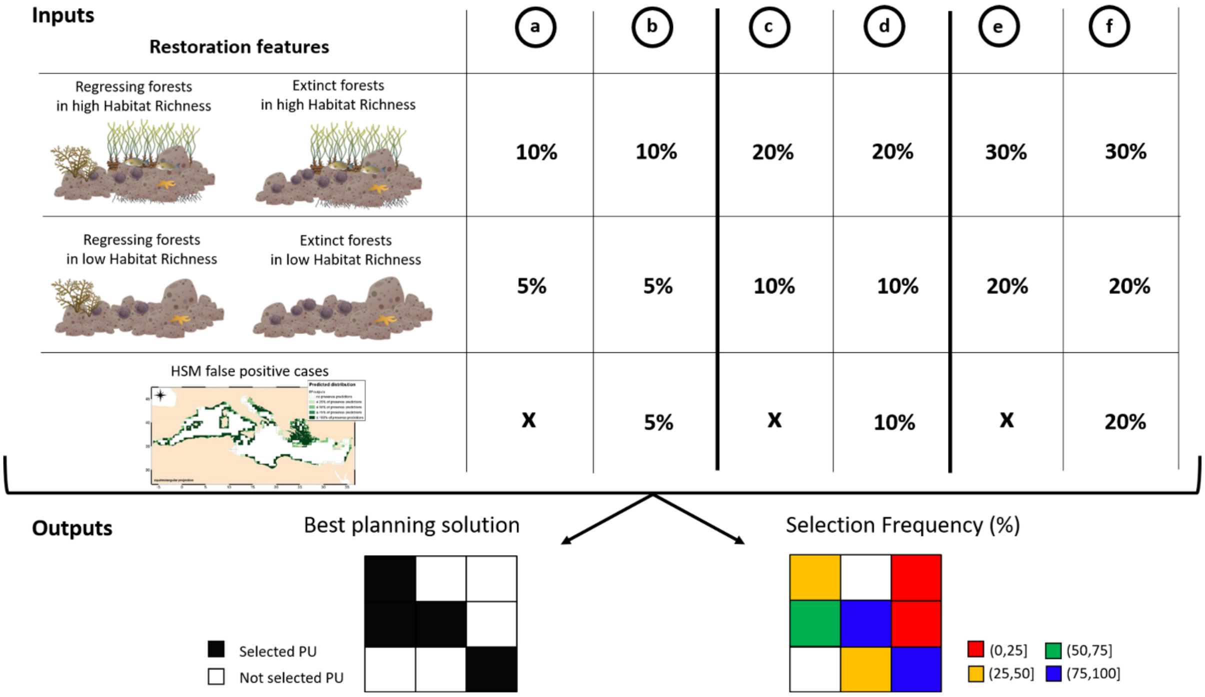

Figure 4.2 Graphical representation of Marxan inputs and outputs. Letters a, b, c, d, e and f correspond to the six scenarios while percentage numbers represent the targets set for each restoration feature in each scenario. In the scenarios b, d, and f, letter "x" means that the HSMf (i.e., the HSM false positive cases) were not taken into account.

4.4 Results

4.4.1 Planning Units and Restoration features

After comparing current and historical distribution of the selected species, 93 Rf and 762 Ef were identified. According to the level of HR surrounding the forests, restoration features were grouped in 88 Rf in low HR, 5 Rf in high HR, 735 Ef in low HR and 27 Ef in high HR. In addition, 232 areas were identified as the supplementary features HSMf (Fig. 4.3). Taken together, Rf and Ef were distributed over 310 PUs, of which only 88 were classified as suitable as potential restoration areas.

Other 232 suitable PUs corresponded to the HSMf, for a total of 320 PUs across the whole Mediterranean Sea actually suitable to be restored.

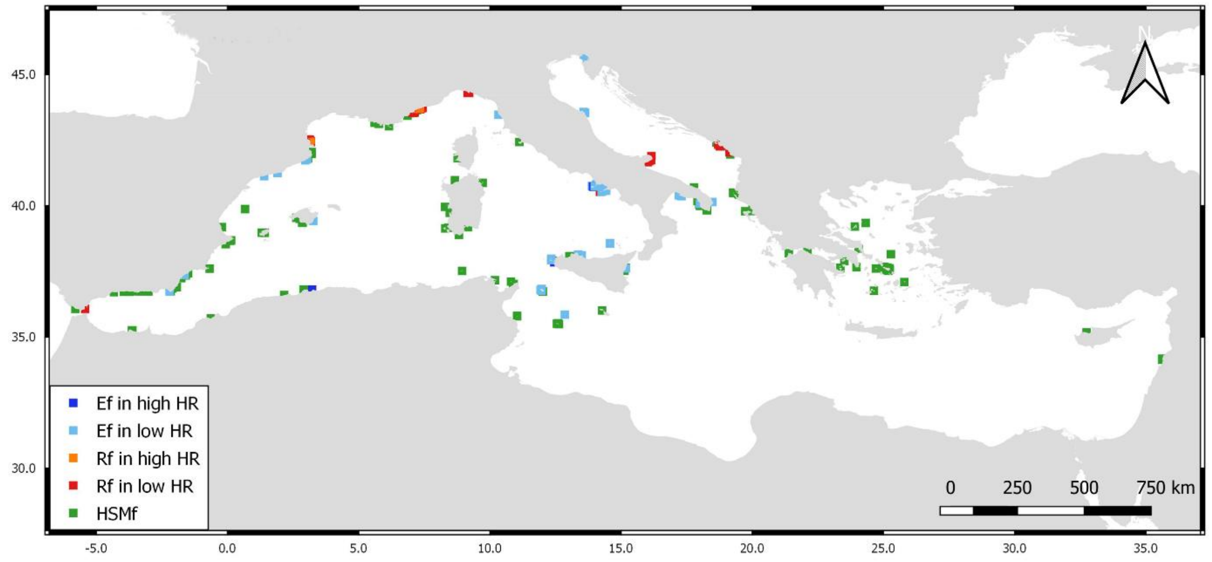

Figure 4.3 Map of the distribution of Regressing forests (Rf) and Extinct forests (Ef) in high and low Habitat Richness (HR) and of the Habitat Suitability Model false positive cases (HSMf) across the Mediterranean Sea.

4.4.2 Restoration scenarios and costs

For each scenario, we explored the best planning solution and the selection frequency of PUs, i.e., the number of times a PU is selected over the 100 runs as a measure of its relative priority. Costs associated to PUs ranged between € 1178.5 to € 1,261.7 (Supplementary Fig. 4.1).

In the scenarios *a* and *b* (i.e., restoring 10% of restoration features in high HR and 5% of those in low HR, respectively excluding and including the HSMf), all targets were reached. The best solution included 18 PUs as priority areas corresponding to an estimated cost of about € 21,225 for a restored surface area of 450 m^2, not considering (Fig. 4.4a) and considering (Fig. 4.4b) the HSMf.

In the scenario *c* (i.e., restoring 20% of restoration features in high HR and 10% of those in low HR, excluding the HSMf), 49 PUs were indicated as priority areas for a total cost of about € 57 817 for a restored surface area of 1,225 m^2, without reaching the target for the Ef in low HR (Fig. 4.4c). Conversely, in the scenario *d* (i.e., restoring 20% of restoration features in high HR and 10% of those in low HR, including the HSMf) all targets were met and 52 PUs were selected as priority areas in the best solution corresponding to an estimated cost of about € 61,408 for a restored surface area of 1,300 m^2 (Fig. 4.4d).

Finally, in both the scenario *e* and the scenario *f* (i.e., restoring 30% of restoration features in high HR and 20% of those in low HR, respectively excluding and including the HSMf), targets were not completely reached (Fig. 4.4e and 4.4f). The best solutions included 58 PUs as priority areas, corresponding to an estimated cost of about € 68,424 to restore 1,450 m^2 in the scenario *e* and 136 PUs corresponding to an estimated cost of about € 160,505 to restore 3,400 m^2 in the scenario *f*.

Table 4.1 summarizes the results obtained from the best planning solution in each scenario.

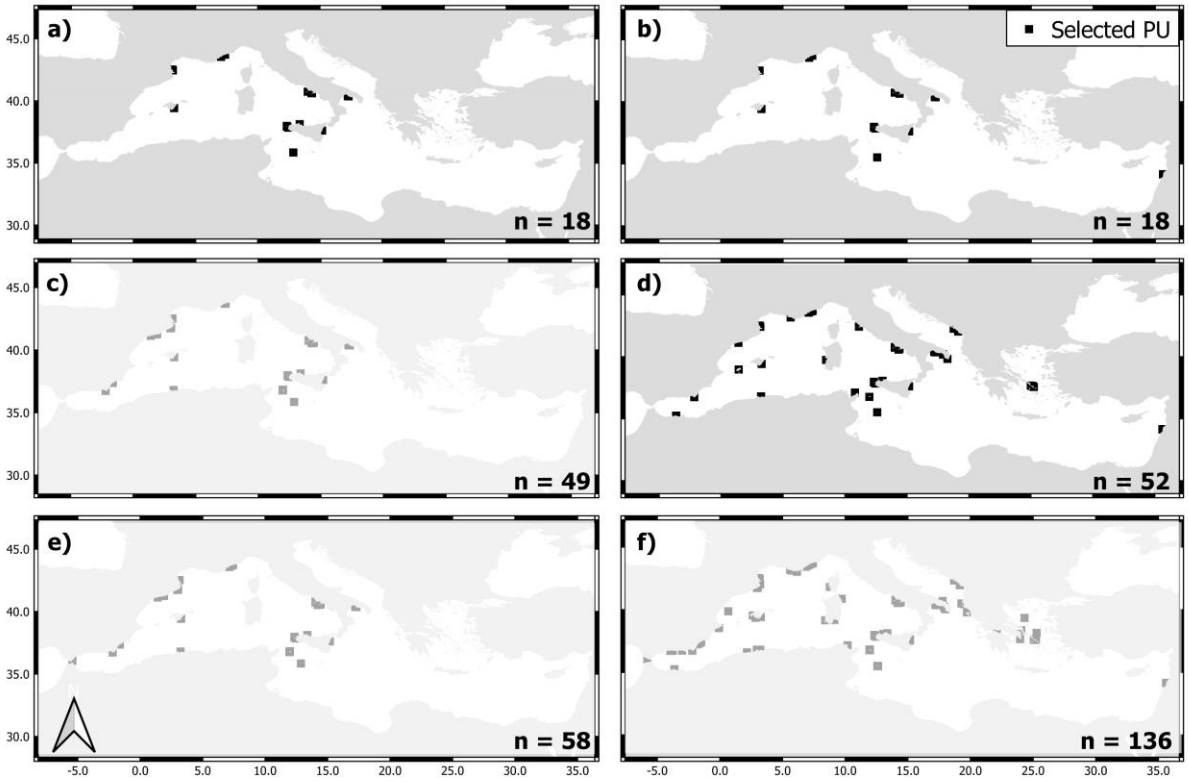

Figure 4.4 Maps of the best solution under each scenario. Figure 4.4a. restoring 10% of fucalean forests in high HR and 5% of those in low HR, excluding HSMf; Figure 4.4b. restoring 10% of fucalean forests in high HR and 5% of those in low HR, including HSMf; Figure 4.4c. restoring 20% of fucalean forests in high HR and 10% of those in low HR, excluding HSMf; Figure 4.4d. restoring 20% of fucalean forests in high HR and 10% of those in low HR, including HSMf; Figure 4.4e. restoring 30% of fucalean forests in high HR and 20% of those in low HR, excluding HSMf; Figure 4.4f. restoring 30% of fucalean forests in high HR and 20% of those in low HR, including HSMf.

Table 4.1 Results obtained from the best planning solution in each scenario. Last column on the right specifies if targets of the corresponding scenario are met ("Y") or not met ("N"). The restored surface area is calculated considering the condition under which 25 m^2 of regressing/extinct forests are restored in each PU.

Restoration scenario	N° of PUs	Costs	Restored surface area (m^2)	Targets met
a	18	€ 21,225	450	Y
b	18	€ 21,225	450	Y
c	49	€ 57,817	1,225	N
d	52	€ 61,408	1,300	Y
e	58	€ 68,424	1,450	N
f	136	€ 160,505	3,400	N

Taking into account the selection frequency for each PU in each scenario we identified the priority level of the selected areas: "low priority" for the areas selected between the 1% and 25% of the solutions; "moderate priority" for those selected between the 26% and 50%; "high priority" for those selected between the 51% and 75%; "top priority" for those selected between the 76% and 100%. The overlaps found comparing in pairs the PU selection frequency among the scenarios without HSMf and those with the HSMf (i.e., scenario *a* vs scenario *b*, scenario *c* vs scenario *d*, scenario *e* vs scenario *f*) represent the consensus among solutions (i.e., consensus areas) and was considered as a validation of the classification obtained from each single scenario. Figure 4.5 shows the distribution of the consensus areas found in the first pair of scenarios, i.e., those for which targets were completely reached. Among all the Mediterranean basin, consensus areas were found to be only 54 and spread out across France, Italy, Montenegro and Spain. Top priority areas corresponded to 5 PUs distributed as follows: 3 in Italy, 1 in France and 1 in Spain.

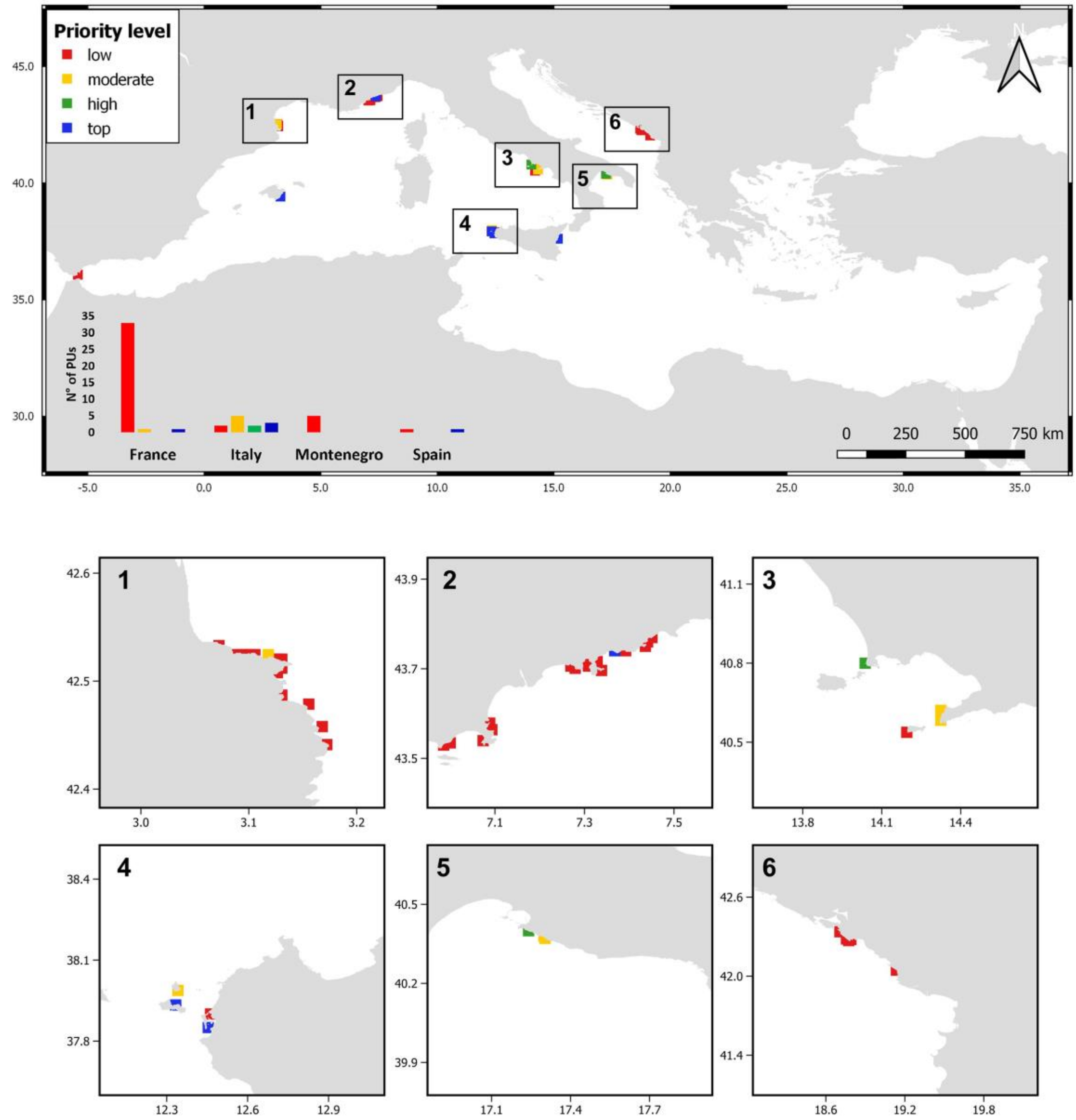

Figure 4.5 Distribution of the consensus areas grouped by priority level found comparing the scenario a to the scenario b. The bar chart expresses the distribution of consensus areas per country.

4.5 Discussion

Our results from literature analyses and data collection documented the occurrence of wide areas of regression and extinction in macroalgal forests across the Mediterranean Sea needing conservation and/or restoration. Most of the regressing and extinct forests occur in areas where the level of habitat richness is low, i.e., areas where ecosystem integrity is already compromised. The consequences of habitat loss on between-habitat diversity have been little explored. Airoldi et al. (2008) suggested that habitat loss causes a major reduction of spatial diversity in species distribution, a process also described as “biotic homogenization” (Bulleri et al., 2002; Thrush et al., 2006; Balata et al., 2007). In other words, restoration success should be higher in areas featured by high habitat diversity. This conclusion has critical consequences in terms of restoration since the successful recovery of one habitat might trigger the recovery of others present in the same area through positive species interactions and facilitation cascades effects (Eger et al., 2020), given that causes of extinctions are removed. The knowledge about drivers and consequences of habitat loss is key to improve the identification of criteria to be adopted in a restoration framework to properly select locations, methodologies and tools for increasing the potential of successful interventions.

While protection criteria have been widely discussed and commonly recognized by many international initiatives and organizations (i.e., UN SDG, EU Biodiversity Strategy for 2030, CBD post-2020) (Grorud-Colvert et al., 2021), restoration criteria are still scarcely investigated, resulting in a lack of shared guidelines to be pursued when implementing restoration. In the marine environment, the process of selecting priority areas for conservation is mainly focused on the ecological coherence of MPA networks, formally assessed through the following criteria: representativity – the MPA network should represent the range of marine habitats and species by protecting all major habitat types and associated biological communities present in the network boundaries, replication – all major habitats should be replicated and distributed throughout the network, connectivity – the MPA network should seek to maximize and enhance linkages amongst individual MPAs and adequacy – the MPA network should be of adequate size to deliver its ecological objectives and ensure ecological viability and integrity of species populations, communities and ecosystems (Gabrié et al., 2012; Giakoumi et al., 2012; UNEP/MAP RAC/SPA, 2014; Boero et al., 2016; Agnesi et al., 2017; COHENET, 2017; Fraschetti et al., 2018). In a restoration framework, criteria should be based on a cost-effective identification of the most suitable locations considering environmental and socio-economic constraints (McGowan et al., 2020), adopting as eligibility criteria the following principles: the historical presence of the habitat/species focus of restoration, the suitability of the current and the future environmental conditions together with the feasibility of the restoration intervention in terms of costs and availability of facilities (Cebrian et al., 2021). However, as far as connectivity, a stepping-stone approach can be adopted to enhance habitat connectivity so that restoration success can be further strengthen and upscaled. Our analysis showed that the number of potential sites from which selecting the priority areas was drastically reduced after the exclusion of unsuitable areas. These unsuitable areas derived from the HSM outputs and from the inclusion of the layer on the thermal anomalies hotspots, determining the lack of environmental requirements for restoration success. The data used for the analyses, together with the cost assessment to show the actual feasibility of restoration actions, are critical elements to inform the process of prioritization, providing guidance for the identification of suitable restoration targets. Clearly, the quality of data feeding the HSM is of paramount importance, and planning large-scale restoration interventions in absence of fine-scale information can seriously compromise outputs accuracy. In this regard, it is worthwhile to stress the urgency of collecting new field data filling gaps about the distribution of fucalean forests, especially along the southern and eastern Mediterranean coasts. Similarly, restoring in the present without considering the effects of climate changes and ocean warming increases the potential of failures (Gann et al., 2019; Wood et al., 2019; Verdura et al., 2021).

Setting different restoration scenarios allowed us to explore the consequences of adopting different targets developed a priori at a Mediterranean scale. In our analysis, only the first set of targets was completely fulfilled, meaning that, in the Mediterranean Sea, restoring 10% of the forests in high HR contexts and 5% of those in low HR is an achievable goal, even when the additional HSMf features (i.e., the areas potentially suitable for fucalean forests growth but where neither restoration features nor fucalean forests occur) were not considered. Increasing targets gradually raises the number of areas to be restored, amplifying the risk of including areas less suitable for restoration and for which a greater economic investment is required. For this reason, targets for scenarios e and f could not be met, even if the supplementary HSMf features were considered. In other words, restoring 30% of the forests in high HR contexts and 20% of those in low HR turned out to be unfeasible in the Mediterranean Sea, demonstrating that environmental constraints cannot be disregarded when setting restoration priorities and confirming the crucial role of the context of where the restoration activity is undertaken in determining restoration success (Fraschetti et al., 2021). The inclusion of the supplementary HSMf features, in the scenario d, had the effect of increasing the possibility to reach higher targets compared to the scenario c, but also increasing restoration costs and uncertainty in the restoration outcome. Thus, creating new forests in areas where the presence of a forest has never been documented and it is only suggested by the HSM predictions, would allow meeting more challenging restoration targets bearing the higher risk associated to these areas. In fact, this would represent the creation of a new habitat, a practice that has been considered as controversial (Boudouresque et al., 2021).

The selection frequency outcomes suggested instead that only few areas can be addressed as "top priority", meeting all the adopted restoration criteria. The consensus across scenarios (Micheli et al., 2013) about these areas can inform decision makers indicating the best candidate locations for macroalgal forests restoration. Restoration initiatives carried out in these areas translate into supporting the establishment of forests in locations where environmental conditions are currently suitable. They could also be addressed as climatic refugia for the presence of fucalean forests, since they have experienced thermal anomalies with a very low frequency.

Even though restoration priorities could widely vary depending on the prioritization criteria used (Strassburg et al., 2020), our case study demonstrates that introducing systematic conservation planning principles and tools in restoration projects is crucial to understand and define how much and where an ecosystem or habitat can be recovered, effectively managing our efforts and assessing the possibility of setting region-specific targets. Indeed, adopting MSP leads to accounting environmental constraints and socio-economic implications affecting restoration activities and the use of Marxan allows to allocate restoration targets identified a priori, combining spatial information from different sources. Future efforts should try to integrate site prioritization into marine spatial plans where restoration is co-optimized with protection, accounting for ecological, social and economic objectives to enhance system resilience.

4.5.1 Final recommendation

Despite the focus of the study was on the identification of criteria for macroalgal forests restoration, our intention was also to improve the spatial planning of future restoration efforts across marine habitats. Setting binding targets should be science-based and data-driven to ensure the effectiveness of restoration actions, as their cost in the marine environment is usually very high. More specifically:

- The collection of new information about current and historical species distribution, especially in data-poor regions, is critical for better understanding the drivers of changes, optimizing the identification of restoration sites. Our dataset underrepresented the southern and eastern Mediterranean biasing the spatial prioritization analysis. In addition, suitable sites were identified pooling different fucalean species together. Effective restoration requires knowledge at species level supporting the development of species-specific restoration plans.
- Since the context of *where* restoration activities are undertaken can be of greater relevance to a successful outcome than *how* (method) the restoration is carried out (Fraschetti et al. 2021), high-quality information on environmental variables and on the distribution and intensity of human threats is urgently needed to support the development of context-dependent restoration plans.
- An effort to advance the knowledge about the distribution and status of habitats is critically needed, associated to an improved understanding and interpretation on how to assess degradation (and thresholds of changes) across habitats. Fine-scale habitat mapping is largely lacking in the marine systems (Halpern et al. 2008, Dailianis et al. 2018, Fraschetti et al. 2018), limiting the consideration of the effects of the between-habitat diversity potentially affecting restoration outcomes. Updated information about the distribution and the status of marine habitats through coordinated monitoring across the Mediterranean countries should be a research priority for supporting future conservation and restoration initiatives.
- Refinement of restoration costs assessment is also recommended, since still large uncertainty can be observed, depending on the disparate restoration techniques, the target species and the involved countries (see Verdura et al. 2018, Tamburello et al. 2019, Gianni et al. 2020, Medrano et al. 2020). Also, selection of areas for restoration should be based on cost-effectiveness analysis to attain the maximum benefit with a limited budget. The development of standardized socio-economic assessments can support decision-makers in selecting the most cost-effective areas to be restored.

GENERAL CONCLUSIONS

In the last twenty years, overall information on the distribution of *Cystoseira s.l.* exponentially increased, as a result of growing attention to the conservation and restoration of these priority habitat-forming species. In parallel, the status of fucalean forests across all the Mediterranean coastlines is strongly characterized by a widespread declining trend with many documented local extinctions. Despite the proportion of protected populations has increased in recent years, it was observed that MPAs have limited efficacy in their conservation, since they cannot constrain regional- or basin-scale stressors such as eutrophication or water turbidity, nor mitigate global change effects. Nonetheless, the knowledge about the distribution of *Cystoseira s.l.* canopies is available for about only 14% of the Mediterranean coastline. Absence data are available only for the 2% of the basin. Research efforts (published and unpublished) differ among countries, and large data gaps have resulted in the eastern and southern part of the Mediterranean Sea, limiting our understanding of the real status of the forests and, as a consequence, the effectiveness of spatial restoration prioritization. Even for MPAs, available information are often insufficient. For instance, in Italy only about one third of MPAs have an updated overview of the canopy status, generally corresponding to the disappearance of most *Cystoseira s.l.* species. Furthermore, research has focused almost exclusively on intertidal reefs, this possibly due to the easiness of sampling and monitoring, improving the available information only for a few target species (mostly *C. compressa* and *E. amentacea*). Fine-scale presence-absence data about *Cystoseira s.l.* forests are still needed to improve the assessments of their status and distribution and to develop species-specific restoration plans.

Numerous variables, natural and anthropogenic, operating at different spatial scales, affect the distribution of fucalean forests, determining their regression or extinction. Identifying them is pivotal to plan future-oriented restoration (and conservation) actions. Cumulative anthropogenic stressors (i.e., impacts from mariculture, pollution, urbanization, agricultural or industrial activities) have been claimed to drive the loss of *Cystoseira s.l.* forests at Mediterranean scale and are negatively correlated with the percentage cover of fucalean species. Overgrazing by sea urchins, water warming and the combination of local stressors are advocated as drivers of decline at the scale of populations. These evidences stress the need of carefully considering scale-dependent ecological processes to support conservation and restoration. Yet, data on the environmental drivers are coarse and largely incomplete reducing our ability to assess the present and the future status of marine forests. As a result, drivers of extinction are actually unknown for the majority of the populations and cause-effects relationships are scarcely documented. A case-by-case identification of stressors is still far from being reached, although this step represents a necessary requisite to plan local conservation interventions, or to identify putative sites for restoration purposes, supporting the development of context-dependent restoration plans.

Despite gaps in knowledge are still present, combining all the available information resulted from each chapter/manuscript allowed to design a spatial planning where the consequences of adopting different restoration targets and criteria on restoration prioritization have been assessed at Mediterranean scale, including the consideration of climate changes. The historical presence of the habitat/species focus of restoration, the suitability of the current and the future environmental conditions together with the feasibility of the restoration intervention with regard to restoration costs and availability of facilities were indicated as the main criteria to be followed when selecting restoration location. Following these criteria, number of suitable sites for restoration is found to be very limited across the basin, and restoration targets can be only achieved when the recovery of 10% of regressing and extinct macroalgal forests is planned. Number of potential sites from which selecting the priority areas is drastically reduced when unsuitable areas are excluded from the analyses. Hence, increasing targets translates into including unsuitable areas for restoration in the spatial planning process, amplifying the risk of ineffective interventions and demonstrating that environmental constraints cannot be disregarded when setting restoration priorities, confirming the crucial role of the context of where the restoration activity is undertaken in determining restoration success.

The spatial planning analysis, building on the outcomes of the first three chapters of the thesis, supports macoralgal forests restorations in a fast-changing Mediterranean Sea, providing guiding principles and criteria to strengthen and upscaling the effectiveness of restoration actions across habitats.

Future Involvement

At present, I am carrying out a study exploring the potential for adaptation of recruits of *Gongolaria barbata* to marine heat waves (MHWs) and future ocean warming, for predicting upcoming changes, and improving current conservation and restoration strategies. A thermo-tolerance experiment was conducted to test physiological effects of short *vs* long MHWs occurring at different timing of the year. Manuscript with results from the analyses performed is already in preparation.

I will continue to study the topic addressed in the present work of thesis (i.e., marine ecosystem restoration and conservation under climate changes, with particular attention to the macroalgal forest restoration, and marine spatial planning) thanks to the involvement in the recently funded EU projects: ACTNOW (WP2-Task2.1 “Field and Laboratory Campaigns”; HORIZON-CL6-2021-BIODIV-01, No. 101060072), FORESCUE (WP3-Task3 “Mapping *Cystoseira* forests; Biodiversa+ 2021-134), LIFE DREAM (WP3-Task3.3 “Multi-criteria analysis”; LIFE-2021-SAP-NAT, No. 10107454) and MarinePlan (WP3-Task 3 “Prioritisation tool and scenario analysis”; HORIZON-CL6-2021-BIODIV-01, No. 101059407).

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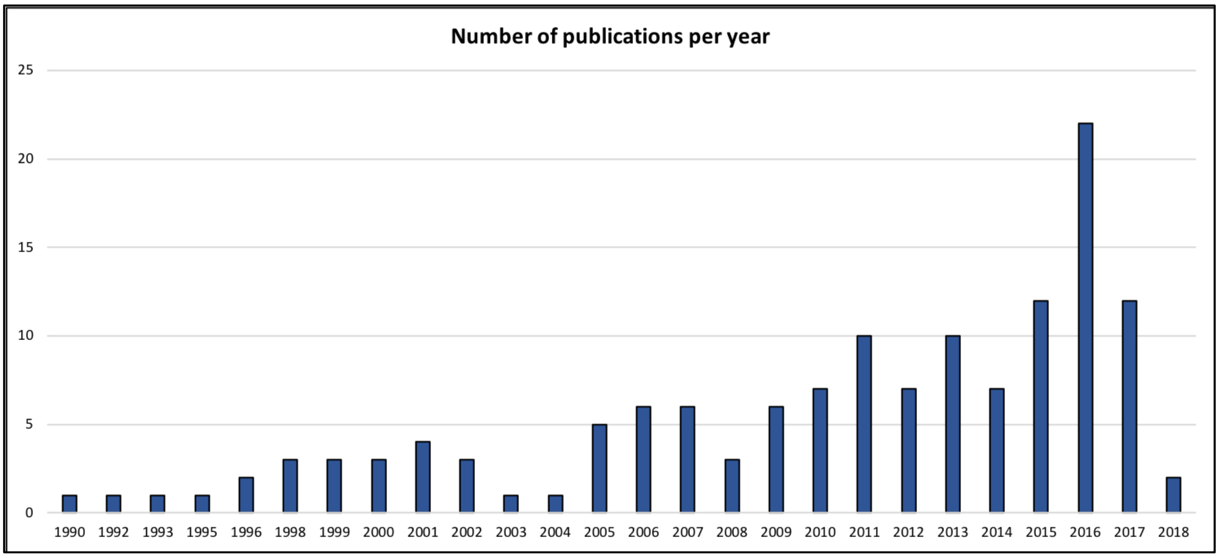
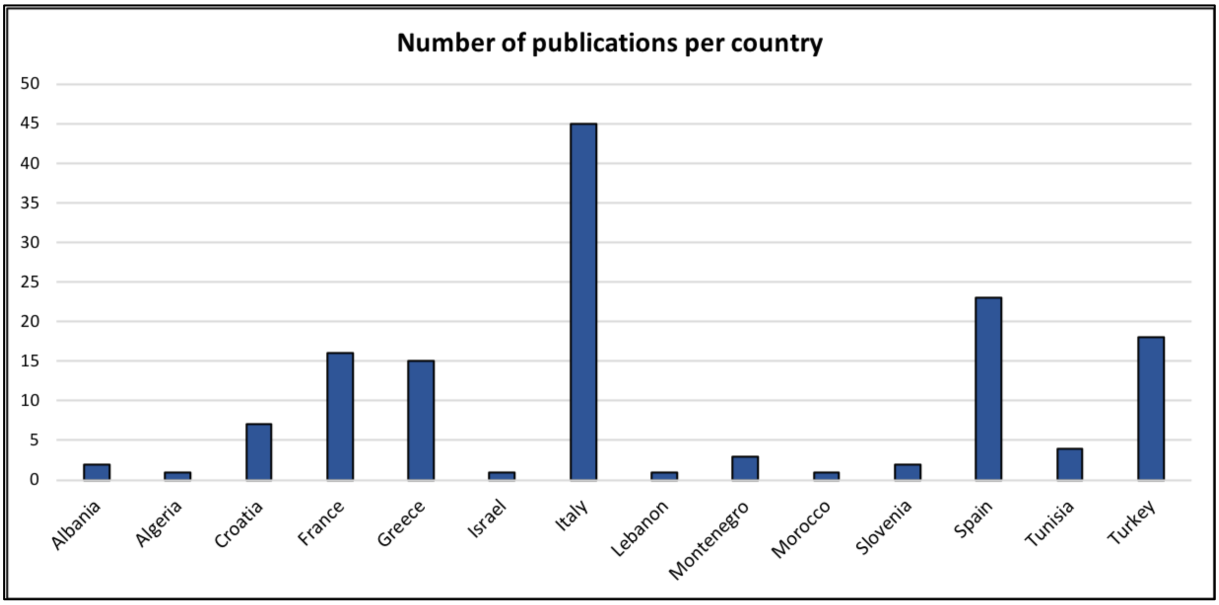
Appendix a: Supplementary Materials

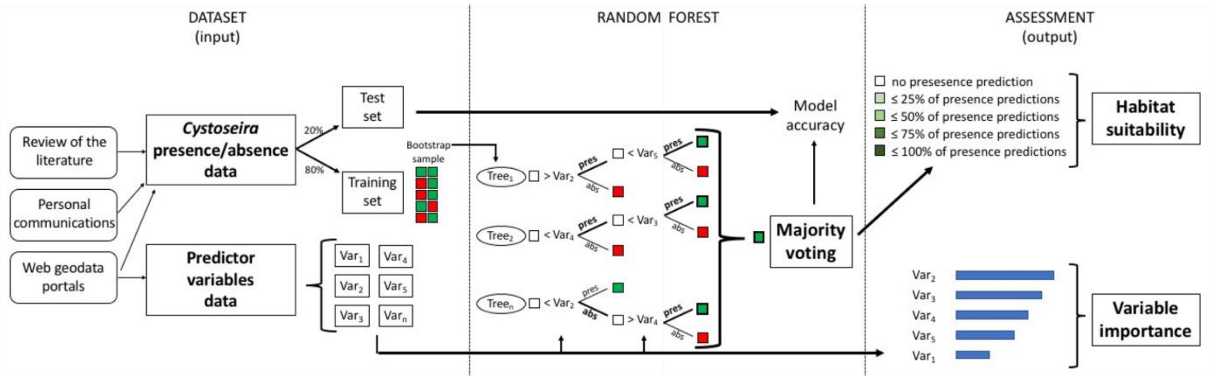
CHAPTER 1 Part I



Supplementary Figure S1.I.1 Number of publications per year.



Supplementary Figure S1.I.2 Number of publications per country.



Supplementary Figure S1.I.3 A flow diagram of the HSM development: the diagram shows the main steps from data access to habitat suitability model and the analysis of the variables relative importance. On the left side, the input datasets for the model development are presented ("Cytoseira presence/absence data" and "Predictor variables data"). The key sources for the data collection are also reported in the first three boxes of the figure. Then, we reported the proportion of Cystoseira data used to train the model (training set) and that used to test the model accuracy (test set). In the central section of the figure, a simple example of a RF is illustrated. In this synthesis, we show the structure of the trees which composes the RF and the way in which the predictions over each record are obtained. A total of 132 RFs, tuned with different combination of parameters, was trained in our work. The parameters tuned are: i) the number of trees in the forest and ii) the number of records in the terminal leaves of the trees (tuned in order to minimize the generalization error and to avoid overfitting); iii) the number of predictor variables randomly selected at each split which remains constant during the forest growth (with large effects on the strength of each individual tree and on the correlation between any pair of them). Then, the best model was selected by evaluating the model accuracy over the test set. In the end, the results of this model was used in order to assess areas classified as suitable with different probability level of Cystoseira occurrence ("Habitat suitability") and to measure the relative importance of the predictor variables ("Variable importance").

Supplementary Table S1.I.1 Integrated in the Supplementary Table S4.1

Supplementary Table S1.I.2 Integrated in the Supplementary Table S4.2

Supplementary Table S1.I.3 Types and number of collected records for each of the 22 Mediterranean countries: the table shows for which countries occurrence data were available and where information was reported at species level.

Country	Point	Line		Species info
		Presence	Absence	
Albania	8	25	1	y
Bosnia and Herzegovina	0	0	0	n
Algeria	6	16	0	y
Croatia	17.161	22	0	y
Egypt	0	0	0	n
France	226	1.773	38	y
Gibraltar	0	0	0	n
Greece	334	7	0	n
Israel	29	0	0	n
Italy	1.317	864	27	y
Lebanon	0	0	0	n
Lybia	0	0	0	n
Malta	30	0	0	y
Monaco	0	0	0	n
Montenegro	352	0	0	y
Morocco	11	29	0	y
Slovenia	2	0	0	y
Spain	192	10.200	498	y
Syria	4	0	0	n
Tunisia	62	17	0	y
Turkey-Cyprus	48	15	0	y
Tot	19.782	12.968	564	

Supplementary Table S1.I.4 Exhaustive list of all the *Cystoseira* species included in the dataset, with their respective number of occurrences.

Species	**line**	**point**	**TOT**
***C.* spp**	297	18.443	18.740
C. amentacea	5.947	277	6.224
C. mediterranea	5.838	56	5.894
C. compressa	857	353	1.210
C. brachycarpa	298	94	392
C. crinita	89	222	311
C. barbata	36	164	200
C. foeniculacea	22	143	165
C. montagnei	16	131	147
C. corniculata	9	50	59
C. algeriensis	27	13	40
C. zosteroides	10	19	29
C. tamariscifolia	23	2	25
C. sauvageauana	2	37	39
C. humilis	13	11	24
C. elegans	9	7	16
C. schiffneri	10	1	11
C. sedoides	10	0	10
C. squarrosa	3	4	7
C. dubia	3	0	3
C. crinitophylla	2	0	2

CHAPTER 1 Part II

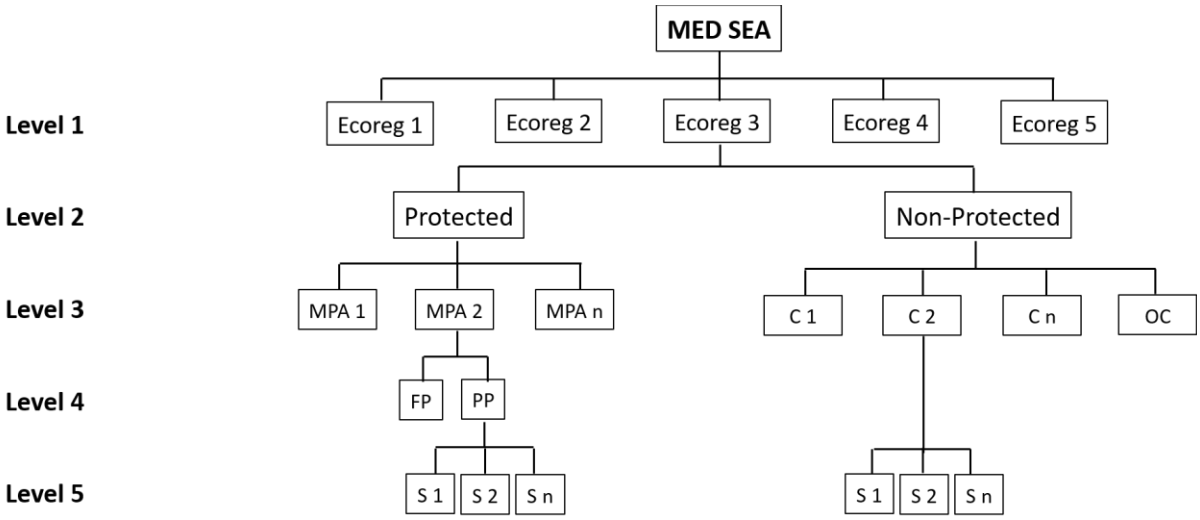


Figure S1.II.1. Nested structure used in the analysis. Level 1, Marine Biogeographic Ecoregions: Mediterranean (MED) areas of homogeneous characteristics and conditions, clearly distinct from adjacent systems; Level 2, Condition: Protected *vs* Non-Protected; Level 3, SAU: Marine Protected Areas (MPAs) considered for each ecoregion for the Protected branch (MPA 1, MPA 2, MPA n) and the relative reference locations for the Non-Protected branch (C 1, C 2, C n). Additional Non-Protected areas (OC = Other Controls), far from the MPAs, were also included; Level 4, Protection: Fully Protected (FP), Partially Protected (PP); Level 5, Sites: sampling sites (S 1, S 2, S n) inside each area.

Table S1.II.1. In the Table, both the Control (C) and the Other Control (OC) areas, grouped by biogeographical ecoregions, are listed. For each of them, it is also indicated whether a Natura 2000 Site is present or not in the area. The Natura 2000 database was acquired from the EEA data service (European Environment Agency, https://www.eea.europa.eu/data-and-maps/data/natura-11). Furthermore, the NEAT assessment, considering both the real extent and the two buffer analyses, is reported. For each area, the Cumulative Human Impact score (CI), based on Halpern et al. (2015) was also included, with values normalized between 0-1, together with the main sources of pressures included in that study**.**

Ecoregion	SAU name	C/OC areas	Natura 2000 Sites	Main pressures	NEAT assessment – real extent	NEAT assessment – buffer 5 and 10 km
Adriatic Sea	Torre Guaceto	C	No	CI: 0.50 Ocean acidification, ocean pollution	moderate	moderate
	Telascica	C	Yes	CI: 0.11 Ocean pollution	moderate	moderate
	Brijuni	C	Yes	CI: 0.55 Ocean acidification, ocean pollution	poor	poor
	Kepi i Rodonit	OC	No	CI: 0.18 Invasives	poor	poor
	Kepi i Lagjit	OC	No	CI: 0.24 Tourism	high	high
Aegean Sea	Kas	C	No	CI: 0.10 Invasives	moderate	moderate
	Cyclades	OC	No	CI: 0.50	poor	poor

				Tourism		
	Crete	OC	No	CI: 0.80 Tourism	poor	poor
	CW Aegean	OC	No	CI: 0.61 Ocean acidification, ocean pollution	moderate	moderate
	N Aegean	OC	No	CI: 0.47 Ocean acidification, sea surface temperature	moderate	moderate
	NE Aegean	OC	No	CI: 0.42 Ocean acidification, sea surface temperature	moderate	moderate
	Dodekanissa	OC	Yes	CI: 0.03 Ocean acidification	poor	poor
Ionian Sea	Zakynthos	C	Yes	CI: 0.44 Ocean acidification, sea surface temperature	poor	poor
	Porto Cesareo	C	No	CI: 0.44 UV radiation, ocean acidification	good	good
	Karaburun-Sazan	C	No	CI: 0.14 Tourism	good	good
	Ksamil	OC	No	CI: 0.09 Invasives	good	good
	Kepi I Stillos	OC	No	CI: 0.55 Ocean acidification, shipping	moderate	moderate
Tunisian plateau	Isole Pelagie	C	Yes	CI: 0.08 Tourism	good	good
Western Med	Cinque Terre	C	Yes	CI: 0.43 Ocean acidification, ocean pollution	good	good
	Portofino	C	No	CI: 0.08 Tourism	good	moderate
	Bergeggi	C	No	CI: 0.64 Ocean acidification, ocean pollution	good	good
	Asinara	C	Yes	CI: 0.37 UV radiation, ocean acidification	moderate	moderate
	Tavolara	C	No	CI: 0.21 Tourism	good	good
	Capo Carbonara	C	No	CI: 0.55 UV radiation, ocean acidification	poor	poor
	Egadi	C	No	CI: 0.50 UV radiation, ocean acidification	bad	bad

	Es Freus	C	Yes	CI: 0.43 Ocean acidification, ocean pollution	moderate	moderate
	Menorca	C	Yes	CI: 0.27 Ocean acidification, shipping	high	high
	Cabo de Palos	C	Yes	CI: 0.27 Ocean acidification, shipping	moderate	moderate
	Medes	C	Yes	CI: 0.44 UV radiation, ocean acidification	good	moderate
	Cap De Creus	C	Yes	CI: 0.70 Tourism	moderate	good
	Bonifacio	C	Yes	CI: 0.60 Tourism	poor	poor
	Banyuls	C	Yes	CI: 0.52 UV radiation, ocean acidification	moderate	moderate
	Cote Bleue	C	Yes	CI: 0.57 Ocean acidification, ocean pollution	moderate	poor
	Cap Roux	C	Yes	CI: 0.57 Ocean acidification, ocean pollution	poor	poor
	Diano Marina	OC	No	CI: 0.68 Ocean acidification, shipping	high	high
	Sanremo	OC	Yes	CI: 0.54 Ocean acidification, shipping	high	high
	Ventimiglia	OC	No	CI: 0.65 Ocean acidification, shipping	high	high
	Ospedaletti	OC	Yes	CI: 0.11 Coastal urbanization	good	good
	Capo Mortola	OC	Yes	CI: 0.91 Ocean acidification, shipping	poor	poor
	Invrea	OC	No	CI: 0.20 Artisanal fishing	poor	poor
	Sestri Levante	OC	Yes	CI: 0.43 Tourism, shipping	good	moderate
	Vesima	OC	No	CI: 0.47 Tourism, shipping	poor	poor
	Capo Comino	OC	No	CI: 0.17 Tourism, shipping	poor	poor
	Capo Montesanto	OC	Yes	CI: 0.44 UV radiation, ocean acidification	poor	poor

	Castelsardo	OC	No	CI: 0.64 UV radiation, ocean acidification	moderate	moderate
	Montirusso	OC	No	CI: 0.47 Ocean acidification, shipping	moderate	moderate
	Punta Niedda	OC	No	CI: 0.41 UV radiation, Ocean acidification	moderate	moderate
	Calafuria	OC	No	CI: 0.62 UV radiation, ocean acidification	poor	poor
	Castiglioncello	OC	No	CI: 0.54 UV radiation, ocean acidification	bad	bad
	Ajaccio	OC	Yes	CI: 0.57 Ocean acidification, ocean pollution	moderate	moderate
	Cap Martin	OC	No	CI: 0.19 Coastal urbanization, tourism	poor	poor
	Blanes	OC	No	CI: 0.45 UV radiation, ocean acidification	poor	poor
	Lloret del Mar	OC	Yes	CI: 0.50 Ocean acidification, shipping	moderate	moderate

Table S1.II.2 Ecosystem components and associated Indicators. For each Indicator the values identified as the worst, the moderate/good (threshold between good and not good) and the best conditions are included in the Table. To include the non-linearity behavior of some of the Indicators, the optional bad/poor and poor/moderate values were set out. References and details for the identification of all the values are shown.Threshold values for *P. oceanica* shoot density were set according to Pergent et al. (1999), allowing the identification of different thresholds across depths. For the abundance of canopy algae (*Cystoseira* spp.), we adopted the values provided by Thibaut et al. (2017), who indicate 50% cover as a threshold value for good condition and 5% for poor condition. Values of erect macroalgae (e.g., Dictyotales) were set according to the canopy values identified in each SAU: for instance, if the maximum value for canopy in a SAU was 30%, the best value considered for erect algae was 70% and the relative threshold 35%. This decision was motivated by the observed pattern of replacement of canopies by erect algae across the Mediterranean Sea (Sala et al., 2012). The threshold value for sea urchin biomass was set considering the trophic conditions of the SAUs (Boada et al., 2017) and their geographical area (Eastern basin (Giakoumi et al., 2012); Western basin (Sala et al., 2012)). Regarding the density of sea urchins, thresholds were set according to Thibaut et al. (2017), Guidetti & Sala (2007), Hereu (2004) and Galasso et al. (2015). Turf and barren cover (considered indicators of bad status) were also included in the analysis, and thresholds were set following the conclusions of Thibaut et al. (2017) and Rindi et al. (2017). For fish, a dataset independent from the one used in this study was adopted (Sala et al., 2012). We assumed that fish biomass within well enforced MPAs could represent GES. More specifically, we set the values from the best enforced buffer zone (i.e. the Partially Protected Area, PPA) recorded in Sala et al. (2012) as the threshold to achieve GES for fish biomass. The rational to use a reference value taken from a successful PPA (and not from a no-take Fully Protected Area) is that GES should be achieved in EU waters open to human uses, including sustainably regulated fishing.

Ecosystem Component		Indicators	Descriptors	References for threshold, worst and best values	Worst value	Bad/Poor (optional)	Poor/Mod (optional)	Mod/Good (threshold)	Good/High (optional)	Best value	Note
Macroalgae	Canopy algae	Abundance of macroalgae [total % cover]	D1, D4, D5, D6	Thibaut et al. (2017), Rindi et al. (2017)	0	5	10	50		100	The most conservative values of Thibaut et al. were adopted where the threshold is indicated in terms of 50% cover and poor conditions in terms of 5%. The 25% indicated by Rindi et al. falls within the two conditions. Keeping the 50% as a threshold value corresponds to a more precautionary approach.
	Erect algae	Abundance of macroalgae [total % cover]	D1, D4, D5, D6	Thibaut et al. (2017)	0	5	30	50		100	See the above. These values depict the scenario in which canopies were historically absent.
		Abundance of macroalgae [total % cover]	D1, D4, D5, D6	Thibaut et al. (2017)	Complementary to canopy values						These values depict the scenario in which canopies were present.

Fish		Biomass of total fish assemblage (fish, cephalopods and macrocrustaceans) [gr/125m^2]	D1, D4	Sala et al. (2012)	0			4250		14875	The dataset by Sala et al. (Sala et al., 2012) was adopted. We assumed that conditions within well-enforced MPAs could represent GES. More specifically, we set the values from the best enforced buffer zone (i.e. the Partially Protected areas) recorded in Sala et al. (Sala et al., 2012) as threshold to achieve GES for fish biomass.
		Biomass of carnivore and high-level predator fishes [gr/125m^2]	D1, D4	Sala et al. (2012)	0			3580		12756	See above "Biomass of total fish assemblage"
Seagrass	*P. oceanica*	Seagrass shoot density (values potentially changing according to depth) [shoot density/m^2]	D1, D4, D6	Pergent et al. (1999)	0 (matte)		822	934		1158	1 m depth
					0 (matte)		413	525		749	5 m depth
					0 (matte)		237	349		573	10 m depth
					0 (matte)		134	246		470	15m depth
					0 (matte)		61	173		397	20m depth
					0 (matte)		4	116		340	25m depth
					0 (matte)			31		255	30m depth
Sea urchins		Biomass of sea urchins [gr/m^2]	D1, D4, D6	Boada et al. (2017), Hereu et al. (2012)	5000	700	225	50		0	Western Mediterranean, low nutrient concentration
					5000	1800	500	85		0	Western Mediterranean, high nutrient concentration

				Giakoumi et al. (2012), Sala et al. (2012)	170		72	30		0	Eastern Mediterranean
		Density of sea urchins [individuals/m^2]	D1, D4, D6	Thibaut et al. (2017), Guidetti & Sala (2007), Hereu (2004), Galasso et al. (2015)	30			5		1	
Turf and Barrens		Abundance of turf [total % cover]	D1, D6	Thibaut et al. (2017), Rindi et al. (2017)	100		95	5		0	
		Abundance of barren [total % cover]	D4, D6	Thibaut et al. (2017), Rindi et al. (2017)	100		95	5		0	

Box S1.II.1. Ecological thresholds

The use of NEAT requires knowing how marine systems respond to incremental changes in human uses or environmental conditions with the identification of ecological thresholds for each indicator. Theory and empirical evidence have shown that, in response to stressors, marine systems can undergo different patterns of change. Despite the fact that linear, additive, and gradual ecological responses to impacts of human uses or natural drivers can occasionally occur (Fig. S1.II.2a), the prevalence of strong nonlinearities has been recently shown (Selkoe et al., 2015; Wernberg et al., 2016) (Fig. S1.II.2b and c). Nonlinear relationships have one or more curves or points of rapid change called ecological thresholds or "tipping points" beyond which the system reorganizes into an alternative regime, resulting in large, and sometimes abrupt, changes in ecosystem structure, functions, and benefits to people. The rapid reorganization of a system from one relatively unchanging state over time to another is called regime shift. When a system is close to an ecological threshold, a large ecological response results from a relatively small change in a driver (Fig. S1.II.2b). Critical transitions (non-linear with hysteresis) are special cases in which alternative states can co-occur under the same set of human pressures or environmental conditions. Graphically, this results in a S-shaped bifurcation curve in which the recovery pathway of an ecosystem differs from its pathway of degradation and the condition variable must improve well beyond the critical threshold to enable the recovery of the original state (Suding & Hobbs, 2009). In Fig. S1.II.2c, in the yellow area, solid lines show paths along stable equilibria featured by reinforcing feedbacks contributing to the stability of the system in each condition, whereas the dashed line marks unstable equilibria separating the two alternative states. Between the alternative states, unstable conditions exist.

Although the theory is clear, current attempts to concretely identify thresholds of change of very common metrics such as macroalgae abundance, sea urchin density and fish biomass are limited by our current scientific knowledge. Our comprehension about thresholds of change in Mediterranean coastal communities, including benthic assemblages and fish, is limited to a few studies carried out on single species (e.g. *Cystoseira amentacea*) (Benedetti-Cecchi et al., 2015; Rindi et al., 2017) and only occasionally this issue has been addressed at ecosystem level, considering that changes in one group are often linked to changes in other groups (Thibaut et al., 2017). There is the awareness that thresholds should be context-dependent but replicated experiments under different environmental conditions have been rarely carried out (Boada et al., 2018). Large scale assessments (Sala et al., 2012) and routine monitoring are key to assess ecological thresholds, together with the consideration that threshold-based systems within smaller geographic areas are more likely to have fine scale outcomes.

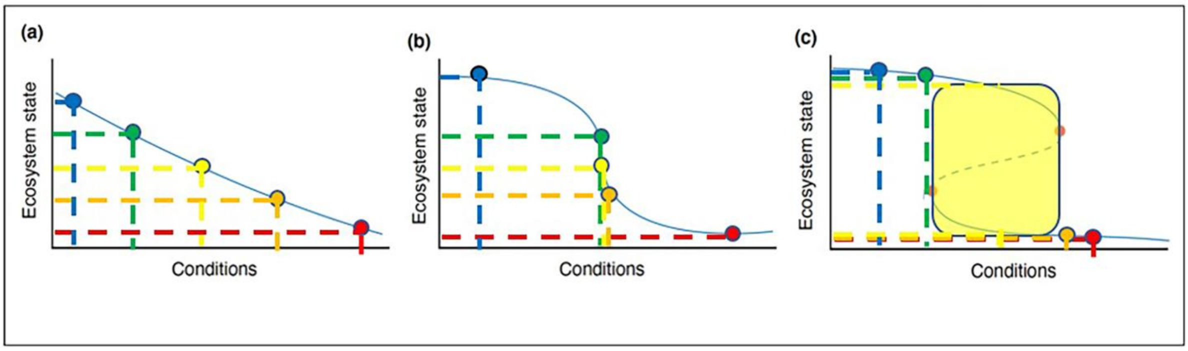


Figure S1.II.2. Different ways in which an ecosystem can respond to change in conditions (modified from Scheffer and Carpenter, 2003: Catastrophic regime shifts in ecosystems: linking theory to observation.)

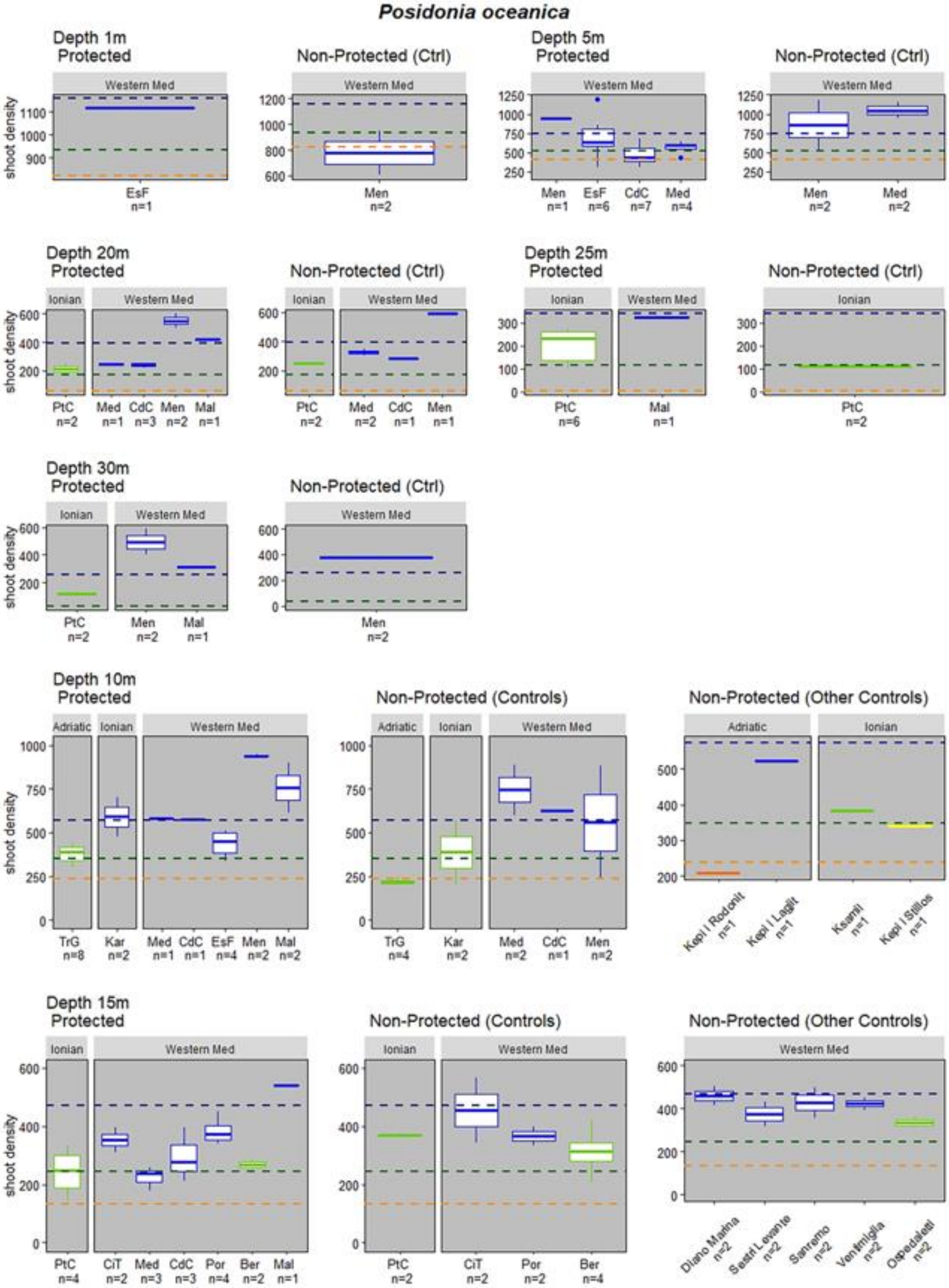


Fig. S1.II.3 The figure shows the distribution of the shoot density (shoot number/m^2) values across sites ("n" = number of sites in each SAU) collected for *P. oceanica* at each depth considered, grouped by protected and non-protected areas and ecoregions. Selected thresholds are also included as dashed lines: orange = poor/moderate

(822 at 1 m depth; 413 at 5 m depth; 237 at 10 m depth; 134 at 15 m depth; 61 at 20 m depth; 4 at 25 m depth); green = moderate/good (934 at 1 m depth; 525 at 5 m depth; 349 at 10 m depth; 246 at 15 m depth; 173 at 20 m depth; 116 at 25 m depth; 31 at 30 m depth); blue = best value (1158 at 1 m depth; 749 at 5 m depth; 573 at 10 m depth; 470 at 15 m depth; 397 at 20 m depth; 340 at 25 m depth; 255 at 30 m depth). Colors of the boxplots corresponds to the outcomes of the NEAT analyses

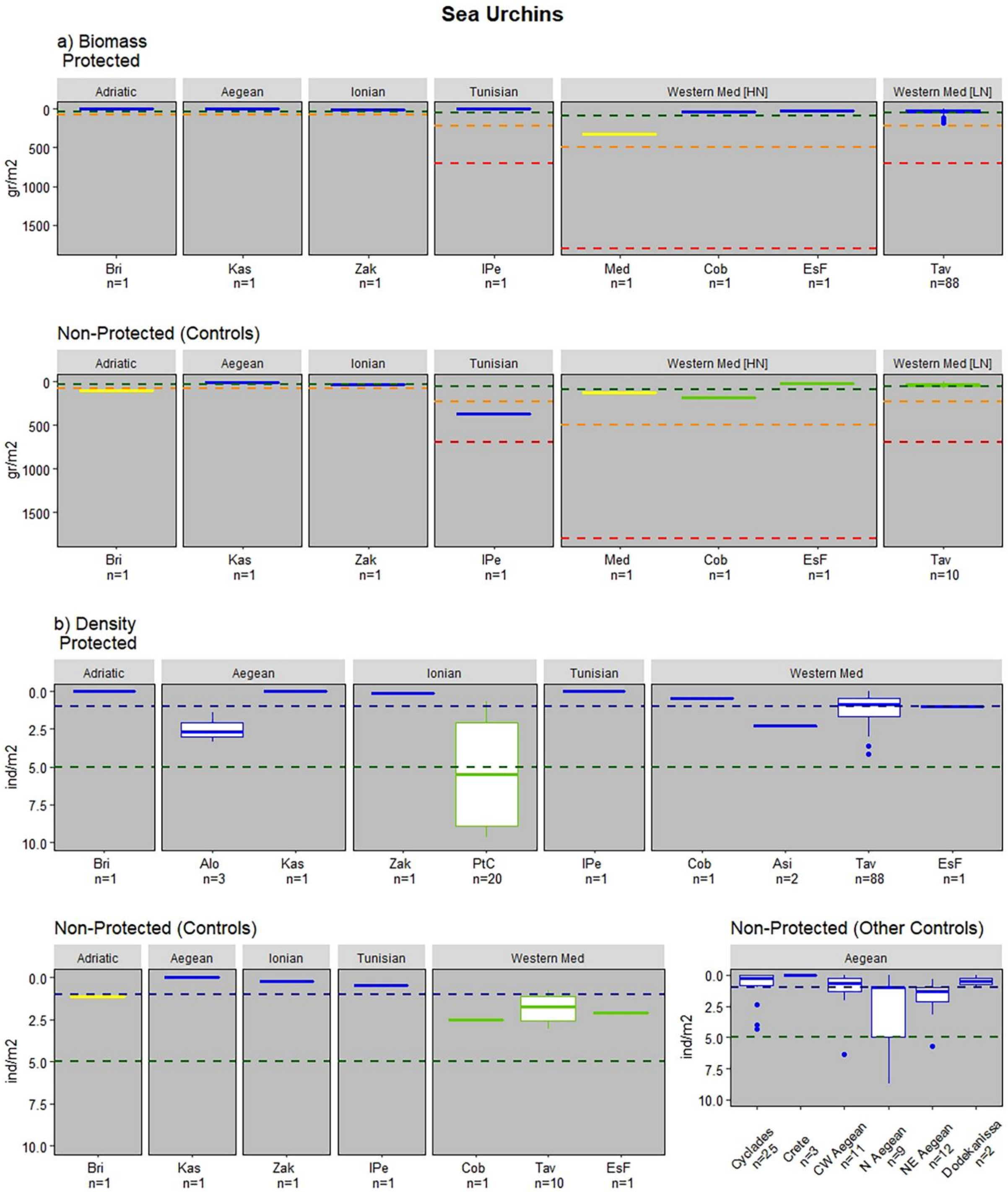


Fig. S1.II.4. The figure shows the distribution of the biomass (a) and density (b) values across sites ("n" = number of sites in each SAU) collected for sea urchins, grouped by protected and non-protected areas and ecoregions. Selected thresholds are also included as dashed lines: Fig. S1.II.4a – red = bad/poor (700 gr/m^2 for the Western Mediterranean at low nutrient concentration; 1800 gr/m^2 for the Western Mediterranean at high nutrient concentration); orange = poor/moderate (225 gr/m^2 for the Western Mediterranean at low nutrient concentration; 500 gr/m^2 for the Western Mediterranean at high nutrient concentration; 72 gr/m^2 for the Eastern Mediterranean); green = moderate/good (50 gr/m^2 for the Western Mediterranean at low nutrient concentration; 85 gr/m^2 for the Western Mediterranean at high nutrient concentration; 30 gr/m^2 for the Eastern Mediterranean). Fig. S1.II.4b – green = moderate/good (5 ind/m^2); blue = best value (1 ind/m^2). Colors of the boxplots corresponds to the outcomes of the NEAT analyses.

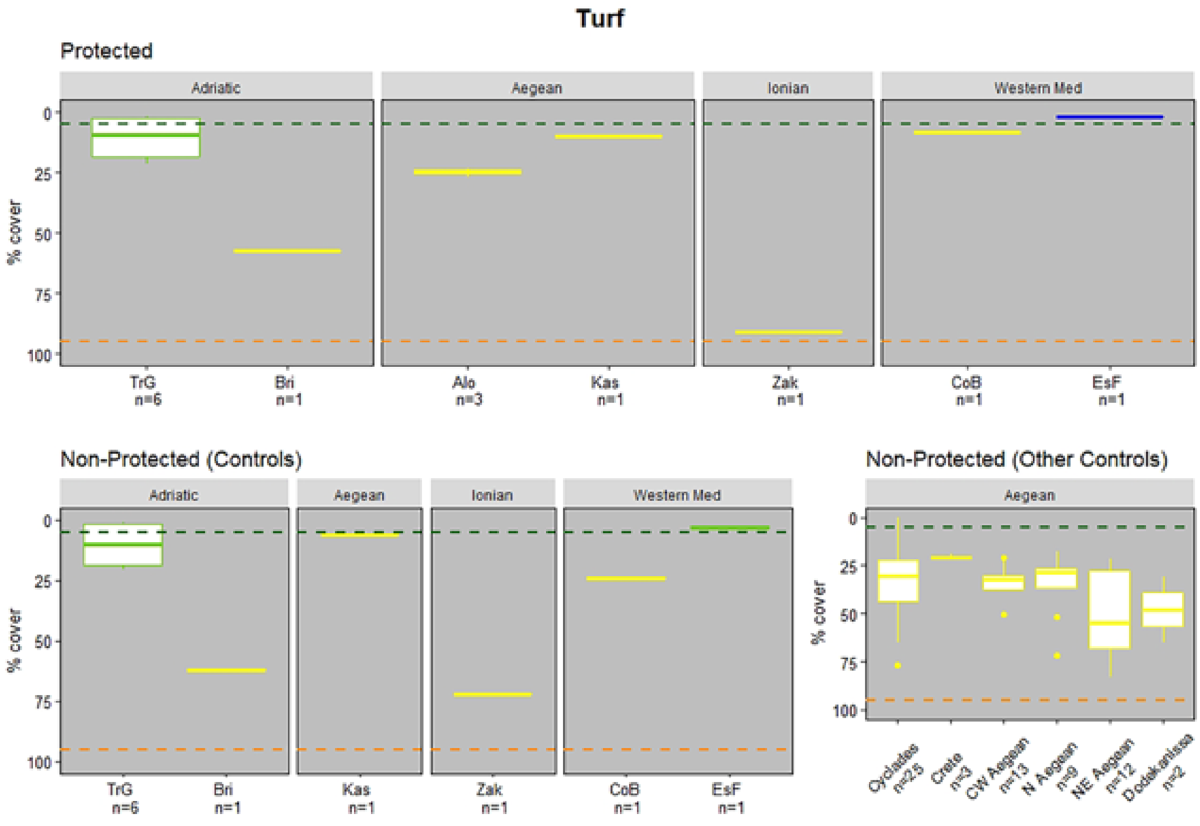


Fig. S1.II.5. The figure shows the distribution of the cover % values across sites ("n" = number of sites in each SAU) collected for turf grouped by protected and non-protected areas and ecoregions. Selected thresholds are also included as dashed lines: orange = poor/moderate (95 %); green = moderate/good (5%). Colors of the boxplots corresponds to the outcomes of the NEAT analyses.

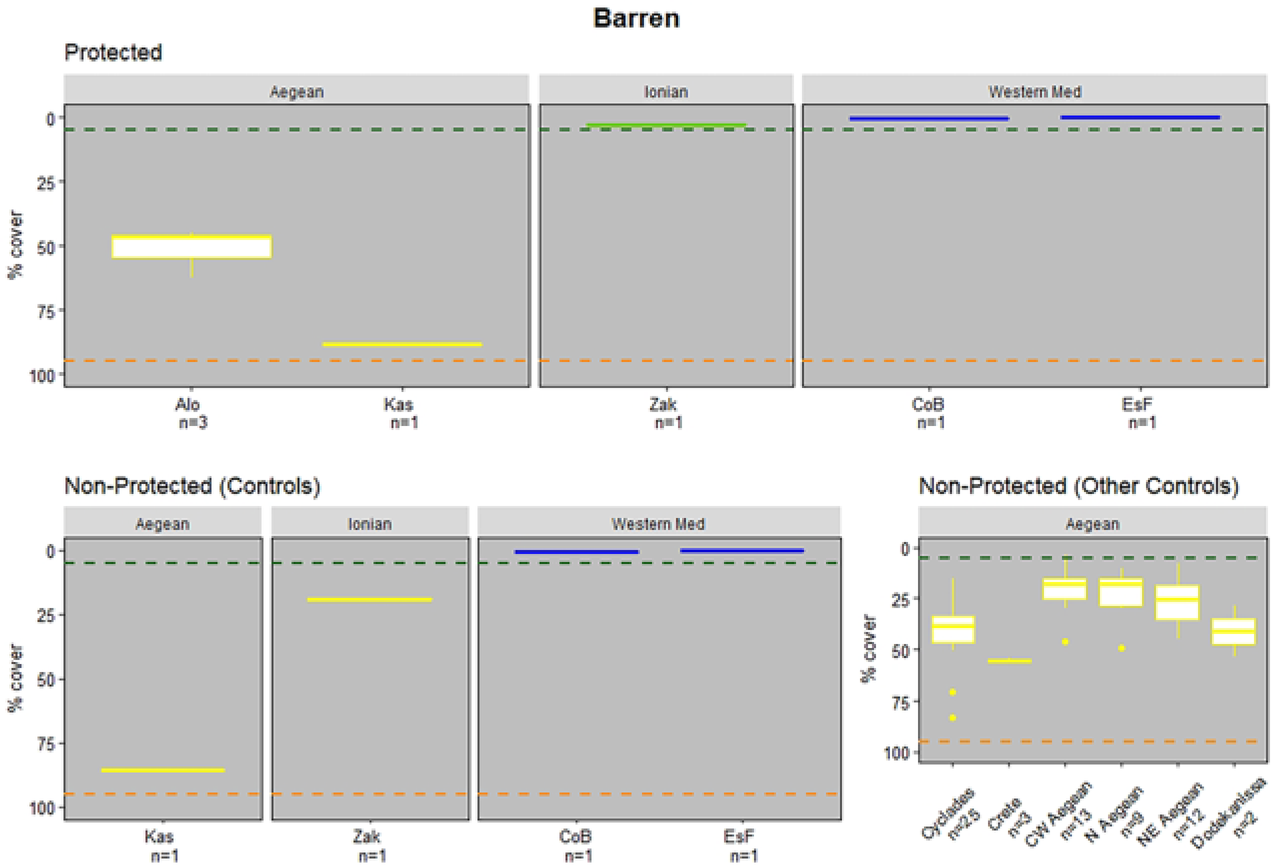


Fig. S1.II.6. The figure shows the distribution of the percentage cover values across sites ("n" = number of sites in each SAU) collected for barren grouped by protected and non-protected areas and ecoregions. Selected thresholds are also included as dashed lines: orange = poor/moderate (95%); green = moderate/good (5%). Colors of the boxplots corresponds to the outcomes of the NEAT analyses.

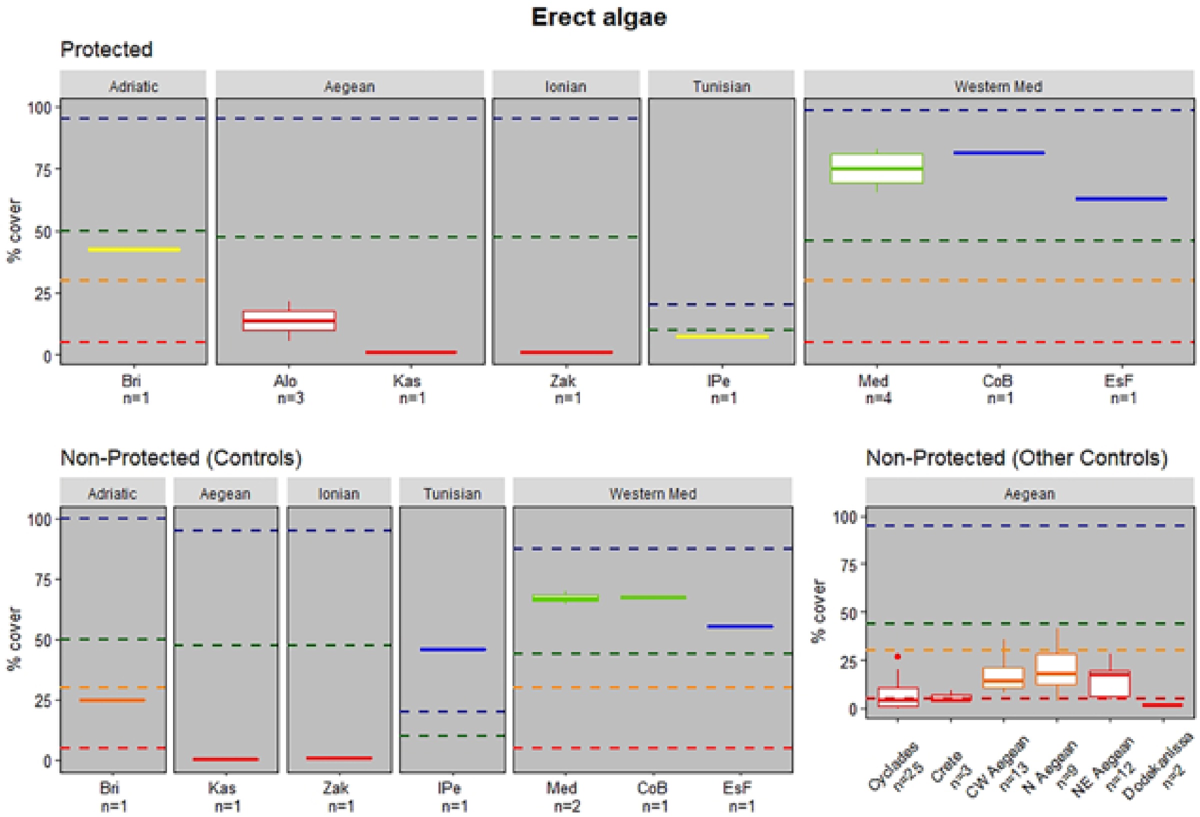
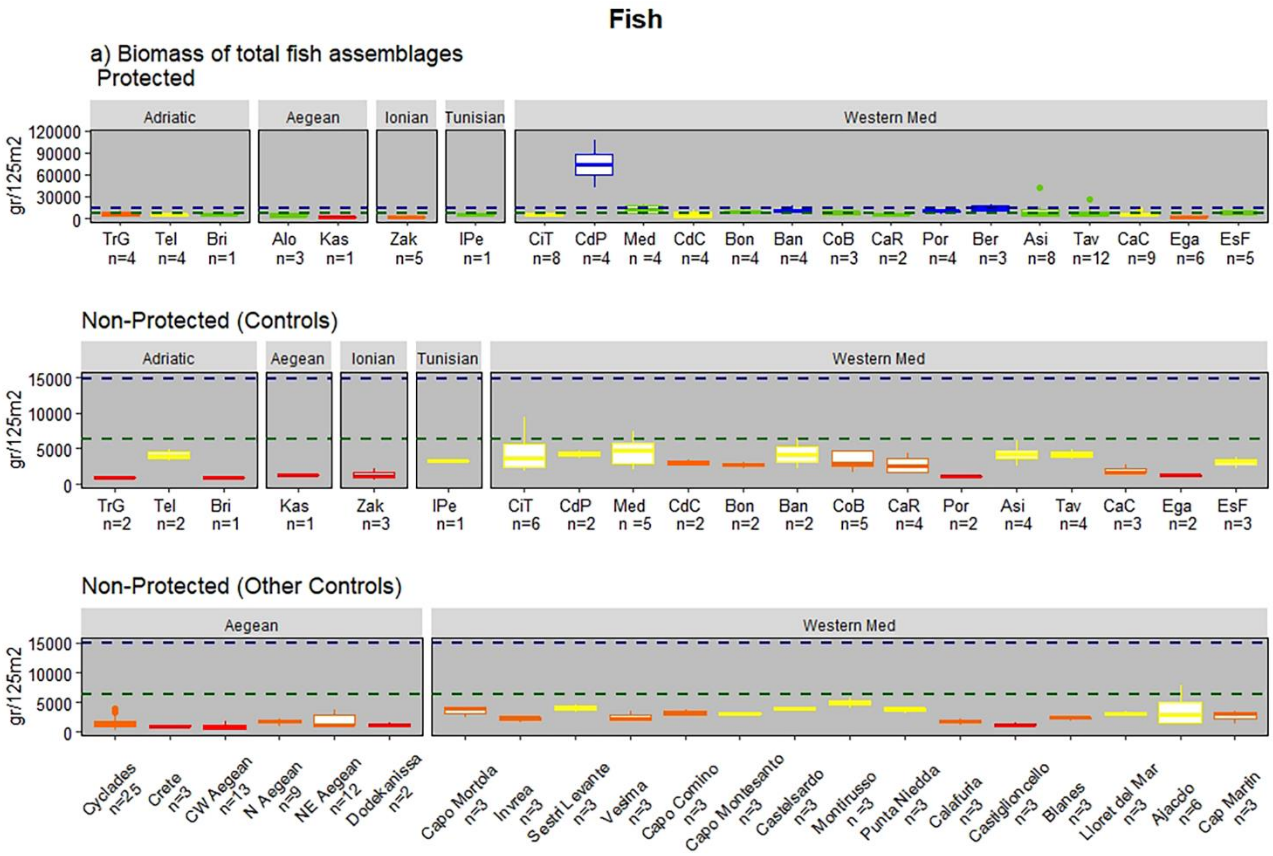


Fig. S1.II.7. The figure shows the distribution of the percentage cover values across sites ("n" = number of sites in each SAU) collected for erect algae grouped by protected and non-protected areas and ecoregions. Selected thresholds are also included as dashed lines: red = bad/poor (5% for areas in which canopies are historically absent; % complementary to canopy value for the areas in which canopies are present); orange = poor/moderate (30% for areas in which canopies are historically absent; % complementary to canopy value for the areas in which canopies are present); green = moderate/good (50% for areas in which canopies are historically absent; % complementary to canopy value for the areas in which canopies are present). Colors of the boxplots corresponds to the outcomes of the NEAT analyses.


Fish
a) Biomass of total fish assemblages
Protected
Adriatic
Aegean
Ionian
Tunisian
Western Med
gr/125m2
120000
90000
60000
30000
0
TrG n=4
Tel n=4
Bri n=1
Alo n=3
Kas n=1
Zak n=5
IPe n=1
CiT n=8
CdP n=4
Med n =4
CdC n=4
Bon n=4
Ban n=4
CoB n=3
CaR n=2
Por n=4
Ber n=3
Asi n=8
Tav n=12
CaC n=9
Ega n=6
EsF n=5
Non-Protected (Controls)
Adriatic
Aegean
Ionian
Tunisian
Western Med
gr/125m2
15000
10000
5000
0
TrG n=2
Tel n=2
Bri n=1
Kas n=1
Zak n=3
IPe n=1
CiT n=6
CdP n=2
Med n =5
CdC n=2
Bon n=2
Ban n=2
CoB n=5
CaR n=4
Por n=2
Asi n=4
Tav n=4
CaC n=3
Ega n=2
EsF n=3
Non-Protected (Other Controls)
Aegean
Western Med
gr/125m2
15000
10000
5000
0
Cyclades n=25
Crete n=3
CW Aegean n=13
N Aegean n=9
NE Aegean n=12
Dodekanissa n=2
Capo Mortola n=3
Invrea n=3
Sestri Levante n=3
Vesima n=3
Capo Comino n=3
Capo Montesanto n=3
Castelsardo n=3
Montirusso n =3
Punta Niedda n=3
Calafuria n=3
Castiglioncello n=3
Blanes n=3
Lloret del Mar n=3
Ajaccio n=6
Cap Martin n=3

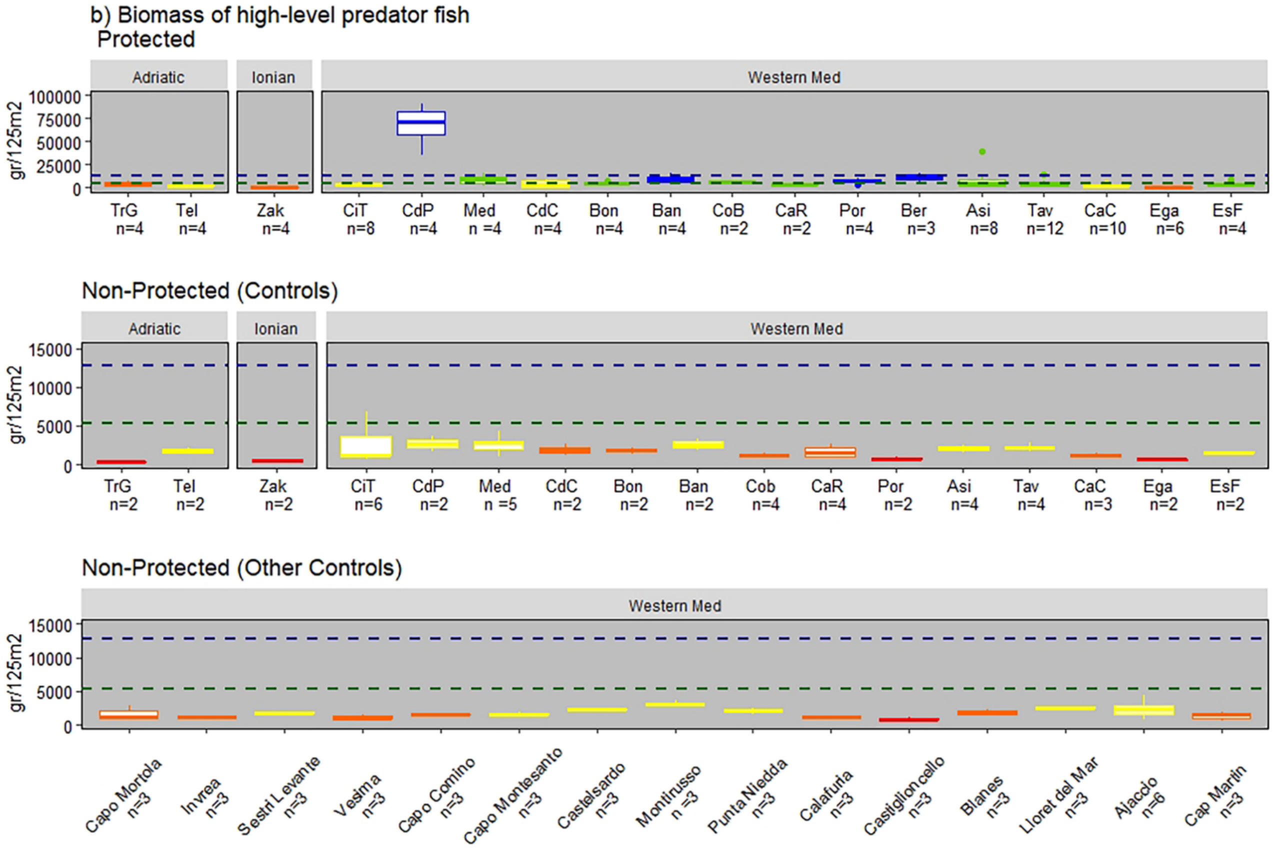


Fig. S1.II.8. The figure shows the distribution of the biomass of total fish assemblages (a) and of high-level predator fish (b) across sites ("n" = number of sites in each SAU) collected for fish, grouped by protected and non-protected areas and ecoregions. Selected thresholds are also included as dashed lines: Fig. S.8a - green = moderate/good (4250 gr/125 m^2); blue = best value (14875 gr/125 m^2). Fig. S.8b - green = moderate/good (3580 gr/125 m^2); blue = best value (12756 gr/125 m^2). Colors of the boxplots corresponds to the outcomes of the NEAT analyses.

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CHAPTER 2

Location description

Open coasts of Eastern Macedonia (G_OCEM): It is located in the North Aegean Sea and consists mainly of soft substrata of plutonic origin alternating with a rocky infralittoral zone dominated by big boulders and cobbles down to 3-4 m depth. The shoreline is exposed to southern winds causing certain periods of particularly intense wave action, leading to coast alterations. *Ericaria barbatula* is the dominant fucalean species in this location, forming patchy but in several parts continuous, dense populations from 0 to 1-2 meters depth. *Ericaria barbatula* forms assemblages together with *C. corniculata* in the most exposed parts of the shore. Prominent seasonal *C. compressa* patches during late spring to early summer co-occur with *E. barbatula* in the rest of the meadows.

Open coasts of Central Macedonia (G_OCCM): Sithonia peninsula is located in the middle of the three "fingers" of Chalkidiki and western of Athos Mountain. The studied location belongs to two different Natura 2000 sites (Oros Itamos – Sithonia -GR1270002, Platanitsi – Sykia: Akr. Rigas – Akr. Adolo GR1270009). The landscape is very rugged and mountainous, with many small villages and many, sometimes deserted sandy beaches. The southeastern part of the location is marked by steep rocky coasts composed mainly of granitic structures followed by steep sloping rocky habitats of bedrock and boulders in the infralittoral zone. In the northeastern part of the Sithonia peninsula, shallow infralittoral habitats covered with sandy and muddy substrate alternate with rocky habitats marked by various fucalean species. The biotopes are exposed to intense wave action. The upper limits of the infralittoral zone are marked by different fucalean species, with *C. corniculata* being the dominant one, which is widespread on the coasts.

Sazan Island-Karaburuni peninsula (AL_SZK): It is located in the western part of Vlora bay, along the Albanian Adriatic Sea coast on the Mediterranean Sea. The whole area is a Specially Protected Area of Mediterranean Importance (SPAMI). The northern part of the studied location consists of small isolated bays sheltered from waves covered with continuous dense vegetation of *E. amentacea*. In the central part of the location the seabed slides vertically up to 40m depth. In these parts, fucalean seaweed forests are patchy and poorly developed. Steep rocky bottom, consisting of rock and boulders, can also be found in the western part of Sazan Island covered with *E. crinita.* The dominant fucalean species found in the area are *E. amentacea*, *G. barbata, C. compressa* and *E. crinita.* Shallow hard substratum there is on the western side of Karaburuni peninsula and Sazan Island.

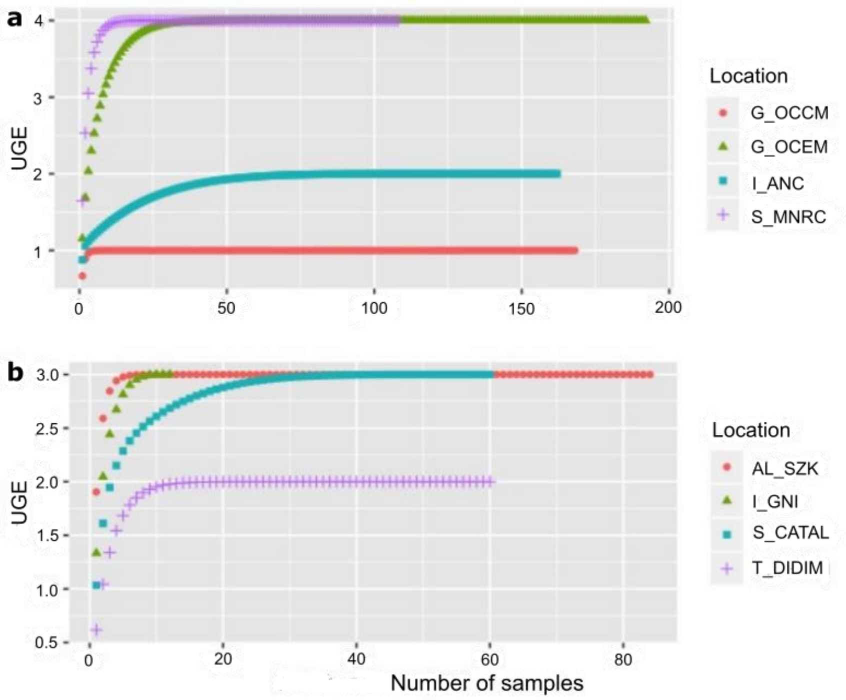
Ancona, Italy (I_ANC): The Conero Riviera is located south of the city of Ancona, on the western coast of the Adriatic Sea, Italy. The location includes the shores of Grotta Azzurra, Passetto di Ancona, Scalaccia, Portonovo, Due Sorelle. The majority of shores in the studied location faces North-Northeast. The area is exposed to northern and north-easterly winds, except the part of the shore located at the Due Sorelle, which is relatively sheltered from northern and north-easterly wind due to the two large rocks from which the site is named. The biotopes are generally exposed to strong wave action. In the northern part of the Riviera, high cliffs alternate with beaches largely covered with concrete. On most of the Riviera, the sea bottom gently slopes until 2-4 meters in depth. The dominant fucalean species in the area is *C. compressa*, with dense but patchy populations. *Gongolaria barbata* is also present but in lower amounts; it tends to be more abundant in partially sheltered conditions.

Gulf of Naples (I_GNI): The location sampled is Ischia Island, located at the northwestern edge of the Gulf. Rocky coasts are mostly high; remains of volcanic structures lead to a system of natural rock-pools in the mesolittoral zone, sheltered from wave action. The biggest of these rock-pools stretch has a 8.6 m^2 surface, with an average depth of 30 cm and a maximum depth of about 1 m. The seafloor is gently sloping, with large boulders protruding close to the surface. The highest values of continuity and coverage for the fucalean species were detected inside the rock-pools. Dominant species in the upper infralittoral zone are *C. compressa*, *E. amentacea* and *E. crinita*. Other fucalean taxa recorded in the area are: *G. elegans*, *C. compressa* var. *pustulata* and *E. mediterranea*.

Fornells -Menorca (S_MNRC): Menorca is one of Spain's Balearic Islands located in the North-Western Mediterranean, recognized as a UNESCO Biosphere Reserve since 1993. Fornells is a sheltered bay with small bedrocks and boulders occasionally covered with sand sediment alternate with a sandy substrate in the shallow infralittoral zone. The shallow infralittoral zone hosts outstanding forests of *C. compressa* subsp. *pustulata* and *G. barbata*. Another species reported in the area is *C. foeniculacea* f. *tenuiramosa*. Although Menorca's forests are quite well preserved, a recent increase in certain anthropogenic pressures, such a urbanization, agriculture and tourism activity may be seen as a future threat.

Catalonia coasts (S_CATAL)**:** The Cap de Creus of Port de la Selva marine reserve is very rugged with steep rocky coasts; rocky coves and sandy beaches follow the cliffs. In the Port de la Selva, *Ericaria crinita* forms a continuous belt between the surface and 0.6 m depth in a sheltered rock pool connected to the open sea. Still, its geomorphological structure somewhat restricts the exchange of water in/out of the pool. *Ericaria mediterranea* forms a continuous belt from 0 m to 1 m depth out from rock pools continuously spreading some km apart. Cala Estreta is a part of the Natura 2000 site Castell-Cap Roig. The coast is steep with rocky coves and sandy beaches. The shallow marine bottoms of rocky substratum are dominated by the largest remaining population (ca. 40 m^2) of *E. crinita* in northern Catalonia. *E. crinita* thrives among rocks within a shallow, open, and semi-exposed mixed substrate habitat, also between the surface and 0.6 m depth, where water exchange is unrestricted. *E. mediterranea* forms a continuous belt from 0 to 0.7 m depth, at all wave-exposed sites, with coverages > 90%.

Didim, Turkey (T_DIDIM): It is located in the Aegean Sea, southwestern coasts of Turkey. It is recognized as a nature park since 2011 and is bounded by Büyük Menderes Delta at the north and the small touristic village of Didim at the south. The studied shore faces West, exposed to winds and high wave action. The coast consists mainly of soft substrata alternating with rocky habitats of bedrock and boulders in the sublittoral zone. The location presents a diverse habitat for macroalgal species, dominated by species such as *Padina pavonica* and *Jania rubens*. In the area, *C. foeniculacea* and *E. crinita* form patchy and scattered communities. Seagrasses *Posidonia oceanica* and *Cymodocea nodosa* are also found in the locality. In total, 64 marine macrophytes (macroalgae and angiosperm) taxa were reported from Didim, and this locality was found as a high ecological status class.



Supplementary Figure S2.1 Comparison of the species–accumulation curves estimating the richness of fucalean species produced by Ugland et al. (2003) T–S estimator (UGE) at different sampled locations (a, b plots). See Figure 1 for more information.

Supplementary Table S2.1 Indirect stressor categories and their scores used to calculate MA-LUSI.

Indirect stressor categories			
Urban (codes 11)	**Commersial & Industrial (codes 12, 13)**	**Agriculture (codes 21-24)**	**Score**
<10%	<10%	<10%	0
10-33%	10-30%	10-40%	1
33-66%	>=30%	40-60%	2
>=66		>=60%	3

Supplementary Table S2.2 Direct stressors of category A and their scores used to calculate MA-LUSI.

Direct stressors of category A		
Mariculture	**Sediment nutrient release**	**score**
Absence	Absence	0
Presence	Presence	1
Presence in shallow waters	Presence in shallow waters	2

Supplementary Table S2.3 Direct stressors of category B and their scores used to calculate MA-LUSI.

Direct stressors of category B			
Sewage outfall	**Irregular Fresh Water inputs**	**Harbour**	**score**
absence	absence	absence	0
>50.000 Inhabitant	presence	marina/fishing	1
>500.000 Inhabitants	moderate	big marina/fishing	2
<500.000 Inhabitants	high	commercial harbour	3

Supplementary Table S2.4 Oceanographic/hydrological stressors and their scores used to calculate MA-LUSI.

Oceanogrphic/Hydrological stressors							
stability of the water column	**score**		**Background trophic status**	**score**		**Confinement**	**score**
Low fresh water inputs	1		Oligotrophic to Mesotrophic	1		Straight	1
Medium Freshwater inputs	1.25		Oligotrophic	0.75		Convex	0.75
High Freshwater inputs	1.5					Concave	1.25

Supplementary Table S2.5 Frequency (%) and abundance at the transect scale of fucalean species recorded in the study.

Species	Frequency (%)	Mean abundance (%)
Cystoseira compressa	57.80	30.15
Ericaria crinita	37.61	19.79
Ericaria amentacea	26.61	24.79
Cystoseira corniculata	15.60	29.39
Ericaria barbatula	13.76	38.84
Gongolaria barbata	9.17	17.50
Cystoseira foeniculacea	5.50	17.33
Cystoseira foeniculacea f. *tenuiramosa*	5.50	2.12
Ericaria mediterranea	3.67	25.29
Cystoseira compressa subsp. *pustulata*	2.75	9.56

Gongolaria elegans	0.92	3.33

Supplementary Table S2.6 Pairwise comparisons of fucalean species cover (%) between six different locations using a test of homogeneity of dispersion (PERMDISP, *P(perm): 0.01*).

Groups	*t*	*P(perm)*
(G_OCEM, G_OCCM)	8.185	0.001
(G_OCEM, I_ANC)	7.442	0.001
(G_OCEM, AL_SZK)	1.556	0.220
(G_OCEM, S_CATAL)	2.407	0.077
(G_OCEM, S_MNRC)	4.180	0.003
(G_OCCM, I_ANC)	0.517	0.677
(G_OCCM, AL_SZK)	13.174	0.001
(G_OCCM, S_CATAL)	13.869	0.001
(G_OCCM, S_MNRC)	19.183	0.001
(I_ANC, AL_SZK)	9.226	0.001
(I_ANC, S_CATAL)	9.160	0.001
(I_ANC, S_MNRC)	13.274	0.001
(AL_SZK, S_CATAL)	1.813	0.110
(AL_SZK, S_MNRC)	3.675	0.002
(S_CATAL, S_MNRC)	1.379	0.313

Supplementary Table S2.7 Mean cover (%) of the fucalean species in the eight studied locations.

Location	*Cystoseira compressa*	*Gongolaria barbata*	*Ericaria amentacea*	*Ericaria barbatula*	*Cystoseira corniculata*	*Ericaria crinita*	*Ericaria mediterranea*	*Cystoseira foeniculacea*	*Cystoseira foeniculacea* f. *tenuiramosa*	*Cystoseira compressa* subsp. *pustulata*	*Gongolaria elegans*
Open coasts_Eastern Macedonia	6.3±3.12	0	0	38.62±5.21	5.96±3.32	5.18±2.89	0	0	0	0	0
Open coasts_Central Macedonia	0	0	0	0	28.77±3.93	0	0	0	0	0	0
Conero Riviera_Ancona	54.81±4.88	2.53±1.33	0	0	0	0	0	0	0	0	0
Gulf of Naples_Ischia	40.42±12.74	0	30±17.37	0	0	5.42±4.88	0	0	0	0	0
Sazan Karaburuni MPA - Sazan	12.31±2.83	0	25.57±2.96	0	0	18.39±3.57	0	0	0	0	0
Catalonia coasts	0	0	0	0	0	18.65±8.08	32.83±12.99	0	0	0	0.67±0.67
Menorca	0	14.85±5.05	0	0	0	18.6±9.34	0	0	1.33±0.55	3.19±1.62	0
Didim, Aegean coasts of Turkey	0	0	0	0	0	8±4.06	0	20.33±4.76	0	0	0

Supplementary Table S2.8 Codes of the samples at three spatial scales: Location, site, transects

A/A	Location	Forest	Transect
1	G_OCEM	Gr_A	GR_1
2			GR_2
3			GR_3
4			GR_4
5		Gr_B	GR_5
6			GR_6
7			GR_7
8			GR_8
9		Gr_C	GR_9
10			GR_10
11			GR_11
12			GR_12
13		Gr_D	GR_13
14			GR_14
15			GR_15
16			GR_16
17	G_OCCM	Gr_F	GR_18
18			GR_19
19			GR_20
20		Gr_G	GR_17
21			GR_21
22			GR_22
23			GR_23
24		Gr_H	GR_24
25			GR_25
26		Gr_I	GR_26
27			GR_27
28			GR_28
29		Gr_J	GR_29
30			GR_30
30			GR_30
31	I_ANC	I_A	I_1
32			I_2
33			I_3
34			I_4
35		I_B	I_5
36			I_6
37			I_7
38			I_8
39		I_C	I_9
40			I_10
41			I_11
42			I_12
43			I_13
44			I_14
45			I_15
46			I_16
47			I_17
48			I_18
49			I_19
50			I_20
51		I_D	I_21
52			I_22
53			I_23
54		I_E	I_24
55			I_25
56			I_26
57			I_27
58	AL_SZK	Al_A	AL_1
59			AL_2
60			AL_3
61			AL_4
62		Al_B	AL_5
63			AL_6
64			AL_7
65			AL_8
66		Al_C	AL_9
67			AL_10
68			AL_11
69			AL_12
70		Al_D	AL_17
71			AL_18
72			AL_19
73			AL_20
74		Al_E	AL_21
75			AL_22
76			AL_23
77			AL_24
78		Al_F	AL_25
79			AL_26
80			AL_27
81			AL_28
82		Al_G	AL_33
83			AL_34
84			AL_35
85			AL_36
86	S_CATAL	Sp_A	Sp_1
87			Sp_2
88			Sp_3
89		Sp_B	Sp_4
90			Sp_5
91	S_MNRC	Sp_C	Sp_6
92			Sp_7
93			Sp_8
94		Sp_D	Sp_9
95			Sp_10
96			Sp_11
97		Sp_E	Sp_15
98			Sp_16
99			Sp_17

Supplementary Table S2.9 Estimation of the anthropogenic pressures on the fucalean assemblages based on the MA-LUSI Index. G_OCEM: Open coasts Eastern Macedonia, Greece; G_OCCM: Open coasts Central Macedonia, Greece; I_ANC: Conero Riviera – Ancona, Italy; I_GNI: Gulf of Naples, Italy; AL_SZK: Sazan Karaburuni MPA – Sazan, Albania; S_MNRC: Menorca, Spain; S_CATAL: Catalonia coasts, Spain; T_DIDIM: Didim, TurkeyA

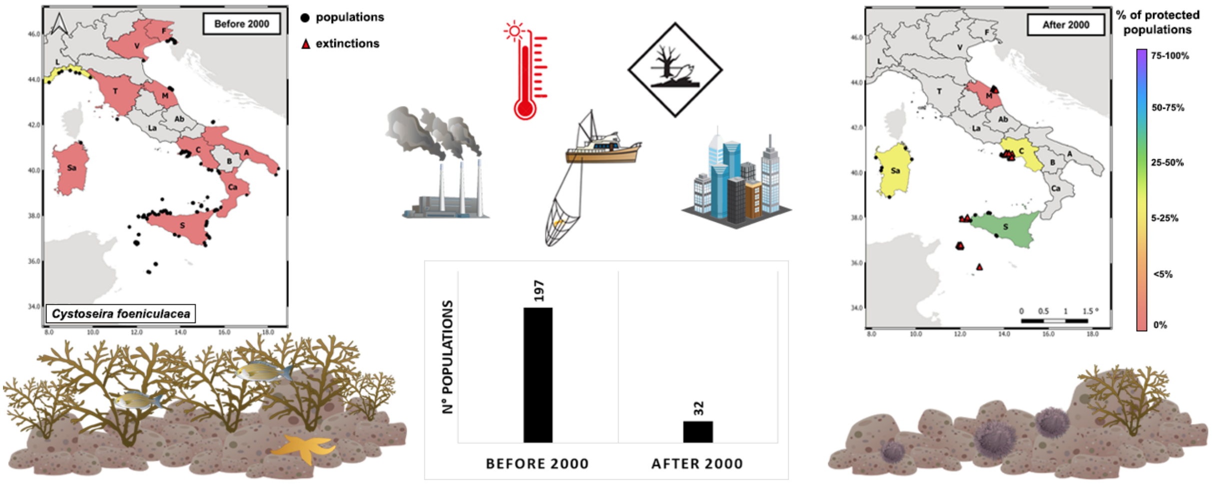
		Indirect pressures			**Direct Pressures category A**		**Direct Pressures category B**			**SUM**				**MA-LUSI**	
Location	**Meadow**	Urban (codes 11)	Commersial & Industrial (codes 12, 13)	Agriculture (codes 21-24)	**Mariculture**	**Sediment nutrient release**	**Sewage outfall**	**Irregular Fresh Water inputs**	**Harbor**		Background trophic status	**Stability of water column**	**Confinement**		**Grazing**
G_OCEM	Gr_A	0	0	2	0	0	0	0	0	2	1	1	1	2	0
G_OCEM	Gr_B	0	0	1	0	0	0	0	0	1	1	1	1	1	1
G_OCEM	Gr_C	0	0	1	0	0	0	0	0	1	1	1	1	1	0
G_OCEM	Gr_D	0	0	1	0	0	0	1	0	2	1	1	1	2	0
G_OCCM	Gr_E	0	0	1	0	0	0	0	0	1	1	1	1.25	1	2
G_OCCM	Gr_F	0	0	1	0	0	0	0	1	2	1	1	1.25	2,5	2
G_OCCM	Gr_G	0	0	1	0	0	0	0	0	1	1	1	1	1	2
G_OCCM	Gr_H	0	0	1	2	0	0	0	0	3	1	1	1.25	3,75	1
G_OCCM	Gr_I	0	0	1	0	0	0	0	0	1	1	1	1	1	2
G_OCCM	Gr_J	0	0	1	0	0	0	0	0	1	1	1	1	1	1
AL_SZK	Al_A	0	0	0	0	0	0	0	0	0	1	1	1	0	3
AL_SZK	Al_B	0	0	0	0	0	0	0	0	0	1	1	1	0	2
AL_SZK	Al_C	0	0	0	0	0	0	0	0	0	1	1	1	0	1
AL_SZK	Al_D	0	0	0	0	0	0	0	0	0	1	1	1	0	1
AL_SZK	Al_E	0	0	0	0	0	0	0	0	0	1	1	1	0	1
AL_SZK	Al_F	0	0	0	0	0	0	0	0	0	1	1	1	0	2
AL_SZK	Al_G	0	0	0	0	0	0	0	0	0	1	1	1	0	2
I_ANC	I_A	0	0	0	2	0	0	0	0	2	1	1	0.75	1,5	0
I_ANC	I_B	0	0	1	2	0	0	0	0	3	1	1	0.75	2,25	0
I_ANC	I_C	1	0	0	2	0	0	0	3	6	1	1	0.75	4,5	0
I_ANC	I_D	1	0	1	2	0	0	0	3	7	1	1	0.75	5,25	0
I_ANC	I_E	1	0	1	2	0	0	0	3	7	1	1	0.75	5,25	0
I_GNI	Is_A	0	0	1	0	0	1	0	0	2	1	1	0.75	1,50	0
S_CATAL	Sp_A	0	0	0	0	0	1	0	1	2	1	1	0.75	1.5	0
S_CATAL	Sp_B	0	0	0	0	0	0	0	2	2	1	1	0.75	1.5	0
S_MNRC	Sp_C	0	0	1	1	0	0	0	1	3	1	1	1.25	3.75	0
S_MNRC	Sp_D	0	0	1	0	0	0	0	1	2	1	1	1.25	2.5	0
S_MNRC	Sp_E	0	0	1	0	0	0	0	0	1	1	1	1.25	1.25	0
T_DIDIM	Tr_A	0	0	2	0	0	0	0	0	2	1	1	1	2	0

Supplementary Table S2.10 Spearman rank order correlation of abiotic and biotic factors. The correlation coefficients (ρ) are shown in the matrix. Numbers in bold $p < 0.05$.

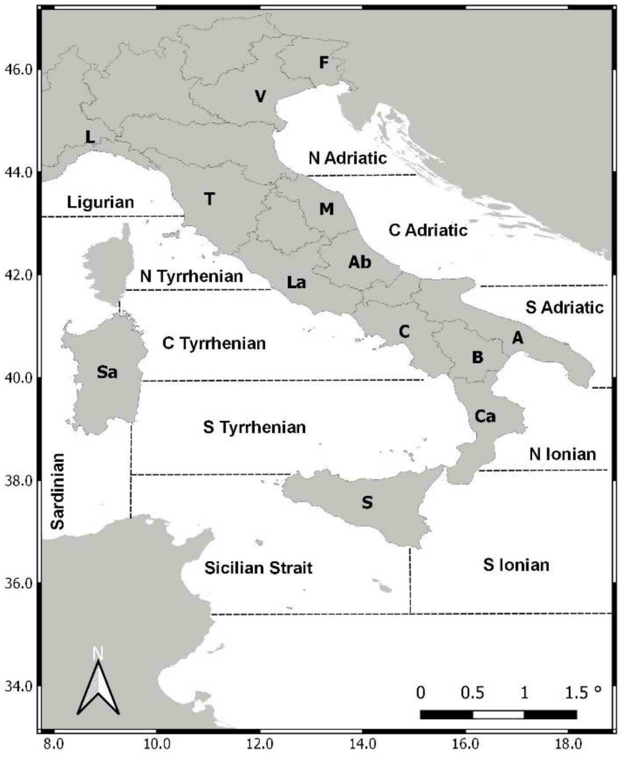
	Urbanization	**Agriculture**	**Mariculture**	**Sewage outfall**	**Harbour**	**MA-LUSI**
Urbanization						
Agriculture	0.000					
Mariculture	**0.627**	0.039				
Sewage outfall	-0.092	-0.083	-0.153			
Harbour	**0.672**	-0.056	**0.423**	0.093		
MA-LUSI	**0.534**	**0.513**	**0.684**	0.049	**0.645**	
Grazing	-0.310	-0.295	**-0.399**	-0.249	**-0.410**	**-0.656**

CHAPTER 3

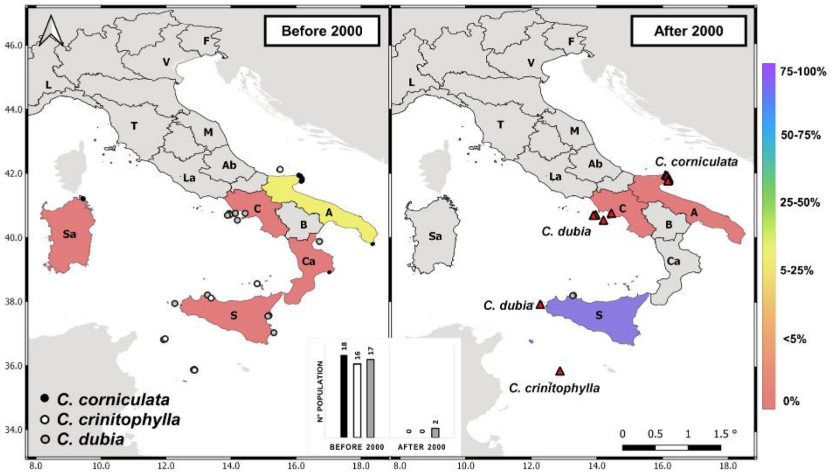
Graphical abstract



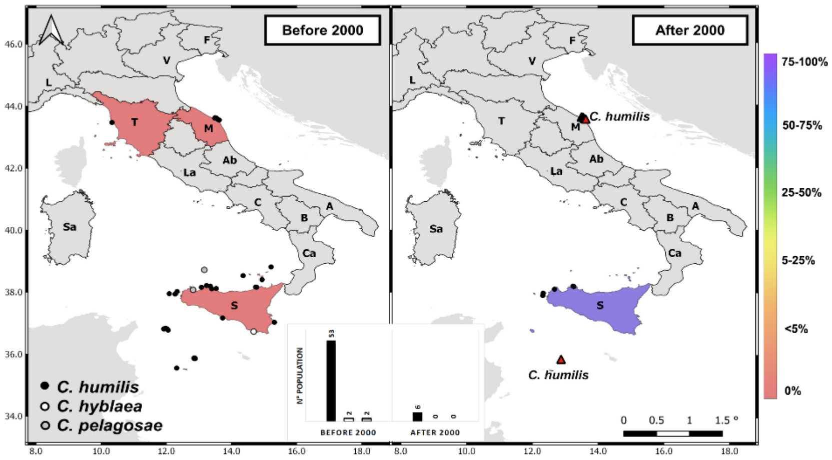
Supplementary Figure S3.1 Map of the subdivision of Italian seas used to classify the spatial distribution of *Cystoseira s.l.* species. The Tyrrhenian and Adriatic Sea are split in Northern, Central and Southern sectors (N, C, S); the Ionian Sea is divided in Northern and Southern sectors (N, S). A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.



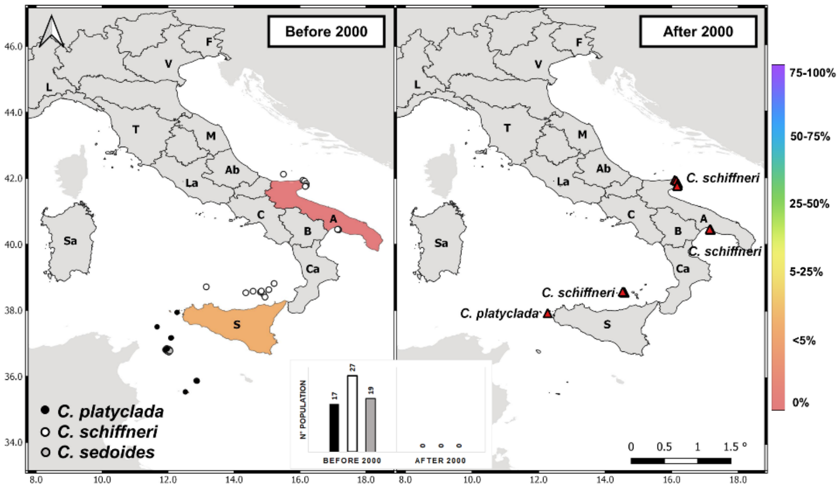
Supplementary Figure S3.2 Historical and recent distribution of *Cystoseira corniculata*, *C. crinitophylla* and *C. dubia* along Italian coasts. Black dots indicate the presence of *C. corniculata*, white dots show populations of *C. crinitophylla*, grey dots represent populations of *C. dubia*, red triangles indicate documented local extinctions. The color of each region represents the percentage of populations protected respectively for *C. crinitophylla* before 2000 and *C. dubia* after 2000. No population of *C. corniculata* was protected. Grey regions = no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.



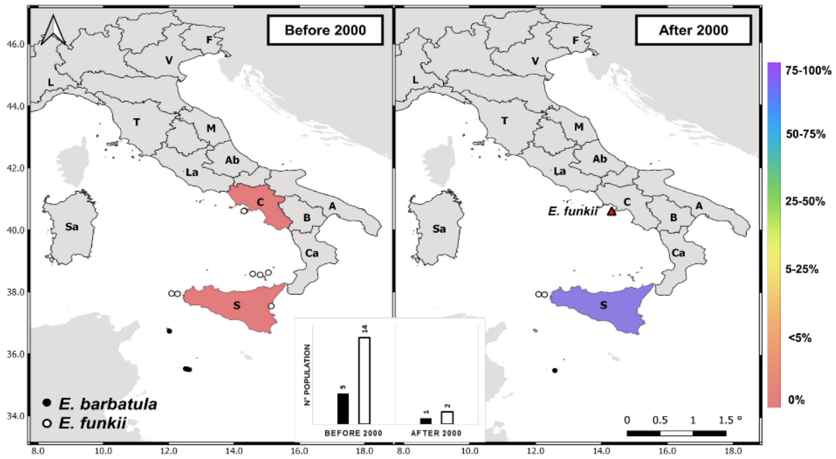
Supplementary Figure S3.3 Historical and recent distribution of *Cystoseira humilis, C. hyblaea* and *C. pelagosae* along Italian coasts. Black dots and lines indicate the presence of *C. humilis*, white dots show populations of *C. hyblaea*, grey dots represent populations of *C. pelagosae,* red triangles indicate documented local extinctions. The color of each region represents the percentage of populations of *C. humilis* protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region. No populations of *C. hyblaea* or *C. pelagosae* were protected. Grey regions = no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.



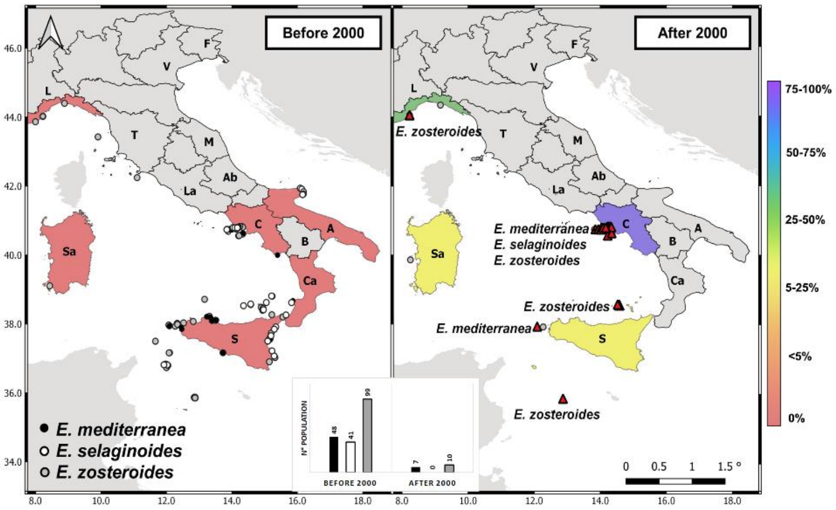
Supplementary Figure S3. 4 Historical and recent distribution of *Cystoseira platyclada*, *C. schiffneri* and *C. sedoides* along Italian coasts. Black dots indicate the presence of C. platyclada, white dots show populations of C. schiffneri, grey dots represent populations of *C. sedoides*, red triangles indicate documented local extinctions. The color of each region represents the percentage of populations of *C. schiffneri* protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region. No populations of *C. platyclada* or *C. sedoides* were protected. Grey regions = no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.



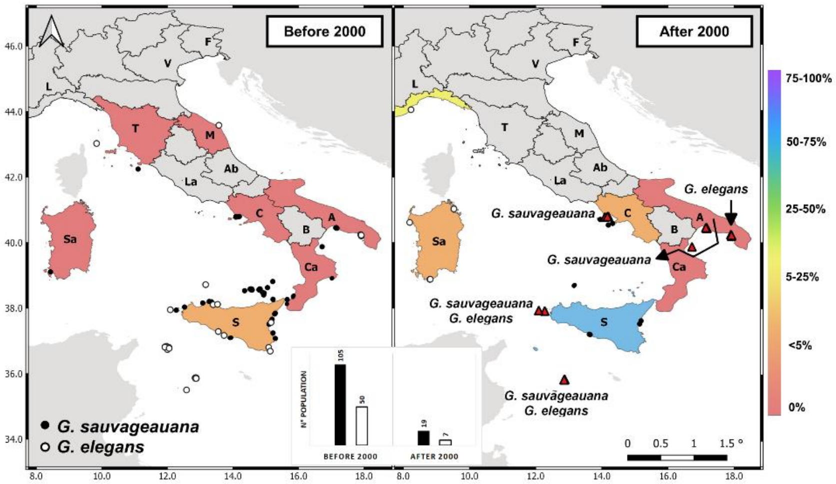
Supplementary Figure S3.5 Historical and recent distribution of *Ericaria barbatula* and *E. funkii* along Italian coasts. Black dots indicate the presence of *E. barbatula*, white dots show populations *of E. funkii*, red triangles represent documented local extinctions. The color of each region represents the percentage of populations of *E. funkii* protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region. No population of *E. barbatula* was protected. Grey regions = no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.



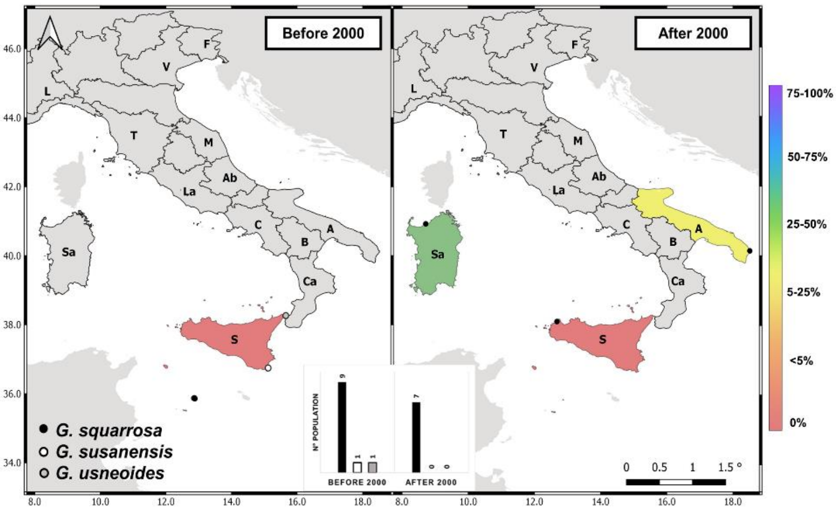
Supplementary Figure S3.6 Historical and recent distribution of *Ericaria mediterranea*, *E. selaginoides and E. zosteroides* along Italian coasts. Black dots and lines indicate the presence of *E. mediterranea*, white dots show populations of *E. selaginoides*, grey dots represent populations of *E. zosteroides*, red triangles indicate documented local extinctions. The color of each region represents the percentage of populations of *E. zosteroides* or *E. mediterranea* protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region. No population of *E. selaginoides* was protected. Grey regions = no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.



Supplementary Figure S3.7 Historical and recent distribution of *Gongolaria sauvageauana* and *G. elegans* along Italian coasts. Black dots and lines indicate the presence of *G. sauvageauana*, white dots show populations of *G. elegans*, red triangles indicate documented local extinctions. The color of each region represents the percentage of populations of the two species protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region. Grey regions = no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.



Supplementary Figure S3.8 Historical and recent distribution of *Gongolaria squarrosa*, *G. susanensis* and *G. usneoides* along Italian coasts. Black dots indicate the presence of *G. squarrosa*, white dots show populations of *G. susanensis*, grey dots represent populations of *G. usneoides*. The color of each region represents the percentage of populations of *G. squarrosa* protected by MPAs, national and underwater parks, Natura 2000 sites or SPAMIs within each region. No populations of *G. susanensis* or *G. usneoides* were protected. Grey regions = no population censused. A = Apulia, Ab = Abruzzo, B = Basilicata, C = Campania, Cl = Calabria, F = Friuli-Venezia Giulia, L = Liguria, La = Lazio, M = Marche, S = Sicily, Sa = Sardinia, T = Tuscany, V = Veneto.



Dataset references: integrated in the Supplementary Table S4.1

CHAPTER 4

Supplementary Table S4.1 Data sources: peer-reviewed and grey literature for the georeferenced occurrence dataset of fucalean forests across the Mediterranean Sea.

ID	Authors	Title	Journal/Report/Thesis	Publication year	Country	Point	Line	Absence info	Cover info
1	Ballesteros, E	Structure and dynamics sf the community of Cystoseira zosteroides (Turner)C. Agardh (Fucales, Phaeophyceae) in the northwestern Mediterranean.	SCIENTIA MARINA	1990	France	y	n	n	n
2	Benedetti-Cecchi, L; Cinelli, F	EFFECTS OF CANOPY COVER, HERBIVORES AND SUBSTRATUM TYPE ON PATTERNS OF CYSTOSEIRA SPP SETTLEMENT AND RECRUITMENT IN LITTORAL ROCKPOOLS	MARINE ECOLOGY PROGRESS SERIES	1992	Italy	y	n	n	n
3	Benedetti-Cecchi, L; Cinelli, F	Seasonality and reproductive phenology of algae inhabiting littoral pools in the Western Mediterranean	MARINE ECOLOGY-PUBBLICAZIONI DELLA STAZIONE ZOOLOGICA DI NAPOLI I	1993	Italy	y	n	n	y
4	Benedetti-Cecchi, L; Cinelli, F	Habitat eterogeneity, sea-urchin grazing and the distribution of algae in littoral rock pools on the West-coast of Italy (Western Mediterranean)	MARINE ECOLOGY PROGRESS SERIES	1995	Italy	y	n	n	y
5	Benedetti-Cecchi, L; Nuti, S; Cinelli, F	Analysis of spatial and temporal variability in interactions among algae, limpets and mussels in low-shore habitats on the west coast of Italy	MARINE ECOLOGY PROGRESS SERIES	1996	Italy	y	n	n	y
6	RodriguezPrieto, C; Polo, L	Effects of sewage pollution in the structure and dynamics of the community of Cystoseira mediterranea (Fucales, Phaeophyceae)	SCIENTIA MARINA	1996	Spain	y	n	n	y
7	Ballesteros, E; Sala, E; Garrabou, J; Zabala, M	Community structure and frond size distribution of a deep water stand of Cystoseira spinosa (Phaeophyta) in the northwestern Mediterranean	EUROPEAN JOURNAL OF PHYCOLOGY	1998	France	y	n	n	y
8	Chryssovergis, F;Panayotidis, P	ETUDE DU PHYTOBENTHOS DES COTES SUD-EST D'ATTIQUE, EN VUE DE L'INSTALLATION DU RÉSEAU D'ASSAINISSEMENT DES EAUX USÉES, DE LA BANLIEUE EST D ATHÈNES (RÉSULTATS PRÉLIMINAIRES)	RAPPORT COMMISSION INTERNATIONAL MER MEDITERRANEE	1998	Greece	y	n	n	n
9	Span, A; Antolic, B; Zuljevic, A	THE GENUS CAULERPA (CAULERPALES, CHLOROPHYTA) IN ADR1ATIC SEA	RAPPORT COMMISSION INTERNATIONAL MER MEDITERRANEE	1998	Croatia	n	n	n	n

10	Panayotidis, P; Feretopoulou, J; Montesanto, B	Benthic vegetation as an ecological quality descriptor in an Eastern Mediterranean coastal area (Kalloni Bay, Aegean Sea, Greece)	ESTUARINE COASTAL AND SHELF SCIENCE	1999	Greece	y	n	n	y
11	Belegratis, MR; Bitis, I; Economou-Amilli, A; Ott, JA	Epiphytic patterns of macroalgal assemblages on Cystoseira species (Fucales, Phaeophyta) in the east coast of Attica (Aegean Sea, Greece)	HYDROBIOLOGIA	1999	Greece	y	n	n	n
12	Alongi, G; Catra, M; Cormaci, M	Cystoseira susanensis (Cystoseiraceae, Phaeophyta): a little known and rare Mediterranean species.	CRYPTOGAMIE ALGOLOGIE	1999	Italy	n	y	n	n
13	Pardi, G; Piazzi, L; Cinelli, F	Demographic study of a Cystoseira humilis Kutzing (Fucales : Cystoseiraceae) population in the western Mediterranean	BOTANICA MARINA	2000	Italy	y	n	n	n
14	Milazzo, M; Chemello, R; Badalamenti, F; Riggio, S	Molluscan assemblages associated with photophilic algae in the Marine Reserve of Ustica Island (Lower Tyrrhenian Sea, Italy)	ITALIAN JOURNAL OF ZOOLOGY	2000	Italy	y	n	y	y
15	Tena, J; Capaccioni-Azzati, R; Torres-Gavila, FJ; Garcia-Carrascosa, AM	Polychaetes associated with different facies of the photophilic algal community in the Chafarinas Archipelago (SW Mediterranean)	BULLETIN OF MARINE SCIENCE	2000	Spain	y	n	n	n
16	Soltan, D; Verlaque, M; Boudouresque, CF; Francour, P	Changes in macroalgal communities in the vicinity of a Mediterranean sewage outfall after the setting up of a treatment plant	MARINE POLLUTION BULLETIN	2001	France	y	n	y	y
17	Montesanto, B; Panayotidis, P	The Cystoseira spp. communities from the Aegean Sea (north-east Mediterranean)	MEDITERRANEAN MARINE SCIENCE	2001	Greece	y	n	n	y
18	Benedetti-Cecchi, L; Pannacciulli, F; Bulleri, F; Moschella, PS; Airoldi, L; Relini, G; Cinelli, F	Predicting the consequences of anthropogenic disturbance: large-scale effects of loss of canopy algae on rocky shores	MARINE ECOLOGY PROGRESS SERIES	2001	Italy	y	y	y	y
19	Castello, J; Carballo, JL	Isopod fauna, excluding Epicaridea, from the Strait of Gibraltar and nearby areas (Southern Iberian Peninsula)	SCIENTIA MARINA	2001	Spain	y	n	n	n
20	Chemello, R; Milazzo, M	Effect of algal architecture on associated fauna: some evidence from phytal molluscs	MARINE BIOLOGY	2002	Italy	y	n	n	n
21	Turna, I,I; Ertan, Ö, O; Cormaci,M; Furnari, G	Seasonal Variations in the Biomass of Macro-Algal Communities from the Gulf of Antalya (north-eastern Mediterranean)	Turk J Bot	2002	Turkey	y	n	n	n
22	Milazzo, M; Chemello, R; Badalamenti, F; Riggio, S	Short-term effect of human trampling on the upper infralittoral macroalgae of Ustica Island MPA (western Mediterranean, Italy)	JOURNAL OF THE MARINE BIOLOGICAL ASSOCIATION OF THE UNITED KINGDOM	2002	Italy	y	n	n	n
23	Al-Masri, MS; Mamish, S; Budier, Y	Radionuclides and trace metals in eastern Mediterranean Sea algae	JOURNAL OF ENVIRONMENTAL RADIOACTIVITY	2003	Lebanon	y	n	n	n
24	Panayotidis, P; Montesanto, B; Orfanidis, S	Use of low-budget monitoring of macroalgae to implement the European Water Framework Directive	JOURNAL OF APPLIED PHYCOLOGY	2004	Greece	y	n	n	y
25	Cormaci, M; Furnari, G; Giaccone, G; Serio, D	Two rare mediterranean species recorded from Sardinia: Laurencia pyramidalis (Rhodophyta, Rhodomelaceae) e Cystoseira squarrosa (Ochrophyta, Cystoseiraceae)	BIOLOGIA MARINA MEDITERRANEA	2005	Croatia, Italy	n	n	n	n

26	Thibaut, T; Pinedo, S; Torras, X; Ballesteros, E	Long-term decline of the populations of Fucales (Cystoseira spp. and Sargassum spp.) in the Alberes coast (France, North-western Mediterranean)	MARINE POLLUTION BULLETIN	2005	France	n	y	n	y
27	Tsirika, A; Haritonidis, S	A survey of the benthic flora in the National Marine Park of Zakynthos (Greece)	BOTANICA MARINA	2005	Greece	y	n	n	n
28	Fraschetti, S; Terlizzi, A; Benedetti-Cecchi, L	Patterns of distribution of marine assemblages from rocky shores: evidence of relevant scales of variation	MARINE ECOLOGY PROGRESS SERIES	2005	Italy	n	y	n	n
29	Parlakay, A; Sukatar, A; Şenkardeşler, A	Marine Flora Between South Çeşme and Cape Teke (Izmir, Aegean Sea, Turkey)	E.Ü. SU ÜRÜNLERI DERGISI 2005 - E.U. JOURNAL OF FISHERIES & AQUATIC SCIENCES	2005	Turkey	y	n	y	n
30	Fraschetti, S; Terlizzi, A; Bevilacqua, S; Boero, F	The distribution of hydroids (Cnidaria, Hydrozoa) from micro- to macro-scale: Spatial patterns on habitat-forming algae	JOURNAL OF EXPERIMENTAL MARINE BIOLOGY AND ECOLOGY	2006	Italy	y	n	n	n
31	Falace, A; Zanelli, E; Bressan, G	Algal transplantation as a potential tool for artificial reef management and environmental mitigation	BULLETIN OF MARINE SCIENCE	2006	Italy	y	n	y	n
32	Falace, A; Bressan, G	Seasonal variations of Cystoseira barbata (Stackhouse) C. Agardh frond architecture	HYDROBIOLOGIA	2006	Slovenia	y	n	n	n
33	V. Aysel, H. Erdugan, E. Ş. Okudan.	Marine Algae and Seagrasses of Adana (Mediterranean, Turkey)	J. BLACK SEA/MEDITERRANEAN ENVIRONMENT	2006	Turkey	n	n	n	n
34	V. Aysel, E. Ş. Okudan, Erduğan, H.	Marine Algae and Seagrasses of Mersin Shore (Mediterranean, Turkey).	J. BLACK SEA/MEDITERRANEAN ENVIRONMENT	2006	Turkey	n	n	n	n
35	Serio, L; Alongi, G; Catra, M; Cormaci, M; Furnari, G	Changes in the benthic algal flora of Linosa Island (Straits of Sicily, Mediterranean Sea)	BOTANICA MARINA	2006	Italy	y	n	n	n
36	Susini, ML; Thibaut, T; Meynesz, A; Forcioli, D	A preliminary study of genetic diversity in Cystoseira amentacea (C. Agardh) Bory var. stricta Montagne (Fucales, Phaeophyceae) using random amplified polymorphic DNA	PHYCOLOGIA	2007	France	y	n	n	n
37	Panayotidis, P; Orfanidis, S; Tsiamis, K	CYSTOSEIRA CRINITA COMMUNITY IN THE AEGEAN SEA	RAPPORT COMMISSION INTERNATIONAL MER MEDITERRANEE	2007	Greece	y	n	n	n
38	Arevalo, R; Pinedo, S; Ballesteros, E	Changes in the composition and structure of Mediterranean rocky-shore communities following a gradient of nutrient enrichment: Descriptive study and test of proposed methods to assess water quality regarding macroalgae	MARINE POLLUTION BULLETIN	2007	Spain	y	n	y	y
39	Pinedo, S; Garcia, M; Satta, MP; de Torres, M; Ballesteros, E	Rocky-shore communities as indicators of water quality: A case study in the Northwestern Mediterranean	MARINE POLLUTION BULLETIN	2007	Spain	y	n	n	n
40	Datça-Bozburun Özel Çevre Koruma Bölgesi'nin Biolojik Çeşitliliği (Güneydoğu Ege Denizi, Türkiye) Erdoğan Okuş, Ahsen Yüksek, İ. Noyan Yılmaz, Aslı Aslan	Marine biodiversity of Datça-Bozburun specially protected area (Southeastern Aegean Sea, Turkey) Datça-Bozburun özelçevre koruma bölgesi'nin biyolojik çeşitliliği (Güneydoğu Ege Denizi, Türkiye)	J. BLACK SEA/MEDITERRANEAN ENVIRONMENT	2007	Turkey	n	n	n	n

	Yılmaz, S. Ünsal Karhan, M. İdil Öz, Nazlı Demirel, Seyfettin Taş, Volkan Demir, Sibel Zeki, Ertuğrul M. Koç, Umut Tural, Baki Yokeş, Evrim Kalkan, Niyazi Deniz, Ziya Çaylarbaşı, Eren Savut and Elif Murat								
42	Hereu, B; Mangialajo, L; Ballesteros, E; Thibaut, T	On the occurrence, structure and distribution of deep-water Cystoseira (Phaeophyceae) populations in the Port-Cros National Park (north-western Mediterranean)	EUROPEAN JOURNAL OF PHYCOLOGY	2008	France	y	n	y	y
43	Mangialajo, L; Chiantore, M; Cattaneo-Vietti, R	Loss of fucoid algae along a gradient of urbanisation, and structure of benthic assemblages	MARINE ECOLOGY PROGRESS SERIES	2008	Italy	y	n	n	n
44	Caparkaya, D; Cavas, L	Biosorption of Methylene Blue by a Brown Alga Cystoseira barbatula Kutzing	ACTA CHIMICA SLOVENICA	2008	Slovenia	y	n	n	n
45	Ballesteros, E; Garrabou, J; Hereu, B; Zabala, M; Cebrian, E; Sala, E	Deep-water stands of Cystoseira zosteroides C. Agardh (Fucales, Ochrophyta) in the Northwestern Mediterranean: Insights into assemblage structure and population dynamics	ESTUARINE COASTAL AND SHELF SCIENCE	2009	France	y	n	n	n
46	Maggi, E; Bertocci, I; Vaselli, S; Benedetti-Cecchi, L	Effects of changes in number, identity and abundance of habitat-forming species on assemblages of rocky seashores	MARINE ECOLOGY PROGRESS SERIES	2009	Italy	y	n	n	y
47	Riccato, F; Fiorin, R; Curiel, D; Rismondo, A; Cerasuolo, C; Torricelli, P	INTERAZIONE TRA IL POPOLAMENTO ITTICO E LE ALGHE BRUNE DEL GENERE CYSTOSEIRA IN UN AMBIENTE DI SCOGLIERA ARTIFICIALE DEL GOLFO DI VENEZIA	BOLLETTINO MUSEO CIVICO DI STORIA NATURALE DI VENEZIA	2009	Italy	y	n	n	n
48	Sales, M; Ballesteros, E	Shallow Cystoseira (Fucales: Ochrophyta) assemblages thriving in sheltered areas from Menorca (NW Mediterranean): Relationships with environmental factors and anthropogenic pressures	ESTUARINE COASTAL AND SHELF SCIENCE	2009	Spain	y	n	y	y
49	Verges, A; Alcoverro, T; BallesterOS, E	Role of fish herbivory in structuring the vertical distribution of canopy algae Cystoseira spp. in the Mediterranean Sea	MARINE ECOLOGY PROGRESS SERIES	2009	Spain	y	n	n	y
50	Macic, V; Antolic, B; Thibaut, T; Svircev, Z	DISTIBUTION OF THE MOST COMMON Cystoseira C. AGARDH SPECIES (HETEROKONTOPHYTA, FUCALES) ON THE COAST OF MONTENEGRO (SOUTH-EAST ADRIATIC SEA)	FRESENIUS ENVIRONMENTAL BULLETIN	2010	Montenegro	y	n	n	n
51	Sales, M; Ballesteros, E	Long-term comparison of algal assemblages dominated by Cystoseira crinita (Fucales, Heterokontophyta) from Cap Corse (Corsica, North Western Mediterranean)	EUROPEAN JOURNAL OF PHYCOLOGY	2010	France	y	n	n	y
52	Conti, ME; Bocca, B; Iacobucci, M; Finoia, MG; Mecozzi, M; Pino, A; Alimonti, A	Baseline Trace Metals in Seagrass, Algae, and Mollusks in a Southern Tyrrhenian Ecosystem (Linosa Island, Sicily)	ARCHIVES OF ENVIRONMENTAL CONTAMINATION AND TOXICOLOGY	2010	Italy	y	n	n	n

53	Macic, V	PERIODISM OF CYSTOSEIRA SPINOSA SAUV. (FUCALES) FROND SIZE ON THE MONTENEGRIN COAST	RAPPORT COMMISSION INTERNATIONAL MER MEDITERRANEE	2010	Montenegro	y	n	n	n
54	Zerzeri, A; Djellouli, A,S; Mezgui, Y; Ben Hassine, O, K	CONTRIBUTION A LA CARACTERISATION DE LA MACROFLORE BENTHIQUE DES REGIONS DE BIZERTE, CAP-ZEBIB ET RAF - RAF (TUNISIE, MEDITERRANEE)	RAPPORT COMMISSION INTERNATIONAL MER MEDITERRANEE	2010	Tunisia	y	n	n	n
55	Özvarol,Y; Ertan, Ö.O; Turna, İ.İ	DETERMINATION MACROBENTIC FLORA OF NORTHEASTERN MEDITERRANEAN (GAZIPASA-ISKENDERUN)	E-JOURNAL OF NEW WORLD SCIENCES ACADEMY	2010	Turkey	y	n	y	y
56	Perkol-Finkel, S; Airoldi, L	Loss and Recovery Potential of Marine Habitats: An Experimental Study of Factors Maintaining Resilience in Subtidal Algal Forests at the Adriatic Sea	PLoS ONE	2010	Italy	y	n	n	y
57	Fraschetti, S; Terlizzi, A; Guarnieri, G; Pizzolante, F; D'Ambrosio, P; Maiorano, P; Beqiraj, S; Boero, F	Effects of Unplanned Development on Marine Biodiversity: A Lesson from Albania (Central Mediterranean Sea)	JOURNAL OF COASTAL RESEARCH	2011	Albania	y	n	n	n
58	Sala E, Kizilkaya Z, Yildirim D, Ballesteros E	Alien Marine Fishes Deplete Algal Biomass in the EasternMediterranean	PLoS ONE	2011	Israel	y	n	y	n
59	Renzi, M; Romeo, T; Guerranti, C; Perra, G; Italiano, F; Focardi, SE; Esposito, V; Andaloro, F	Temporal trends and matrix-dependent behaviors of trace elements closed to a geothermal hot-spot source (Aeolian Archipelago, Italy)	2ND INTERNATIONAL WORKSHOP ON RESEARCH IN SHALLOW MARINE AND FRESH WATER SYSTEMS	2011	Italy	y	n	n	n
60	Navarro, L; Ballesteros, E; Linares, C; Hereu, B	Spatial and temporal variability of deep-water algal assemblages in the Northwestern Mediterranean: The effects of an exceptional storm	ESTUARINE COASTAL AND SHELF SCIENCE	2011	Spain	y	n	n	n
61	Sales, M; Cebrian, E; Tomas, F; Ballesteros, E	Pollution impacts and recovery potential in three species of the genus Cystoseira (Fucales, Heterokontophyta)	ESTUARINE COASTAL AND SHELF SCIENCE	2011	Spain	y	n	n	n
63	Durucan,F; Turna, I,	Antalya Batı Kıyılarının (Antalya – Kalkan) Makrobentik Deniz Algleri.	JOURNAL OF SCIENCE (E-JOURNAL)	2011	Turkey	y	n	n	n
64	Spatharis, S; Orfanidis, S; Panayotidis, P; Tsirtis, G	Assembly processes in upper subtidal macroalgae: The effect of wave exposure	ESTUARINE COASTAL AND SHELF SCIENCE	2011	Greece	y	n	n	y
65	Porzio, L; Buia, MC; Hall-Spencer, JM	Effects of acidification on macroalgal communities	JOURNAL OF EXPERIMENTAL MARINE BIOLOGY AND ECOLOGY	2011	Italy	y	n	n	n
66	Sfriso, A; Facca, C	Macrophytes in the anthropic construction of the Venice littorals and their ecological assessment by integration of the "CARLIT" index	ECOLOGICAL INDICATORS	2011	Italy	y	n	n	n
67	Rozic, S; Puizina, J; Samanic, I; Zuljevic, A; Antolic, B	Molecular identification of the brown algae, Cystoseira spp. (Phaeophycae, Fucales) from the Adriatic Sea - preliminary results	ACTA ADRIATICA	2012	Croatia	y	n	n	n
68	Robvieux, P; Videment, J; Ribout, C; Forcioli, D; Meinesz, A; Thibaut, T; Bottin, L	First characterization of eight polymorphic microsatellites for Cystoseira amentacea var. stricta (Fucales, Sargassaceae)	CONSERVATION GENETICS RESOURCES	2012	France	y	n	n	n

69	Giakoumi, S; Cebrian, E; Kokkoris, GD; Ballesteros, E; Sala, E	Relationships between fish, sea urchins and macroalgae: The structure of shallow rocky sublittoral communities in the Cyclades, Eastern Mediterranean	ESTUARINE COASTAL AND SHELF SCIENCE	2012	Greece	y	n	n	n
70	Perkol-Finkel, S; Ferrario, F; Nicotera, V; Airoldi, L	Conservation challenges in urban seascapes: promoting the growth of threatened species on coastal infrastructures	JOURNAL OF APPLIED ECOLOGY	2012	Italy	y	n	n	n
71	Fraschetti, S; Bevilacqua, S; Guarnieri, G;Terlizzi, A	Idiosyncratic effects of protection in a remote marine reserve	MARINE ECOLOGY PROGRESS SERIES	2012	Italy	y	n	n	y
72	Sales, M; Ballesteros, E	Seasonal dynamics and annual production of Cystoseira crinita (Fucales: Ochrophyta)-dominated assemblages from the northwestern Mediterranean	SCIENTIA MARINA	2012	Spain	y	n	y	y
73	Sales, M; Ballesteros, E; Anderson, MJ; Ivesa, L; Cardona, E	Biogeographical patterns of algal communities in the Mediterranean Sea: Cystoseira crinita-dominated assemblages as a case study	JOURNAL OF BIOGEOGRAPHY	2012	Spain,Italy,Greece,Turkey	y	n	n	n
74	Nikolic, V; Zuljevic, A; Mangialajo, L; Antolic, B; Kuspilic, G; Ballesteros, E	Cartography of littoral rocky-shore communities (CARLIT) as a tool for ecological quality assessment of coastal waters in the Eastern Adriatic Sea	ECOLOGICAL INDICATORS	2013	Croatia	n	n	n	n
75	Cheminee, A; Sala, E; Pastor, J; Bodilis, P; Thiriet, P; Mangialajo, L; Cottalorda, JM; Francour, P	Nursery value of Cystoseira forests for Mediterranean rocky reef fishes	JOURNAL OF EXPERIMENTAL MARINE BIOLOGY AND ECOLOGY	2013	France	y	n	n	y
76	Tsiamis, K; Panayotidis, P; Salomidi, M; Pavlidou, A; Kleinteich, J; Balanika, K; Kupper, FC	Macroalgal community response to re-oligotrophication in Saronikos Gulf	MARINE ECOLOGY PROGRESS SERIES	2013	Greece	y	n	n	n
77	Tamburello, L; Bulleri, F; Bertocci, I; Maggi, E; Benedetti-Cecchi, L	Reddened seascapes: experimentally induced shifts in 1/f spectra of spatial variability in rocky intertidal assemblages	ECOLOGY	2013	Italy	n	n	n	n
78	Alagna, A; Fernandez, TV; Terlizzi, A; Badalamenti, F	Influence of microhabitat on seedling survival and growth of the Mediterranean seagrass Posidonia oceanica (L.) Delile	ESTUARINE COASTAL AND SHELF SCIENCE	2013	Italy	y	n	n	n
79	Bermejo, R; de la Fuente, G; Vergara, JJ; Hernandez, I	Application of the CARLIT index along a biogeographical gradient in the Alboran Sea (European Coast)	MARINE POLLUTION BULLETIN	2013	Spain	y	n	n	n
81	Taşkın, E; Öztürk, M; Kurt, Ö; Ulcay, S	Benthic marine algae in Northern Cyprus (Eastern Mediterranean Sea)	J. BLACK SEA/MEDITERRANEAN ENVIRONMENT	2013	Cyprus	y	n	y	n
82	Tsiamis, K; Economou-Amilli, A; Katsaros, C; Panayotidis, P	First account of native and alien macroalgal biodiversity at Andros Island (Greece, Eastern Mediterranean)	NOVA HEDWIGIA	2013	Greece	y	n	n	y
83	Ivеša,L; Devescovi, M.	DISTRIBUTION AND COMPOSITION OF CYSTOSEIRA STANDS ALONG THE WEST ISTRIAN COAST (NORTHERN ADRIATIC, CROATIA) AND COMPARISON WITH HISTORICAL DATA	PROCEEDINGS OF THE 5th MEDITERRANEAN SYMPOSIUM ON MARINE VEGETATION Portorož, Slovenia, 27-28 October 2014	2014	Croatia	n	n	n	n

84	Giakoumi, S	Distribution patterns of the invasive herbivore Siganus luridus (Ruppell, 1829) and its relation to native benthic communities in the central Aegean Sea, Northeastern Mediterranean	MARINE ECOLOGY-AN EVOLUTIONARY PERSPECTIVE	2014	Greece	y	n	n	y
85	Bedini, R; Bonechi, L; Piazzi, L	Mobile epifaunal assemblages associated with Cystoseira beds: comparison between areas invaded and not invaded by Lophocladia lallemandii	SCIENTIA MARINA	2014	Italy	y	n	n	n
86	Macic, V; Svircev, Z	MACROEPIPHYTES ON CYSTOSEIRA SPECIES (PHAEOPHYCEAE) ON THE COAST OF MONTENEGRO	FRESENIUS ENVIRONMENTAL BULLETIN	2014	Montenegro	y	n	n	n
87	Bouafif, C; Verlaque, M; Langar, H.	Cystoseira taxa new for the marine flora of Tunisia	CRYPTOGAMIE,ALGOLOGIE	2014	Tunisia	y	n	n	n
88	Bouafif,C; Ouerghi, A; Langar, H.	CYSTOSEIRA SEDOIDES (DESFONTAINES) C. AGARDH DES COTES TUNISIENNES : ETAT ACTUEL DES CONNAISSANCES	PROCEEDINGS OF THE 5th MEDITERRANEAN SYMPOSIUM ON MARINE VEGETATION Portorož, Slovenia, 27-28 October 2014	2014	Tunisia	y	n	n	n
89	Tsiamis, K; Taskin, E; Orfanidis, S; Stavrou, P; Argyrou, M; Panayotidis, P; Tsioli, T; Cicek, B,A; Marcou, M; Kupper, F.C	Checklist of seaweeds of Cyprus (Mediterranean Sea)	BOTANICA MARINA	2014	Turkey	y	n	n	n
90	Thibaut, T.; Blanfune, A.; Boudouresque, C. F.; Verlaque, M.	Decline and local extinction of Fucales in the French Riviera: the harbinger of future extinctions?	MEDITERRANEAN MARINE SCIENCE	2015	France	y	y	y	y
91	Agnetta, D.; Badalamenti, F.; Ceccherelli, G.; Di Trapani, F.; Bonaviri, C.; Gianguzza, P.	Role of two co-occurring Mediterranean sea urchins in the formation of barren from Cystoseira canopy	ESTUARINE COASTAL AND SHELF SCIENCE	2015	Italy	y	n	n	n
92	Conti, M.E; Mecozzi, M; Finoia, M.G	Determination of trace metal baseline values in Posidonia oceanica, Cystoseira sp., and other marine environmental biomonitors: a quality control method for a study in South Tyrrhenian coastal areas	ENVIRONMENTAL SCIENCE AND POLLUTION RESEARCH	2015	Italy	y	n	n	n
93	Strain, E.M.A.; Van Belzen, J; Van Dalen, J; Bouma, T.J.; Airoldi, L	Management of Local Stressors Can Improve the Resilience of Marine Canopy Algae to Global Stressors	PLOS ONE	2015	Italy	y	n	n	n
94	Benedetti-Cecchi, L; Tamburello, L; Maggi, E; Bulleri, F	Experimental pertubations modify the performance of early warning indicators of rgime shift	CURRENT BIOLOGY	2015	Italy	y	n	n	n
95	Macic, V; Boris, A	Distribution of rare Cystoseira species along the Montenegro coast (South-Eastern Adriatic Sea)	PERIODICUM BIOLOGORUM	2015	Montenegro	y	n	n	n
97	Bermejo, R; Ramirez-Romero, E; Vergara, J.J.; Hernandez, I	Spatial patterns of macrophyte composition and landscape along the rocky shores of the Mediterranean-Atlantic transition region (northern Alboran Sea)	ESTUARINE COASTAL AND SHELF SCIENCE	2015	Spain	y	n	n	n

98	Capdevila, P.; Linares, C.; Aspillaga, E.; Navarro, L.; Kersting, D. K.; Hereu, B.	Recruitment patterns in the Mediterranean deep-water alga Cystoseira zosteroides	MARINE BIOLOGY	2015	Spain	y	n	n	n
99	Pinedo, S; Arevalo, Rl; Ballesteros, E	Seasonal dynamics of upper sublittoral assemblages on Mediterranean rocky shores along a eutrophication gradient	ESTUARINE COASTAL AND SHELF SCIENCE	2015	Spain	y	n	n	n
100	Ozgun, S; Turan, F	Biochemical composition of some brown algae from Iskenderun Bay, the northeastern Mediterranean coast of Turkey	JOURNAL OF THE BLACK SEA MEDITERRANEAN ENVIRONMENT	2015	Turkey	y	n	n	n
101	Akgul, R; Kizilkaya, B; Akgul, F; Erdugan, H	Total Lipid and Fatty Acid Composition of Twelve Algae from Canakkale (Turkey)	INDIAN JOURNAL OF GEO-MARINE SCIENCES	2015	Turkey	y	n	n	n
102	Blanfune, A; Boudouresque, C.F; Verlaque, M; Beqiraj, S; Kashta, L; Nasto, I; Ruci, S; Thibaut, T	Response of rocky shore communities to anthropogenic pressures in Albania (Mediterranean Sea): Ecological status assessment through the CARLIT method	MARINE POLLUTION BULLETIN	2016	Albania	y	y	y	y
103	Ivesa, L; Djakovac, T; Devescovi, M	Long-term fluctuations in Cystoseira populations along the west Istrian Coast (Croatia) related to eutrophication patterns in the northern Adriatic Sea	MARINE POLLUTION BULLETIN	2016	Croatia	y	n	n	y
104	Thibaut, T; Blanfune, A; Boudouresque, C.F; Cottalorda, J.M; Hereu, B; Susini, M.L; Verlaque, M	Unexpected temporal stability of Cystoseira and Sargassum forests in Port-Cros, one of the oldest Mediterranean marine National Parks	CRYPTOGAMIE ALGOLOGIE	2016	France	n	y	n	y
106	Blanfune, A.; Boudouresque, C. F.; Verlaque, M.; Thibaut, T.	The fate of Cystoseira crinita, a forest-forming Fucale (Phaeophyceae, Stramenopiles), in France (North Western Mediterranean Sea)	ESTUARINE COASTAL AND SHELF SCIENCE	2016	France	y	n	y	n
108	Mannino, A. M.; Vaglica, V.; Cammarata, M.; Oddo, E.	Effects of temperature on total phenolic compounds in Cystoseira amentacea (C. Agardh) Bory (Fucales, Phaeophyceae) from southern Mediterranean Sea	PLANT BIOSYSTEMS	2016	Italy	y	n	n	n
109	Ferrario, F; Ivesa, L; Jaklin, Aj; Perkol-Finkel, S; Airoldi, L	The overlooked role of biotic factors in controlling the ecological performance of artificial marine habitats	JOURNAL OF APPLIED ECOLOGY	2016	Italy	y	n	y	y
110	Guarnieri, G; Bevilacqua, S; De Leo, F; Farella, G; Maffia, A; Terlizzi, A; Fraschetti, S	The Challenge of Planning Conservation Strategies in Threatened Seascapes: Understanding the Role of Fine Scale Assessments of Community Response to Cumulative Human Pressures	PLOS ONE	2016	Italy	y	n	n	y
111	Gianguzza, P.; Di Trapani, F.; Bonaviri, C.; Agnetta, D.; Vizzini, S.; Badalamenti, F.	Size-dependent predation of the mesopredator Marthasterias glacialis (L.) (Asteroidea)	MARINE BIOLOGY	2016	Italy	y	n	n	n
112	Mancuso, F.P.; D'Hondt, S; Willems, A; Airoldi, L; De Clerck, O	Diversity and Temporal Dynamics of the Epiphytic Bacterial Communities Associated with the Canopy-Forming Seaweed Cystoseira cornpressa (Esper) Gerloff and Nizamuddin	FRONTIERS IN MICROBIOLOGY	2016	Italy	y	n	n	n

113	Bulleri, F; Cucco, A; Dal Bello, M; Maggi, E; Ravaglioli, C; Benedetti-Cecchi, L	The role of wave exposure and human impacts in regulating the distribution of alternative habitats on NW Mediterranean rocky reefs	ESTUARINE COASTAL AND SHELF SCIENCE	2016	Italy	y	n	n	n
114	Celis-Pla, P.S.M.; Bouzon, Z.L.; Hall-Spencer, J.M.; Schmidt, E.C.; Korbee, N; Figueroa, F.L.	Seasonal biochemical and photophysiological responses in the intertidal macroalga Cystoseira tamariscifolia (Ochrophyta)	MARINE ENVIRONMENTAL RESEARCH	2016	Spain	y	n	n	n
115	Bermejo, R; de la Fuente, G; Ramirez-Romero, E; Vergara, J.J.; Hernandez, I	Spatial variability and response to anthropogenic pressures of assemblages dominated by a habitat forming seaweed sensitive to pollution (northern coast of Alboran Sea)	MARINE POLLUTION BULLETIN	2016	Spain	y	n	n	n
116	Thiriet, P.D.; Di Franco, A; Cheminee, A; Guidetti, P; Bianchimani, O; Basthard-Bogain, S; Cottalorda, J.Ml; Arceo, H; Moranta, J; Lejeune, P; Francour, P; Mangialajo, L	Abundance and Diversity of Crypto- and Necto-BenthicCoastal Fish Are Higher in Marine Forests than in Structurally Less Complex Macroalgal Assemblages	PLOS ONE	2016	France	y	n	n	n
117	Capdevila, Pol; Hereu, Bernat; Lluis Riera, Juan; Linares, Cristina	Unravelling the natural dynamics and resilience patterns of underwater Mediterranean forests: insights from the demography of the brown alga Cystoseira zosteroides	JOURNAL OF ECOLOGY	2016	Spain	y	n	n	n
118	Bouafif, C; Verlaque, M; Langar, H	New contribution to the knowledge of the genus Cystoseira C. Agardh in the Mediterranean Sea, with the reinstatement of species rank for C-schiffneri Hamel	CRYPTOGAMIE ALGOLOGIE	2016	Tunisia	y	n	n	n
120	Montalvao, S; Demirel, Z; Devi, P; Lombardi, V; Hongisto, V; Perala, M; Hattara, J; Imamoglu, E; Tilvi, S.S; Turan, G; Dalay, M.C; Tammela, P	Large-scale bioprospecting of cyanobacteria, micro- and macroalgae from the Aegean Sea	NEW BIOTECHNOLOGY	2016	Turkey	y	n	n	n
122	Tsiamis, K; Salomidi, M; Kytinou, E; Issaris, Y; Gerakaris, V	On two new records of rare Cystoseira taxa (Fucales, Phaeophyceae) from Greece (Eastern Mediterranean)	BOTANICA MARINA	2016	Greece	y	n	n	n
123	Gianni, F; Mangialajo, L	Monitoring Mediterranean Marine Protected Areas: A set of guidelines to support the development of management plans.	MMMPA Supervisory Board	2016	Italy	n	y	n	n
124	Gianni, F; Bartolini, F; Pey, A; Laurent, M; Martins, G.M.; Airoldi, L; Mangialajo, L	Threats to large brown algal forests in temperate seas: the overlooked role of native herbivorous fish	SCIENTIFIC REPORTS	2017	France	y	n	n	n
125	Buonomo, R; Assis, J; Fernandes, F; Engelen, A.H.; Airoldi, L; Serrao, E.A.	Habitat continuity and stepping-stone oceanographic distances explain population genetic connectivity of the brown alga Cystoseira amentacea	MOLECULAR ECOLOGY	2017	Italy	y	n	y	n

127	Rindi, L; Dal Bello, M; Dai, L; Gore, J; Benedetti-Cecchi, L	Direct observation of increasing recovery length before collapse of a marine benthic ecosystem	NATURE ECOLOGY & EVOLUTION	2017	Italy	y	n	n	n
128	Mancuso, F.P; Strain, E.M.A.; Piccioni, E; De Clerck, O; Sarà, G; Airoldi, L	Status of vulnerable Cystoseira population along the Italian infralittoral fringe, and relationship with environmental and anthropogenic variables	MARINE POLLUTION BULLETIN	2017	Italy	y	n	n	y
129	El Asri, O; Ramdani, M; Latrach, L; Haloui, B; Mohamed, R; Afilal, M.E	Energetic valorization of Nador lagoon algae and proposal to use it as a means of elimination of the eutrophication in this lagoon	ECOLOGICAL ENGINEERING	2017	Morocco	y	n	n	n
130	Kersting, D. K.; Cebrian, E.; Verdura, J.; Ballesteros, E.	A new Cladocora caespitosa population with unique ecological traits	MEDITERRANEAN MARINE SCIENCE	2017	Spain	y	n	n	n
131	Melero, I.; Lopez-Velasco, S.; Lopez, E.	On the role of turf species as refuge in disturbed environments: A case study with polychaetes (Annelida: Polychaeta) in the SW Mediterranean Sea	MEDITERRANEAN MARINE SCIENCE	2017	Spain	y	n	n	n
132	Boada, J; Arthur, R; Alonso, D; Pages, J.F.; Pessarrodona, A; Oliva, S; Ceccherelli, G; Piazzi, L; Romero, J; Alcoverro, T	Immanent conditions determine imminent collapses: nutrient regimes define the resilience of macroalgal communities	PROCEEDINGS OF THE ROYAL SOCIETY B-BIOLOGICAL SCIENCES	2017	Spain	y	n	n	n
133	Celis-Pla, Paula S. M.; Martinez, Brezo; Korbee, Nathalie; Hall-Spencer, Jason M.; Figueroa, Flix L.	Ecophysiological responses to elevated CO2 and temperature in Cystoseira tamariscifolia (Phaeophyceae)	CLIMATIC CHANGE	2017	Spain	y	n	n	n
134	Aydin-Onen, S.; Ozturk, M.	Investigation of heavy metal pollution in eastern Aegean Sea coastal waters by using Cystoseira barbata, Patella caerulea, and Liza aurata as biological indicators	ENVIRONMENTAL SCIENCE AND POLLUTION RESEARCH	2017	Turkey	y	n	n	n
135	Sciuto, F; Sanfilippo, R; Alongi, G; Catra, M; Serio, D; Bejaoui, S; Leonardi, R; Viola, A; Rosso, A	First data on ostracods and foraminifera living in *Cystoseira* communities in western Ionian Sea (southern Italy, Mediterranean Sea)	MEDITERRANEAN MARINE SCIENCE	2017	Italy	y	n	n	n
136	Tsirintanis, K; Sini, M; Doumas, O; Trygonis, V; Katsanevakis, S	Assessment of grazing effects on phytobenthic community structure at shallow rocky reefs: An experimental field study in the North Aegean Sea	JOURNAL OF EXPERIMENTAL MARINE BIOLOGY AND ECOLOGY	2018	Greece	y	n	n	n
137	Crowe, P.T; Cusson, M; Bulleri, F; Davoult, D; Arenas, F; Aspden, R; Benedetti-Cecchi, L; Bevilacqua, S; Davidson, I; Defew, E; Fraschetti, S; Golléty, C; Griffin, N. J; Herkul, K; Kotta, J; Migné, A; Molis, M; Nicol, K. S; Noel, L. M-L J, Pinto, S. I;	Large-scale variation in combined impacts of canopy loss and disturbance on community structure and ecosystem functioning	PLoS ONE	2013	Italy	y	n	n	n

	Valdivia, N; Vaselli, S; Jenkins, R. S									
154	Chabane, K.; Bahbah, L.; Seridi, H.	Ecological Quality Status of the Algiers coastal waters by using macroalgae assemblages as bioindicators (Algeria, Mediterranean Sea)	MEDITERRANEAN MARINE SCIENCE	2018	Algeria	y	n	n	n	
155	Badreddine, A.; Abboud-Abi Saab M.; Gianni, F.; Ballesteros, E.; Mangialajo, L.	First assessment of the ecological status in the Levant Basin: Application of the CARLIT index along the Lebanese coastline	ECOLOGICAL INDICATORS	2018	Lebanon	n	y	y	y	
156	Mrdak, D.; Petrović, D.; Katnić, A.; Erceg, M.	Integrated study to support the designation of the trans-boundary Lake Skadar/Shkodra as biosphere reserve in the frame of the project, supporting the proposed trans-boundary biosphere reserve of Lake Skadar/Shkodra area through a participatory approach	University of Montenegro, Faculty of Sciences and Mathematics.	2011	Montenegro	y	n	n	n	
157	Kashta, L.; Zuna, V.; Dodbiba, E.; Beqiraj, S.; Kromidha, G.; Kocu, E.; Zotaj, A.; Tilot, V.; Burgt, N.	VLERËSIMI I MANGËSIVE NË ZONAT E MBROJTURA, BIODIVERSITETI DETAR DHE LEGJISLACIONI PËR ZONAT E MBROJTURA		2010	Albania		y	y	y	n
158	Kashta, L.; Beqiraj, S.; Tilot, V; Zuna, V.; Dodbiba, E.	The first MPA in Albania, Sazani Island - Karaburuni Peninsula, as a regional priority conservation area for marine biodiversity	VARSTVO NARAVE	2011	Albania	n	y	n	n	
159	Kashta, L.; Beqiraj, S.; Gace, A.; Mato, X.	STUDIM EKOLOGJIK I GJIRIT TË PORTO PALERMOS DHE ZONAVE PËRRETH	Nëntor	2013	Albania	n	y	n	n	
160	Kashta, L.; Beqiraj, S.; Mato, X.; Xhulaj, M.; Gaçe, A.; Mullaj, A	The inventory of habitats with Posidonia oceanica and littoral habitats in Albania.	Technical Report, APAWA, Tirana, supported by Ministry of Environment	2005	Albania	n	y	n	n	
161	Ruci, S.	Studim i makrozoobentosit të brigjeve shkëmbore të Adriatikut shqiptar	PhD Thesis	2015	Albania	y	y	n	n	
162	Hoda, P.; Mullaj, A.; Kashta, L.; Mahmutaj, E.; Mesiti, A.	Some rare and threatened habitats of Kavaja district	BSHN (UT)	2018	Albania	n	y	n	n	
163	Sacaj		MSc Thesis	2019	Albania	n	y	n	n	
168	Sala, E; Ballesteros, E; Dendrinos, P; Di Franco, A; Ferretti, F; Foley, D; Fraschetti, S; Friedlander, A; Garrabou, J; Güçlüsoy, H; Guidetti, P; Halpern, BS; Hereu, B; Karamanlidis, AA; Kizilkaya, Z; Macpherson, E; Mangialajo, L; Mariani, S; Micheli, F; Pais, A;	The Structure of Mediterranean Rocky Reef Ecosystems across Environmental and Human Gradients, and Conservation Implications	PLoS ONE	2012	Spain,Italy,Greece,Turkey	y	n	n	y	

	Riser, K; Rosenberg, AA; Sales, M; Selkoe, KA; Starr, R; Tomas, F; Zabala, M								
169	Tamburello, L; Ravaglioli, C; Mori, G; Nuccio, C; Bulleri, F	Enhanced nutrient loading and herbivory do not depress the resilience of subtidal canopy forests in Mediterranean oligotrophic waters	MARINE ENVIRONMENTAL RESEARCH	2019	Italy	y	n	n	y
170	Hinz, H; Renones, O; Gouraguine, A; Johnson, AF; Moranta, J	Fish nursery value of algae habitats in temperate coastal reefs	PEERJ	2019	Spain	y	n	y	y
171	Piazzi, L; Ceccherelli, G	Effect of sea urchin human harvest in promoting canopy forming algae restoration	ESTUARINE COASTAL AND SHELF SCIENCE	2019	Italy	y	n	n	y
172	Chiarore, A; Bertocci, I; Fioretti, S; Meccariello, A; Saccone, G; Crocetta, F; Patti, FP	Syntopic *Cystoseira* taxa support different molluscan assemblages in the Gulf of Naples (southern Tyrrhenian Sea)	MARINE AND FRESHWATER RESEARCH	2019	Italy	y	n	n	y
173	Cuadros, A; Moranta, J; Cardona, L; Thiriet, P; Francour, P; Vidal, E; Sintes, J; Cheminee, A	Juvenile fish in *Cystoseira* forests: influence of habitat complexity and depth on fish behaviour and assemblage composition	MEDITERRANEAN MARINE SCIENCE	2019	Spain	y	n	n	y
174	Catra, M; Alongi, G; Leonardi, R; Negri, MP; Sanfilipo, R; Sciuto, F; Serio, D; Viola, A; Rosso, A	Degradation of a photophilic algal community and its associated fauna from eastern Sicily (Mediterranean Sea)	MEDITERRANEAN MARINE SCIENCE	2019	Italy	y	n	y	y
175	Gianni, F; Bartolini, F; Airoldi, L; Mangialajo, L	Reduction of herbivorous fish pressure can facilitate focal algal species forestation on artificial structures	MARINE ENVIRONMENTAL RESEARCH	2018	France	y	n	y	y
176	Kletou, D; Savva, I; Tsiamis, K; Hall-Spencer, JM	Opportunistic seaweeds replace *Cystoseira* forests on an industrialised coast in Cyprus	MEDITERRANEAN MARINE SCIENCE	2018	Cyprus	y	n	n	y
177	Piazzi, L; Gennaro, P; Atzori, F; Cadoni, N; Cinti, MF; Frau, F; Ceccherelli, G	ALEX index enables detection of alien macroalgae invasions across habitats within a marine protected area	MARINE POLLUTION BULLETIN	2018	Italy	y	n	n	y
178	Buosi, A; Sfriso, A	Macrophyte assemblage composition as a simple tool to assess global change in coastal areas. Freshwater impacts and climatic changes	SCIENCE OF THE TOTAL ENVIRONMENT	2017	Italy	y	n	n	y
179	Bulleri, F; Benedetti-Cecchi, L; Ceccherelli, G; Tamburello, L	A few is enough: a low cover of a non-native seaweed reduces the resilience of Mediterranean macroalgal stands to disturbances of varying extent	BIOLOGICAL INVASIONS	2017	Italy	y	n	n	y
180	Piazzi, L; Pardi, G; Balata, D; Cecchi, E; Cinelli, F	Seasonal dynamics of a subtidal north-western Mediterranean macroalgal community in relation to depth and substrate inclination	BOTANICA MARINA	2002	Italy	y	n	y	y
181	Russo, AR	Epifauna living on sublittoral seaweeds around Cyprus	HYDROBIOLOGIA	1997	Cyprus	y	n	n	y

182	Benedetti-Cecchi, L; Cinelli, F	Canopy removal experiments in *Cystoseira*-dominated rockpools from the western coast of the Mediterranean (Ligurian Sea)	JOURNAL OF EXPERIMENTAL MARINE BIOLOGY AND ECOLOGY	1992	Italy	y	n	n	y
183	Devescovi, M	Effects of bottom topography and anthropogenic pressure on northern Adriatic Cystoseira spp. (Phaeophyceae, Fucales)	AQUATIC BOTANY	2015	Croatia	y	n	n	y
184	Cardona, L; Moranta, J; Renones, O; Hereu, B	Pulses of phytoplanktonic productivity may enhance sea urchin abundance and induce state shifts in Mediterranean rocky reefs	ESTUARINE COASTAL AND SHELF SCIENCE	2013	Spain	y	n	n	y
185	Alongi, G; Catra, M; Cormaci, M; Furnari, G; Serio, D	Spring marine vegetation on rocky substrata of Pantelleria Island (the Straits of Sicily, Italy)	NOVA HEDWIGIA	2004	Italy	y	n	y	y
186	Orlando-Bonaca, M; Lipej, L	Benthic Macroalgae as bioindicators of the ecological status in the Gulf of Trieste	VARSTVA NARAVE	2009	Slovenia	y	n	n	y
187	Orlando-Bonaca, M; Lipej, L; Orfanidis, S	Benthic macrophytes as a tool for delineating, monitoring and assessing ecological status: The case of Slovenian coastal waters	MARINE POLLUTION BULLETIN	2008	Slovenia	y	n	n	y
188	Orlando-Bonaca, M; Rotter, A	Any sing of replacement of canopy-forming algae by turf forming algae in the northern Adriatic Sea?	ECOLOGICAL INDICATORS	2018	Slovenia	y	n	n	n
189	Tomas, F; Cebrian, E; Ballesteros, E	Differential herbivory of invasive algae by native fish in the Mediterranean Sea	ESTUARINE COASTAL AND SHELF SCIENCE	2011	Spain	y	n	n	n
190	Agnetta, D.; Bonaviri, C.; Badalamenti, F.; Scianna. C.; Vizzini, S.; Gianguzza, P.	Functional traits of two co-occurring sea urchins across a barren/forest patch system	JOURNAL OF SEA RESEARCH	2013	Italy	y	n	n	n
191	Benfares, R.; Seridi, H.; Belkacem, Y.; Inal, A.	Heavy Metal Bioaccumulation in Brown Algae Cystoseira compressa in Algerian Coasts, Mediterranean Sea	ENVIRONMENTAL PROCESSES	2015	Algeria	n	n	n	n
192	Bianchi, C.N.; Corsini-Foka, M.; Morri, C.; Zenetos, A.	Thirty years after: dramatic change in the coastal marine ecosystems of Kos Island (Greece), 1981-2013	MEDITERRANEAN MARINE SCIENCE	2014	Greece	y	n	y	n
193	Blanfunè, A.; Boudouresque, C.F.; Verlaque, M.; Thibaut, T.	The ups and downs of a canopyforming seaweed over a span of more than one century	SCIENTIFIC REPORTS	2019	France	y	y	y	y
194	Capdevila, P.; Linares, C.; Aspillaga, E.; Riera, J.L.; Hereu, B.	Effective dispersal and density-dependence in mesophotic macroalgal forests: Insights from the Mediterranean species Cystoseira zosteroides	PLOS ONE	2018	Spain	y	n	n	n
195	Cheminée, A.; Pastor, J.; Bianchimani, O.; Thiriet, P.; Sala, E.; Cottalorda, J.M.; Dominici, J.M.; Lejeune, P.; Francour, P.	Juvenile fish assemblages in temperate rocky reefs are shaped by the presence of macro-algae canopy and its three-dimensional structure	SCIENTIFIC REPORTS	2017	France	y	n	n	n
196	De La Fuente, G.; Asnaghi, V.; Chiantore, M.M Thrush, S.; Povero, P.;	The effect of Cystoseira canopy on the value of midlittoral habitats in NW Mediterranean, an emergy assessment	ECOLOGICAL MODELLING	2019	Italy	y	n	n	n

	Vassallo, P.; Petrillo, M.; Paoli, C.								
197	Mariani, S.; Cefalì, M.E.; Chappuis, E.; Terradas, M.; Pinedo, S.; Torras, X.; Jordana, E.; Medrano.; Verdura, J.; Ballesteros, E.	Past and present of Fucales from shallow and sheltered shores in Catalonia	REGIONAL STUDIES IN MARINE SCIENCE	2019	Spain	y	n	y	y
198	Medrano, A.; Linares, C.; Aspillaga, E.; Capdevila, P.; Montero-Serra, I.; Pagès-Escolà, M.; Zabala, M.; Hereu, B.	Long-term monitoring of temperate macroalgal assemblages inside and outside a No take marine reserve	MARINE ENVIRONMENTAL RESEARCH	2020	Spain	y	n	n	n
199	Milazzo, M.; Badalamenti, F.; Riggio, S.; Chemello, R.	Patterns of algal recovery and small-scale effects of canopy removal as a result of human trampling on a Mediterranean rocky shallow community	BIOLOGICAL CONSERVATION	2004	Italy	y	n	n	n
200	Pizzuto, P.	On the structure, typology and periodism of a cystoseira brachycarpa J. Agardh emend. Giaccone community and of a cystoseira crinita duby community from the eastern coast of Sicily (Mediterranean Sea)	PLANT BIOSYSTEMS	1999	Italy	y	n	n	n
201	Thibaut, T.; Blanfuné, A.; Markovic, L.; Verlaque, M.; Boudouresque, C.; Perret-Boudouresque, M.; Macic, V.; Bottin, L.	Unexpected abundance and long-term relative stability of the brown alga Cystoseira amentacea, hitherto regarded as a threatened species, in the north-western Mediterranean Sea	MARINE POLLUTION BULLETIN	2014	France	n	y	n	y
202	Curiel, D.; Dri, C., Scattolin, M.; Marzocchi, M.	Area minima di campionamento per lo studio qualitativo di un popolamento di Cystoseira barbata (Stackhouse) C. Agardh var. barbata e Undaria pinnatifida (Harvey) Suringar in Laguna di Venezia.	Boll. Mus. civ. St. nat. Venezia	2002	Italy	y	n	n	n
203	Demoulin, V.; Janssen, M.-P.; Licot, M.	Mise au point d'une méthode de cartographie des macroalgues marines application à la région de Calvi (Corse).	Lejeunia	1980	France	y	n	y	n
204	Feoli, E. & Bressan, G.	Affinità floristica dei tipi di vegetazione bentonica della Cala di Mitigliano (Massa Lubrense, Napoli).	Giornale Botanico Italiano	1972	Italy	y	n	n	n
205	Ferrario, F.	Analysis and development of ecologically based approaches to coastal defense.	PhD Thesis Università di Bologna	2013	Italy	y	y	y	y
206	Flores-Moya, A. & Conde, F.	Fragmentos taxonómicos, corológicos, nomenclaturales y fitocenológicos. Nueva citas de macroalgas marinas para las Islas Chafarinas.	Acta Botanica Malacitana	1998	Morocco	y	n	n	n
207	Feldmann, J.	Les algues marines de la côte des Albères. I-III. Cyanophycées, Chlorophycées, Phaéophycées.	Revue Algologique	1937	France	y	n	n	n
208	Gennaro, P., Piazzi, L., Persia, E. & Porrello, S.	Responses of macroalgal assemblages dominated by three Mediterranean brown macroalgae with different life strategies to nutrient enrichment.	European Journal of Phycology	2019	Italy	y	n	n	y
209	Einav, R., & Israel, A.	Checklist of seaweeds from the Israeli Mediterranean: taxonomical and ecological approaches.	Israel Journal of Plant Sciences	2008	Israel	y	n	n	n

210	Fiorin R., Cerasuolo C., Curiel D. & Riccato F.	Il popolamento ittico e macroalgale delle scogliere del litorale veneziano: interazione tra le alghe brune del genere Cystoseira e alcune specie di pesci	Biol. Mar. Mediterr.	2008	Italy	n	y	n	n
211	Bachir Bouiadjra, B.; Belbachir, N.; Youcef Benkada, M.; Maarouf, A.; Riadi, H.	Sur la presence de l'algue marine Caulerpa racemosa (Forsskal) J. Agardh (Caulerpales, Chlorophyta) devant la côte Mostaganemoise (Ouest Algerie).	Acta Botanica Malacitana	2010	Algeria	y	n	n	n
212	Baggini, C.	Assessing the effects of long-term ocean acidification on benthic communities at CO2 seeps.	PhD Thesis Plymouth and Bremen University	2014	Greece, Italy	y	n	n	y
213	Orfanidis, S., Ivesa, L., Gounaris, S., Tsioli, S., Devescovi, M. & Papathanasiou, V.	Cystoseira scale-based biometric relationships.	Botanica Marina	2017	Greece, Cyprus, Croatia	y	n	n	n
214	Ould-Ahmed, N., Gómez Garreta, A., Ribera Siguan, M.A. & Bouguedoura, N.	Checklist of the benthic marine macroalgae from Algeria. I. Phaeophyceae.	Anales del Jardín Botánico de Madrid	2013	Algeria	y	n	n	n
215	Percot, A., Yalcin, A., Aysel, V. Erdugan, H., Dural, B. & Guven, K.C.	B-Phenylethylamine content in marine algae around Turkish coasts.	Botanica marina	2009	Turkey	y	n	n	n
216	Perret-Boudouresque, M. & Seridi, H.	Inventaire des algues marines benthiques d'algerie	GIS Posidonie publ., Marseille.	1989	Algeria	y	n	n	n
217	Pérez-Ruzafa, A., Hegazi M.M.I., Pérez-Ruzafa, I.M., & Marcos, C.	Differences in spatial and seasonal patterns of macrophyte assemblages between a coastal lagoon and the open sea.	Marine Environmental Research	2008	Spain	y	n	y	n
218	Piccone, A.	Elenco delle alghe della Crociera del Corsaro alle Baleari.	Genova [Genoa]: Tipografia del r. Istituto Sordo-Muti.	1889	Spain, Italy	y	n	n	n
219	Ramon, E.	Cystoseira rayssiae: a new Cystoseira (Cystoseiraceae, Fucophyceae) from the shores of Israel, Eastern Mediterranean Sea.	Israel Journal of Plant Sciences	2000	Israel	y	n	n	n
220	Ribera, G., Coloreu, M., Rodriguez Prieto, C. & Ballesteros, E.	Phytobenthic assemblages of Addaia Bay (Menorca, Western Mediterranean): composition and distribution.	Botanica Marina	1997	Spain	y	n	n	n
221	Ribera, M.A., Gómez-Garreta, A., Barcelo, M.A., Rull Lluch, J.	Mapas de distribución de algas marinas de la Península Ibérica e Islas Baleares. VIII. Cystoseira C. Agardh y Sargassum C. Agardh.	Botanica Complutensis	1996	Spain	y	n	n	n
222	Robvieux, P.	Conservation des populations de Cystoseira en régions Provence-Alpes-Côte-d'Azur et Corse.	PhD thesis Université de Nice-Sophia Antipolis	2013	France	y	n	n	n
223	Rodriguez Prieto, C. & Polo Alberti, L.	Anàlisi fitosociològica de la comunitat de Cystoseira mediterranea de Palamós (Mediterrània nordoccidental).	Acta Botanica Barcinonensia. Universitat de Barcelona	1998	Spain	y	n	n	y
224	Rodríguez-Prieto, C. & Vergés, A.	Addicións i noves senyalitzacions per a la flora d'algues bentòniques marines de la costa catalana.	Acta Botanica Barcinonensia. Universitat de Barcelona	2000	Spain	y	n	n	n
225	Rull Lluch, J. & Gómez Garreta, A.	Estudio de la comunidad de Cystoseira mediterranea en dos localidades de la peninsula del cabo de Creus (Alt Empordà, Nordeste de España).	Collec. Bot. (Barcelona)	1990	Spain	y	n	n	y

226	Rull Lluch, J., Gómez Garreta, A., Barceló, M.C. & Ribera, M.A.	Mapas de distribución de algas marinas de la Península Ibérica e Islas Baleares. VII. Cystoseira C. Agardh (Grupo C. baccata) y Sargassum C. Agardh (S. muticum y S. vulgare).	Botanica Complutensis	1994	Spain	y	n	n	n
227	Sellam, L.N., Blanfuné, A., Boudouresque, C.F., Thibaut, T., Rebzani Zahaf, C. & Verlaque, M.	Cystoseira montagnei J. Agardh and C. spinosa Sauvageau (Phaeophyceae, Sargassaceae): a taxonomic reappraisal of misused names, with the proposal of Cystoseira michaelae Verlaque et al. nom. et stat. nov.	Cryptogamie Algologie	2017	Algeria	y	n	n	n
228	Seoane-Camba, J.	Estudios sobre las algas bentónicas en la costa sur de la Península Ibérica (litoral de Cádiz).	Investigacion Pesquera	1965	Spain	y	n	n	n
229	Seoane-Camba, J.A.	Algas bentónicas españolas en los herbarios Thuret-Bornet y Sauvageau del Muséum National d'Histoire Naturelle de Paris. II.	Anal. Inst. Bot. Cavanilles	1975	Spain	y	n	n	n
230	Ballesteros, E.	Contribució al coneixement algològic de la Mediterrània Espanyola: algues bentòniques i litorales de Tossa de Mar (Girona).	Butll. Inst. Cat. Hist. Nat	1981	Spain	y	n	n	n
231	Ballesteros, E.	Contribució al coneixement algològic de la Mediterrània Espanyola. VI.	Collectanea Botanica (Barcelona)	1985	Spain	y	n	n	n
232	Barceló i Marti, M.C. & Seoane-Camba, J.A.	Nota sobre la variacion fenologica de Cystoseira sauvageauana Hamel en las costas de Alicante.	Anales de Biologia	1984	Spain	y	n	n	n
233	Barcelo, M.C. & Seoane, J.A.	Aportació al coneixement de les algues marines de les costes del País Valencià.	Collectanea Botanica	1982	Spain	y	n	n	n
234	Barceló, M.C., Gómez Garreta, A., Rull Lluch, J. & Ribera, M.A.	Mapas de distribución de algas marinas de la Península Ibérica e Islas Baleares. VI. Cystoseira C. Agardh: Grupos C. spinifero-opuntioides y C. discors-abrotanifolioides.	Botanica Complutensis	1994	Spain	y	n	n	n
235	Ben Maiz, N., Boudouresque, C.-F. & Quahchi, F.	Inventaire des algues et phanérogames marines benthiques de la Tunise.	Giornale Botanico Italiano	1987	Tunisia	y	n	n	n
236	Alongi, G.; Catra, M.; Cormaci, M.; Furnari, G.	Observations on Cystoseira squarrosa De Notaris (Fucophyceae, Fucales), a rare and little known Mediterranean species and its typification	BOTANICA MARINA	2002	Italy	y	n	n	n
237	Bottalico, A.; Alongi, G.; Perrone, C.	Macroalgal diversity of Santa Cesarea-Castro (Salento Peninsula, southeastern Italy)	ANALES DEL JARDIN BOTANICO DE MADRID	2016	Italy	y	n	n	n
238	Catra, M.; Alongi, G.; Serio, D.; Cormaci, M.; Furnari, F.	The benthic algal flora in rocky substrata of the Egadi Islands, a marine protected archipelago off the western coast of Sicily (Italy, Mediterranean Sea)	NOVA HEDWIGIA	2006	Italy	y	n	n	n
239	Cormaci, M; Lanfranco, E.; Borg, J.A.; Buttigieg, S.; Furnari, G.; Micallef, S.A.; Mifsud, C.; Pizzuto, F.; Scammacca, B.; Serio, D.	Contribution to the Knowledge of benthic marine algae on rocky substrata of the Maltese Islands (Mediterranean Sea).	BOTANICA MARINA	1997	Malta	y	n	y	n
240	Cormaci, M.; Furnari, G; Alongi, G.; Catra, M.; Pizzuto, F.; Serio, D.	Spring marine vegetation on rocky substrata of the Tremiti Islands (Adriatic Sea, Italy)	MEDITERRANEAN ECOSYSTEMS STRUCTURES AND PROCESSES	2001	Italy	y	n	y	n

241	Thibaut, T.; Bottin, L.; Aurelle, D.; Boudouresque, C.F., Blanfuné, A.; Verlaque, M.; Pairaud, I.; Millet, B.	Connectivity of populations of the seaweed Cystoseira amentacea within the Bay of Marseille (Mediterranean Sea): genetic structure and hydrodynamic connections	CRYPTOGAMIE ALGOLOGIE	2016	France	y	n	n	n
242	Cecere, E.; Cormaci, M.; Furnari, G.; Petrocelli, A.; Saracino, O.; Serio, D.	Benthic algal flora of Cheradi Islands (Gulf of Taranto, Mediterranean Sea)	NOVA HEDWIGIA	1996	Italy	y	n	y	n
243	Gianni, F.	Conservation et restauration écologique des forêts marines Méditerranéennes.	PhD Thesis University of Nizza	2016	France, Italy	y	y	y	y
244	Hamza, A.	Étude des Cytoseiras du Golfe de Gabès: notes préliminaires.	Bull. Inst. natn tech. Océanogr. Pêche Salammbó	1987	Tunisia	y	n	n	n
245	Haritonidis, S. & Tsekos, I.	Marine algae of the Greek west coast.	Botanica Marina	1976	Greece	y	n	n	n
246	Hoffman, R., Sternberg, M. & Serio, D.	First report of Laurencia chondrioides (Ceramiales, Rhodophyta) and its potential to be invasive in the eastern Mediterranean Sea.	Botanica Marina	2014	Israel	y	n	y	n
247	Mancuso, F.P.	Effects of local and global stressors on the status and future persistence of intertidal canoyp-forming algae.	PhD Thesis Università di Bologna	2016	Italy	y	n	y	y
248	Medrano Cuevas, A.	Macroalgal forests ecology, long-term monitoring, and conservation in a Mediterranean Marine Protected Area.	PhD Thesis Universitat de Barcelona	2020	Spain	y	n	n	y
249	Giaccone, G.	Raccolte di fitobenthos sulla banchina continentale Italiana.	Giornale Botanico Italiano	1969	Italy	n	y	y	n
250	Gómez Garreta, A.	Estudio fenológico de la vegetación marina de la isla de Mallorca.	Universidad Complutense de Madrid	1983	Spain	n	y	n	y
251	Marzocchi M., Curiel D., Dri C., Scattolin M.	Fenologia morfologica e riproduttiva di Cystoseira barbata (Stachkouse) C. Agardh var. barbata (Fucales, Fuciphyceae) nella laguna di Venezia (nord Adriatico).	Soc. Ven. Sc. Nat.	2003	Italy	n	y	n	n
252	Kletou, D.	Human impacts on oligotrophic marine ecosystems: case studies from Cyprus, Mediterranean Sea.	PhD Thesis Plymouth University	2019	Cyprus	n	y	y	y
253	Grech, D.	Historical Records And Current Status Of Fucales (Cystoseira and Sargassum spp.) In The Gulf of Naples.	PhD thesis The Open University.	2017	Italy	y	y	y	n
254	Falace F. & Bressan G	Some observations on periphyton colonization of artificial substrata in the Gulf of Trieste	Bulletin of Marine Science	1994	Italy	y	n	n	n
255	Gianguzza P., Bonaviri C., Milisenda G., Barcellona A., Agnetta D., Vega Fernández T., Badalamenti F	Macroalgal assemblage type affects predation pressure on sea urchins by altering adhesion strength.	Marine environmental research	2010	Italy	y	n	n	n
256	Schiapparelli S., Castellano M., Povero P., Sartoni G., Cattaneo-Vietti R.	A benthic mucilage event in North-Western Mediterranean Sea and its possible relationships with the summer 2003 European heatwave: short term effects on littoral rocky assemblages.	Marine Ecology	2007	Italy	y	n	n	n

257	Catra M., Giardina S., Giaccone T., Basos D, Giaccone G.	Il coralligeno del Secchitello di Ustica (Palermo).	Biol. Mar. Mediterr.	2007	Italy	y	n	n	n
258	Bellón, L.	Contribución al estudio de la flora algológica del Mediterráneo español.	Boletín de Pescas	1921	Spain	y	n	n	n
261	Diapoulis, A., Haritonidis, S. & Koussouris, T.	Spring benthic flora of Rhodos Island, Greece.	Thalassographica	1986	Greece	y	n	n	y
262	Moussa, H., Hassoun, M., Salhi, G., Zbakh, H. & Riadi, H.	Checklist of seaweeds of Al-Hoceima National Park of Morocco (Mediterranean Marine Protected Area).	Acta Botanica Malacitana	2018	Morocco	y	n	n	n
263	Tsiamis, K., Panayotidis, P., Economou-Amilli, A. & Katsaros, C.	Seaweeds of the Greek coasts. I. Phaeophyceae.	Mediterranean Marine Science	2013	Greece	y	n	n	n
264	González García, J.A., Paredes Ruiz, P. & Enrique Mirón, C.	Catálogo de macroalgas de los Peñones de Alhucemas y Vélez de la Gomera (Mediterráneo occidental, Mar de Alborán).	Algas	2017	Morocco	y	n	n	n
265	Pérez-Ruzafa, I. & Honrubia, M.	Aportación al conocimiento de la flora algal bentónica de la costa murciana. III.	Anales de Biología. Secretariado de Publicaciones. Universidad de Murcia	1984	Spain	y	n	n	n
266	Verlaque, M., Ballesteros, E., Sala, E. & Garrabou, J.	Cystoseira jabukae (Cystoseiraceae, Fucophyceae) from Corsica (Mediterranean) with notes on the previously misunderstood species of C. funkii.	Phycologia	1999	France, Italy	y	n	y	n
267	Verlaque, M.	Biologie des juvéniles de l'oursin herbivore Paracentrotus lividus (Lamarck): séléctivité du broutage et impact de l'espèce sur les communautés algales de substrat rocheux en Corse (Méditerranée, France).	Botanica Marina	1984	France	y	n	n	n
268	Bermejo, R., Chefaoui, R.M., Engelen, A.H., Buonomo, R., Neiva, J., Ferreira-Costa, J., Pearson, G.A., Marbà, N., Duarte, C.M., Airoldi, L., Hernández, I., Guiry, M.D. & Serrão, E.A.	Marine forests of the Mediterranean-Atlantic Cystoseira tamariscifolia complex show a southern Iberian genetic hotspot and no reproductive isolation in parapatry.	Scientific Reports	2018	Spain,Italy	y	n	n	n
269	Gómez Garreta, A., Ribera, M.A., Barceló, M.C. & Rull Lluch, J.	Mapas de distribución de algas marinas de la Península Ibérica e Islas Baleares. V. Cystoseira C. Agardh: Grupos C. ericaefolia y C. crinito-selaginoides.	Botanica Complutensis	1994	Spain	y	n	n	n
270	Christia, C., Tziortzis, I., Fyttis, G., Kashta, L. & Papastergiadou, E.	A survey of the benthic aquatic flora in transitional water systems of Greece and Cyprus (Mediterranean Sea).	Botanica marina	2011	Greece	y	n	y	n
271	Drago D., Catra M., Giaccone G., Giardina S., Grimaldi S., Mannino A.	La flora e la vegetazione sommersa del porto di Cala Santa Maria nell'isola di Ustica (Pa): contributo allo Studio di Impatto Ambientale (SIA).	Bollettino dell'Accademia Gioenia di Scienze Naturali di Catania	2004	Italy	y	n	n	n
272	Gianguzza P., Bonaviri C., Chemello R., Riggio S.	Remarkable presence of Aplysia punctata (Mollusca, Opisthobranchia, Anaspidea) in the Ustica Island MPA (Western Mediterranean, Italy).	Biologia Marina Mediterranea	2004	Italy	y	n	n	n

273	Augier H., Boudouresque C.F.	Végétation marine de Port–Cros (Parc national) XVI: contribution a` l’étude de l’épiflore du détritique coˆ tier.	Trav. Sci. Parc Nation. Port–Cros	1978	France	y	n	n	n
274	Belsher, T., Augier, H., Boudouresque, C.F., Coppejans, E.	Inventaire des algues marines benthiques de la rade et des ıˆles d’Hyères (Méditerranée, France).	Trav. Sci. Parc Nation. Port–Cros	1976	France	y	n	n	n
275	Coppejans, E.	Bijdrage tot de studie van de wierpopulaties (Chlorophyceae, Phaeophyceae, Rhodophyceae) van het fotofiel infralittoraal in het noordwestelijk mediterraan bekken. Deel I: tekst	Thesis, University of Ghent, Belgium	1977	France	y	n	n	y
276	Curiel D., Bellemio G., Marzocchi M., Iuri M., Scattolin M.	Variazioni della flora bentonica macroalgale nei substrati duri della laguna di Venezia: confronto con rilievi del 1938	Lav. Soc. Ven. Sc. Nat.	2001	Italy	y	n	y	n
277	Hereu, B., Zabala, M., Ballesteros, E.	On the occurrence of a population of Cystoseira zosteroides Turner and Cystoseira funkii Schiffner ex Gerloff et Nizamuddin (Cystoseiraceae, Fucophyceae) in the Port-Cros National Park (Northwestern Mediteranean, France).	Sci. Rep. Port-Cros Natl. Park	2003	France	y	n	n	n
278	Serio, D.	On the structure, typology and periodism of Cystoseira spinosa Sauvageau community and of Cystoseira zosteroides C. Agardh community from Eastern coast of Sicily (Mediterranean Sea)	Giornale Botanico Italiano	1994	Italy	y	n	n	y
279	Bottalico, A. & Delle Foglie, C.I.	Contribution to the knowledge of the benthic marine flora along the eastern coastline of the Gulf of Taranto (Ionian Sea).	Flora Mediterranea	2003	Italy	y	n	n	y
280	Cecere, E., Petrocelli, A. & Saracino	Biodiversity of phyto-benthic communities in the marine reserve of Porto Cesareo.	Biologia Marina Mediterranea	2005	Italy	y	n	y	n
281	Calvo, S., G. Giaccone and S. Ragonese.	Tipologia della vegetazione sommersa dello stagnone di Marsala (TP)	Naturalista Siciliano	1982	Italy	y	n	y	y
282	Cinelli, F., Fresi, E., Idato, E., & Mazzella L.	L'aire minima du phytobenthos dans un peuplement à *Cystoseira mediterranea* de l'Ile d'Ischia (Golfe de Naples)	Rapp. Comm. Int. Mer Médit	1977	Italy	y	n	n	n
283	De Toni, G. B.	Sylloge algarum omnium hucusque cognitarum. Vol III Fucoideae	patavii	1895	Italy	y	n	n	n
284	Ghirardell, Orel & Giaccone	L'inquinamento del golfo di Trieste	Atti Mus. Civico Storia Nat. Trieste	1973	Italy	y	n	n	n
285	Giaccone, G.	Lineamenti della vegetazione lagunare dell' Alto Adriatico ed evoluzione in conseguenza dcII' inquinamento	BolI. Mus. Civico Storia Naturale Venezia	1974	Italy	y	n	n	n
286	De Leo, A. & Giaccone, G.	Flora e vegetazione algale del Golfo di Palermo: litorale dell'Allaura. I controbuto	Lav. Ist. Bot. Giard. Col. Palermo	1964	Italy	n	y	n	y
287	Giaccone, G.	Le fitocenose marine del settore rosso di Capo Zafferano (Palermo)	Lav. Ist. Bot. Giard. Colon. Palermo	1967	Italy	n	y	y	y
288	Giaccone, G. & De Leo, A.	Flora e vegetazione algale del Golfo di Palermo: litorale dell'Allaura. I controbuto	Lav. Ist. Bot. Giard. Colon. Palermo	1964	Italy	n	y	n	y

289	Giaccone, G. & Pignatti, S.	La vegetazione del Golfo di Trieste	Inform. Bot. Ital.	1967	Italy	y	n	n	y
290	Giaccone, G. Cormaci, M., Furnari, G., Scammacca, B., Alongi, G., Catra, M., Di Martino, V., Marino, G., Serio, D.	Biodiversità vegetale marina dell'arcipelago 'Isole Eolie	Boll. Accad. Gioenia Sci.Nat.	1999	Italy	y	n	n	n
291	Giaccone, G., Alongi, G., Cossu, A.V.L., Di Geronimo, R., Serio, D.	La vegetazione marina bentonica del Mediterraneo: I. Sopralitorale e Mesolitorale. Proposte di aggiornamento	Boll. Accad. Gioenia Sci.Nat.	1993	Italy	y	n	n	n
292	Cecere, E., Petrocelli, A., Saracino, O.D., Cormaci, M. & Furnari, G.	Marine benthic flora of the Gargano promontory (Adriatic Sea, south-ern Italy).	Flora Mediterranea	2000	Italy	y	n	y	n
293	Serio, D., Furnari, G. & Cormaci, M.	On the occurrence of Laurencia pyramidalis Bory ex Kutzing (Rhodophyta, Rhodomelaceae) in the Mediterranean Sea.	Cryptogamie Algologie	2004	Italy	y	n	n	n
294	Tsekos, I. & Haritonidis, S.	A survey of the marine algae of the Ionian Islands, Greece.	Botanica Marina	1977	Greece	y	n	n	n
295	Tsiamis, K., Verlaque, M., Panayotidis, P. & Montesanto, B.	New macroalgal records for the Aegean Sea (Greece, eastern Mediterranean Sea).	Botanica Marina	2010	Greece	y	n	y	n
296	Verlaque, M.	Checklist of the macroalgae of Thau Lagoon (Hérault, France), a hot spot of marine species introduction in Europe.	Oceanologica Acta	2001	France	y	n	n	n
297	Serio, D.	Fenologia morfologica e riproduttiva di Cystoseira spinosa Sauvageau v. compressa (Ercegovic) Cormaci et al. (Fucales, Fucophyceae).	Boll. Acc. Gioenia. Sci. Nat.	1995	Italy	y	n	n	n
298	Cecere E.	Economically important seaweeds in Mar Piccolo, Taranto (southern Italy): a survey.	Proc. 13th International Seaweed Symposium, Hydro biol	1990	Italy	y	n	n	n
299	Cecere E. and Perrone C	First contribution to the knowledge of macrobenthic flora of the Amendolara sea-mount (Ionian sea)	Oebalia	1988	Italy	y	n	n	n
300	CORMACI, M. & G. FURNARI	Sulla presenza nell'Italia meridionale di alcune alghe marine bentoniche rare per il Mediterraneo	Giorn. Bot. Ital.	1988	Italy	y	n	n	n
301	CORMACI , M., G. FURNARI, B. SCAMMACCA, D. SERIO, F. PIZZUTO, G. ALONGI & R. DINARO	La vegetazione marina di substrato duro dell'isola di Salina (Isole Eolie)	Boli. Ace.GioeniaSe.Nat.	1992	Italy	y	n	n	y
302	Giaccone G., B. Scammacca, F. Cinelli, G. Sartoni and G. Furnari	Studio preliminare sulla tipologia della vegetazione sommersa del Canale di Sicilia e isole vicine.	Giorn. Bot. Ital.	1972	Italy	y	n	n	n
303	Pignatti S., De Cristini P., Rizzi L.	Le associazioni algali della Grotta delle Viole nell'isola di S. Domino (Is. Tremiti).	Giorn. Bot. Ital.	1967	Italy	y	n	n	n
304	Rizzi Longo L.	La flora sottomarina delle Isole Tremiti	Atti Ist. Veneto Sci. Lett. Arti, CI. Sci. Mat. Nat.	1972	Italy	y	n	n	n

305	Sfriso, A.	Flora and vertical distribution of macroalgae in the lagoon of Venice: a comparison with previous studies	Giorn. Bot. Ital.	1987	Italy	y	n	y	n
306	Falace, A.; Zanelli, E. and Bressan G.	Morphological and Reproductive Phenology of Cystoseira Compressa (Esper) Gerloff & Nizamuddin (Fucales, Fucophyceae) in the Gulf of Trieste (North Adriatic.	Annales. Series Historia Naturalis	2005	Italy	y	n	n	n
307	Pignatti, A. & Rizzi-Longo, L.	Raccolte di alghe bentoniche nelle acque dell'Arcipelago Toscano	Atti Ist. Veneto Sci. Lett e Arti	1987	Italy	y	n	n	n
308	Cinelli, F.	Alghe bentoniche di profondità raccolte alla punta S. Pancrazio nell'Isola di Ischia (Golfo di Napoli).	Giornale Botanico Italiano	1971	Italy	y	n	n	n
309	Celis Plà, P.S.M.	Vulnerability and acclimation of Mediterranean sea macroalgae, to environmental stress related to climate change: use of indicators physiological state.	PhD Thesis Universitat de Barcelona	2015	Spain	y	n	n	n
310	Capdevila Lanzaco, P.	Life history, population dynamics and conservation of underwater Mediterranean forests: insights from the long-lived alga Cystoseira zosteroides.	PhD Thesis Universitat de Barcelona	2017	Spain	y	n	n	n
311	Buonomo, R.	Genetic background, range shifts and associated microbial responses of canopy algae under changing environment.	PhD Thesis Università di Bologna	2017	Italy	y	n	n	n
312	Boisset, F. & Fomez Garreta, A.	Aproximación al análisis fitosociológico de la flora algal del subestrato de Cystoseira mediterranea Sauv. en el litoral valenciano (España, Mediterráneo).	Botanica Complutensis	1989	Spain	y	n	n	n
313	Coppejans, E.	Végétation marine de la Corse (Méditerranée). III. Documents pour la flore des algues.	Botanica Marina	1979	France	y	n	n	n
314	Pardi, G., I. Papi, L. Piazzi and F. Cinelli	Benthic marine flora in the Tuscan Archipelago. A second contribution: Isle of Gorgona	Giorn.Bot.Ital.	1993	Italy	y	n	n	n
315	Serio, D.	Fenologia morfologica e riproduttiva di Cystoseira zosteroides C. Agardh (Fucales, Phaeophyceae).	Boll. Acc. Gioenia. Sci. Nat.	1995	Italy	y	n	n	n
316	GIACCONE G., COLONNA P., GRAZIANO c., MANNINO A.M., TORNATOREE., CORMACI M., FURNARI G. & SCAMMACCA B	Revisione della flora marina di Sicilia e isole minori	Boll Acc Gioenia Sci Nat, Catania	1985	Italy	y	n	n	n
317	Giaccone G.	Una nuova specie mediterranea del genere CystoseiraC. Agardh (Phaeophyta, Fucales): C. hyblaea G. Giaccone, con osservazioni critiche su alcune entita tassonomiche poco note o imperfettamente descritte	Boll Acc Gioenia Sci Nat, Catania	1985	Italy	y	n	n	y
318	Giaccone, G., M.Sortino, A.Solazzi and C.Tolomio.	Tipologia e distribuzione estiva della vegetazione sommersa dell'isola di Pantelleria	Lav. Reale Ist. Bot. Reale Giard. Colon., Palermo	1973	Italy	y	n	n	n
319	Scammacca, B., G. Giaccone, F. Pizzuto and G. Alongi.	La vegetazione marina di substrato duro dell'isola di Lampedusa (Isole Pelagie).	Boli.Ace.GioeniaSe.Nat.Catania	1993	Italy	y	n	n	n
320	BRAMBATI A., GHIRARDELLl E.,	Bionomia del Canale di S. Pietro (Sardegna): ricerche sedimentologiche, idrologiche e rilievo	NovaThalassia	1980	Italy	y	n	n	n

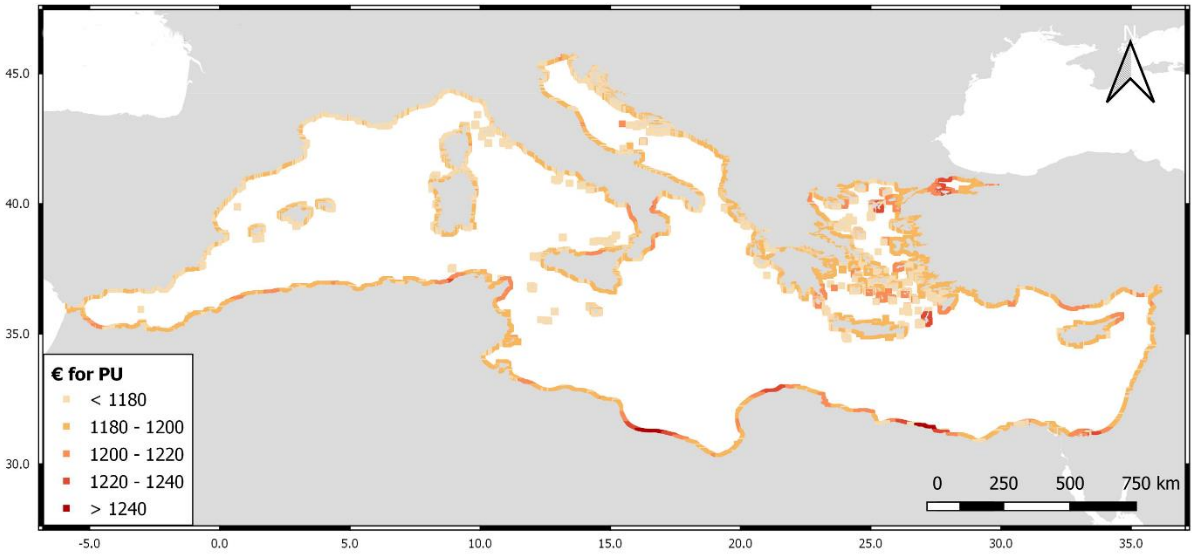
	GIACCONE G., OREL G. & VIO E. 1980.	aerofotogrammetico in funzione della tipologia e della distribuzione delle comunità bentoniche							
321	Buen, O. de	Notas de un viaje a Menorca.	Bol. Soc. Esp. Hist. Nat.	1906	Spain	y	n	n	n
322	Conde, F. & Flores Moya, A.	Nuevas adiciones al conocimiento de las macroalgas marinas de la isla de Alboran (Mediterraneo Occidental).	Acta Botanica Malacitana	2000	Spain	n	n	n	n
323	Conde Poyales, F. & Seoane-Camba, J.A.	Corología de las especies de algas en relación a ciertos factores ecológicos en el litoral Malagueño.	Collectanea Botanica	1982	Spain	y	n	n	n
324	Conde Poyales, F.	Sobre la colección de algas del herbario de la Sociedad Malagueña De Ciencias (S. XIX).	Acta Botanica Malacitana	1992	Spain	y	n	n	n
325	Solazzi, A.	Primi dati sulle alghe della scogliera "I travi" di Portonovo (Ancona).	Giorn. Bot. Ital.	1964	Italy	y	n	y	n
326	Solazzi, A.	Primi dati sulle alghe macroscopiche bentoniche della costa neretina	Giorn. Bot. Ital.	1967	Italy	y	n	n	n
327	Schiffner V.,	Beitrage zur Kenntnis der Meeresalgen. III. Einige algen von Riccione bei Rimini	Hedwigia	1926	Italy	y	n	n	n
328	Simonetti G.,	I consorzi a fanerogame marine nel Golfo di Trieste.	Atti Ist. Veneto Sci. Lett. Arti, CI. Sci. Mat. Nat.	1972	Italy	y	n	n	n
329	Solazzi, A.	Le alghe della costiera Marchigiana	Giorn. Bot. Ital.	1976	Italy	y	n	n	n
330	Zavodnik, N., Ivesa, L., Travizi, A.	Note on recolonisation by fucoid algae Cystoseira spp. and Fucus virsoides in the North Adriatic Sea	Acta Adriatica	2002	Croatia	y	n	n	n
331	Mangialajo L., Barberis G., Cattaneo-Vietti R.	Contributo alla conoscenza della biodiversità macroalgale delle Aree Marine Protette liguri.	Informatore Botanico Italiano	2004	Italy	y	n	n	n
332	Ceccherelli, G., Casu, D., Sechi, N.	Spatial variation of intertidal assemblages at Tavolara-Capo Coda Cavallo MPA (NE Sardinia): geographical vs. protection effect.	Mar. Environ. Res.	2005	Italy	y	n	y	n
422	Valiante, R.	Die Cystoseiren. Fauna und Flora des Golfes von Neapel, Zoologischen Station zu Neapel	Monographie.	1883	Italy	y	n	n	n
437	Acunto S., Maltagliati F., Cinelli F	Osservazioni sui popolamenti bentonici di un'area interessata da attività idrotermali nei pressi dell'Isola di Panarea (Isole Eolie)	Biol. Mar. Medit	1996	Italy	y	n	n	n
438	Alongi G., Catra M.	La flora sommersa dell'isolotto di Lampione (Isole Pelagie).	Boll. Accad. Gioenia Sci. Nat. Catania	1999	Italy	y	n	n	n
439	Alongi G., Catra M., Cormaci M.	Fenologia morfologica e riproduttiva di Cystoseira elegans Sauvageau (Cystoseiraceae, Phaeophyta).	Boll. Accad. Gioenia Sci. Nat. Ca-tania,	1999	Italy	y	n	n	n
442	Battiato A., Cormaci M., Furnari G., Scammacca B.	Osservazioni sulla zonazione dei popolamenti fitobentonici di substrato duro della Penisola della Maddalena (Siracusa).	Thalassia Salentina	1979	Italy	n	y	n	n
444	Calvo S., Drago D., Sortino M.	Distribuzione estiva del fitobentos e biomassa del-le specie di interesse merceologico dello Stagnone (Costa meridionale della Sicilia).	Thalassia Salentina	1980	Italy	y	n	n	n
445	Campisi M.R., Di Geronimo I., Furnari G., Scammacca B.	Premières observa-tions sur les Algues, les Bryozoaires et les Mollusques d'un peuplement de Cystoseira dubia Valiante a l'Ile Lachea (Sicile orientale).	Rapp. Commis. Int. Explor. Sci. Mer Médit.	1973	Italy	y	n	n	n

446	Catra M., Alongi G., Giaccone G.	La flora marina bentonica dell'isola di Filicudi (Isole Eolie).	Boll. Accad. Gioenia Sci. Nat. Catania	1999	Italy	y	n	y	n
447	Cecere E., Saracino O. D., Fanelli M., Petrocelli A.	Presence of drifting algal bed in the Mar Piccolo basin, Taranto (Ionian Sea, Southestern Italy).	J. Appl. Phycol.	1992	Italy	y	n	n	n
450	Felicini G.P.	Alghe macroscopiche del mare d'Otranto.	Webbia	1965	Italy	n	y	n	n
452	Furnari G., Scammacca B.	Flora algale dell'Isola Lachea (Golfo di Catania). Primo contributo.	Giorn. Bot. Ital.	1970	Italy	y	n	n	n
455	Giaccone G.	Associazioni algali e fenomeni secondari di vulcanesimo nelle acque marine di Vulcano (Mar Tirreno).	Giorn. Bot. Ital.	1969	Italy	y	n	n	n
458	Lenzi M.	Le récif-barrière de Posidonia oceanica (L.) Delile de Santa Liberata (Toscane, Italie): cartographie et biometrie.	Giorn. Bot. Ital.	1987	Italy	y	n	n	n
459	Papi I., Pardi G., Lenzini S., Benedetti Cecchi L., Cinelli F.	Benthic marine flora in the Tuscan Archipelago. A first contribution: Isles of Capraia, Elba, Formiche di Grosseto, Giglio, Scoglio d'Africa, Montecristo and Giannutri.	Giorn. Bot. Ital.	1992	Italy	y	n	n	n
460	Pastore M.	Osservazioni preliminari sull'infralitorale di substrato roccioso lungo la costa salentina (Golfo di Taranto).	Thalassia Salentina,	1981	Italy	y	n	n	n
461	Pizzuto F., Presti C., Serio D.	Struttura e periodismo di un popolamento a Cystoseira amentacea v. stricta Montagne (Fucales, Fucophyceae) del litorale catanese.	Boll. Accad. Gioenia Sci. Nat. Catania,	1995	Italy	y	n	n	y
462	Pizzuto F., Serio D., Furnari G.	First record of tetrasporophytes of Polisi-phonia funebris De Notaris ex J. Agardh (Rhodomelaceae, Rhodophyta) from the Meditterranea Sea, with taxonomic considerations.	Giorn. Bot. Ital.	1996	Italy	y	n	n	n
463	Sartoni G., Boddi S.	Morphological observations on some fleshy crustose algae in the Island of Gorgona (Tuscan Archipelago).	Giorn. Bot. Ital.	1992	Italy	y	n	n	n
464	Tita G.	Aspects écologiques d'un peuplement à Cystoseira dubia Valiante dans le circalit-toral de Catane (Sicilie orientale, Italie).	Mar. Life,	1994	Italy	y	n	n	n
465	Tolomio C.	Fitoplancton e fitobenthos lungo le coste Calabro-campane (Mar Tirreno). Primo contributo.	Giorn. Bot. Ital.,	1973	Italy	y	n	n	n
531	Spinelli V.	Le alghe marine della Sicilia orientale.	Atti Accad. Gioenia Sci. Nat. Catania,	1905	Italy	y	y	n	n
533	Tolomio, C.	Osservazioni sull'ambiente e tipologia estiva del fitoplancton e fitobentos lungo il litorale di Tropea (Calabria). Secondo contributo	Giorn. Bot. Ital.,	1976	Italy	y	n	n	n
543	Giaccone	associazioni algali e fenomeni secondari di vulcanismo delle acque marine di vulcano (Mar Tirreno)	Giorn. Bot. Ital.	1969	Italy	y	y	n	y
550	Giaccone G., Letizia A.	aspetto estivo della vegetazione marina di capo d'orlando	Naturalista sicil.	1984	Italy	y	n	n	y
551	Cavaliere A.	Osservazioni nello Stagnone di Marsala (Sicilia)	Delpinoa	1961	Italy	y	y	n	n
552	Cavaliere A.	Ricerca sulla flora algologica dello Stretto di Messina		1957	Italy	n	y	n	n
554	Gerloff J. & Nizamuddin M.	New species of the genus Cystoseira C. Ag.		1976	Italy	y	n	n	n

555	Cattaneo-Vietti, R., Albertelli, G., Bavestrello, G., Bianchi, C.N., Cerrano, C., Chiantore, M., Gaggero, L., Morri, C., Schiapparelli, S.	Can rock composition affect sublittoral rock Epibenthic Communities?	Marine Ecology	2002	Italy	y	n	n	n
556	Balduzzi, A., Bianchi, C.N., Cattaneo-Vietti, R., Cerrano, C., Cocito, S., Cotta, S., Degl'Innocenti, F., Diviacco, G., Morgigni, M., Morri, C., Pansini, M., Salvatori, L., Senes, L., Sgorbini, S., Tunesi, L.	Primi Lineamenti di Bionomia Bentica dell'Isola di Gallinara (Mar Ligure)	Atti del 10° Congresso A.I.O.L.	1992	Italy	y	n	n	y
617	Giaccone, G. & Sortino, M.	Flora e vegetazione algale di Isola delle Femmine	Lav. Ist. Bot. Giard. Colon. Palermo	1964	Italy	y	n	n	n
647	Cinelli, F., Drago, D., Furnari, G., Giaccone, G., Scammacca, B., Solazzi, A., Sortino, M. & Tolomio, C.	Flora marina dell'Isola di Linosa (Arcipelago delle Pelagie). The sea flora of Linosa Island (Pelagie Islands).	Mem. Biol. Mar. Ocean	1976	Italy	y	n	n	n
651	Grech, D., Fallati, L., Farina, S., Guala, I.	The matrix reloaded: CARLIT assessment ten years later in the Sinis coast (Sardinia, Italy) coupled with drone technology. In: UNEP/MAP – SPA/RAC, 2019.	Proceedings of the 6th Mediterranean Symposium on Marine Vegetation	2019	Italy	y	n	n	n
652	Grech, D.	Cystoseira usneoides: a new protected species for the Penisola del Sinis - Isola di Mal di Ventre MPA and Sardinia	Biologia Marina Mediterranea	2019	Italy	y	n	n	n
653	Grech, D., Fallati, L., Farina S., Cabana, D. & Guala, I.	Marine Forests (Fucales, Ochrophyta) in a low impacted Mediterranean coastal area: current knowledge and future perspectives.	Planning, nature and ecosystem services	2019	Italy	y	n	n	n

Supplementary Table S4.2 Other data sources: expert personal data.

ID	Data Providers and othe source of data	Country
139	Simonetta Fraschetti	Albania-Croatia-France-Greece-Italy
140	Luisa Mangialajo	France-Italy-Montenegro
141	Gianluca Sarà-Chiara Giommi.Anna Maria Mannino	Italy
142	Giulia Ceccherelli	Italy
143	Luigi Piazzi	Italy
144	Alan Deidun	Malta
145	Vesna Macic	Montenegro
146	Mohamed Ramdani	Morocco
147	Emma Cebrian	Spain
148	Kike Ballesteros	France-Greece-Italy-Spain-Tunisia-Turkey
149	Jamilia Ben Souissi	Tunisia
150	Ergun Taskin	Turkey
151	Gil Rilov	Israel
152	EMODnet	Spain
153	Annalisa Falace	Italy
164	Ina Nasto	Albania
165	Sotiris Orfanidis	Greece
166	Fabio Rindi	Italy
167	Sabour Brahim	Morocco
650	Daniele Grech	Italy



Supplementary Figure S4.1 Map of the distribution of costs across PUs. Costs are expressed in €/25m2 for each PU.

Appendix b: Activity Report

Ph.D. student: Erika Fabbrizzi

Cycle XXXV

Born in: Rome

On 01/04/1993

With fellowship UNINA and SZN

Tutor (UNINA): Simonetta Fraschetti

External tutor (if any): Luigi Musco, Laura Tamburello (Stazione Zoologica Anton Dohrn)

Institution where the research is conducted:

Biology Department and Ischia Marine Centre (SZN)

Thesis provisional title: Marine forest restoration in a fast-changing Mediterranean Sea

Time spent abroad (mandatory)

From 01/06/2022 to 31/07/2022

Institution:	CEAB – CSIC (Centro de Estudios Avanzados de Blanes)
Supervisor:	Emma Cebrian
Country:	Spain

Title of the project: thermotolerance experiment on different populations of *Gongolaria barbata*

Scientific publications:

1. Fabbrizzi E, Scardi M, Ballesteros E, Benedetti-Cecchi L, Cebrian E, Ceccherelli G, De Leo F, Deidun A, Guarnieri G, Falace A, Fraissinet S, Giommi C, Macic V, Mangialajo L, Mannino AM, Piazzi L, Ramdani M, Rilov G, Rindi L, Rizzo L, Sarà G, Souissi JB, Taskin E and Fraschetti S (2020). Modeling Macroalgal Forest Distribution at Mediterranean Scale: Present Status, Drivers of Changes and Insights for Conservation and Management. *Front. Mar. Sci.* 7:20. doi: 10.3389/fmars.2020.00020
2. Rindi F, Cebrian E, Orfanidis S, Bianchelli S, Belattmania Z, Fabbrizzi E, Hannachi A, Mangialajo L, Nasto I, Sabour B, Tamburello L, Verdura J, Verges A, Danovaro R and Fraschetti S (2021). FUCALEAN ALGAL FORESTS IN THE MEDITERRANEAN SEA: CURRENT STATUS AND CHALLENGES FOR RESTORATION. *Phycologia* 60, 31-31.
3. Orfanidis S, Rindi F, Cebrian E, Fraschetti S, Nasto I, Taskin E, Bianchelli S, Papathanasiou V, Kosmidou M, Caragnano A, Tsioli S, Ratti, S, Fabbrizzi E, Verdura J, Tamburello L, Beqiraj S, Kashta L, Sota D, Papadimitrou A, Mahmoudi E, Kiçaj H, Georgiadis K, Hannachi A and Danovaro R (2021). Effects of Natural and Anthropogenic Stressors on Fucalean Brown Seaweeds Across Different Spatial Scales in the Mediterranean Sea. *Front. Mar. Sci.* 8:658417. doi: 10.3389/fmars.2021.658417
4. Tamburello L, Chiarore A, Fabbrizzi E, Colletti A, Franzitta G, Grech D, Rindi F, Rizzo L, Savinelli B and Fraschetti S (2021). Can we preserve and restore overlooked macroalgal forests? *STOTEN* 150855. doi: 10.1016/j.scitotenv.2021.150855

5. Fraschetti S, Fabbrizzi E, Tamburello L, Uyarra MC, Micheli F, Sala E, Pipitone C, Badalamenti F, Bevilacqua S, Boada J, Cebrian E, Ceccherelli G, Chiantore M, D'Anna G, Di Franco A, Farina S, Giakoumi S, Gissi E, Guala I, Guidetti P, Katsanevakis S, Manea E, Montefalcone M, Sini M, Asnaghi V, Calò A, Di Lorenzo M, Garrabou J, Musco L, Oprandi A, Rilov G, Borja A, 2022. An integrated assessment of the Good Environmental Status of Mediterranean Marine Protected Areas. J. Enviro. Manage. doi :10.1016/j.jenvman.2021.114370
6. Fabbrizzi E, Giakoumi S, De Leo F, Tamburello L, Chiarore A, Colletti, A, Coppola M, Munari M, Musco L Rindi F, Rizzo L, Savinelli B, Franzitta G, Grech D, Cebrian E, Verdura J, Bianchelli S, Mangialajo L, Nasto I, Sota D, Orfanidis S, Papadopoulou NK, Danovaro R, Fraschetti S., 2023. The challenge of setting restoration targets for macroalgal forests under climate changes. J. Enviro Manage. 326, 116834. doi: 10.1016/j.jenvman.2022.116834

Meeting, workshops and conferences

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Date	City/Country	Meeting	Title	PO/OC*	Autori
2-3/12/19	San Sebastiàn, Spain	MARCONS WORKSHOP	Using NEAT to test MPAs GES		
15/10/20	Virtual Conference	MARCONS FINAL CONFERENCE	ADVANCING MARINE CONSERVATION IN THE EUROPEAN AND CONTIGUOUS SEAS		
13-15/4/21	Virtual meeting	AIOL-SItE. Incontro dei dottorandi e giovani ricercatori in Ecologia e Scienze dei Sitemi Acquatici	Spatial Prioritization for fucalean brown algae forests restoration in the Mediterranean Sea	OC	E. Fabbrizzi, F. De Leo, L. Tamburello, M. Coppola, A. Chiarore, A. Colletti, M. Munari, L. Musco, F. Rindi, L. Rizzo, B. Savinelli, G. Franzitta, D. Grech, S. Giakoumi, E. Cebrian, J. Verdura, S. Bianchelli, L. Mangilajo, I. Nasto, S. Orfanidis, N. Papadopoulou, H. Thornton, R. Danovaro, S. Fraschetti
21-24/6/21	Virtual Conference	SER Conference. 9th World Conference on Ecological Restoration. A New Global Trajectory	Spatial Prioritization for fucalean brown algae forests restoration in the Mediterranean Sea	OC	E. Fabbrizzi, F. De Leo, L. Tamburello, M. Coppola, A. Chiarore, A. Colletti, M. Munari, L. Musco, F. Rindi, L. Rizzo, B. Savinelli, G. Franzitta, D. Grech, S. Giakoumi, E. Cebrian, J. Verdura, S. Bianchelli, L. Mangialajo, I. Nasto, S. Orfanidis, K. N. Papadopoulou, H. Thornton, R. Danovaro, S. Fraschetti
25-27/10/21	Virtual Congress	XXX Congresso S.It.E – Ecology for an ecological transition	Spatial Prioritization for fucalean brown algal forests restoration in the Mediterranean Sea	OC	E. Fabbrizzi, F. De Leo, L. Tamburello, M. Coppola, A. Chiarore, A. Colletti, M. Munari, L. Musco, F. Rindi, L. Rizzo, B. Savinelli, G. Franzitta, D. Grech, S. Giakoumi, E. Cebrian, J. Verdura, S. Bianchelli, L. Mangialajo, I. Nasto, D. Sota, S. Orfanidis, K. N. Papadopoulou, H. Thornton, R. Danovaro, S. Fraschetti
27/06/22-01/07/22	San Michele all’Adige, Italy	XXVI Congresso AIOL – Esperienze e approcci innovativi per la	The challenge of setting restoration targets across the Mediterranean Sea	OC	E. Fabbrizzi, S. Giakoumi, F. De Leo, L. Tamburello, A. Chiarore, A. Colletti, M. Coppola, M.

		salvaguardia degli ecosistemi acquatici	under climate changes: the case study of macroalgal forests		Munari, L. Musco, F. Rindi, L. Rizzo, B. Savinelli, G. Franzitta, D. Grech, E. Cebrian, J. Verdura, S. Bianchelli, L. Mangialajo, I. Nasto, D. Sota, S. Orfanidis, N.K. Papadopoulou, R. Danovaro, S. Fraschetti
05/09/22- 09/09/22	Alicante, Spain	13° SERE Conference- Restoring Nature, Reconnecting People	Setting restoration targets under climate changes: the case study of Mediterranean macroalgal forests	OC	E. Fabbrizzi, S. Giakoumi, F. De Leo, L. Tamburello, A. Chiarore, A. Colletti, M. Coppola, M. Munari, L. Musco, F. Rindi, L. Rizzo, B. Savinelli, G. Franzitta, D. Grech, E. Cebrian, J. Verdura, S. Bianchelli, L. Mangialajo, I. Nasto, D. Sota, S. Orfanidis, N.K. Papadopoulou, R. Danovaro, S. Fraschetti
13/09/22- 15/09/22	Siena, Italy	XXXI Congresso S.It.E – Adattamenti degli Ecosistemi alle Pressioni dell'Antropocene	Testing physiological tolerance to marine heatwaves in the canopy-forming macroalgae *Gongolaria barbata*	PO	E. Fabbrizzi, M. Munari, L Tamburello, C. Arena, A. Cannavacciuolo, A. Chiarore, A. Colletti, G. Costanzo, A. S. Fajardo, M. Nannini, B. Savinelli, C. Silvestrini, E. Vitale and S. Fraschetti

Courses

First year

From	To	Course title	Primary teacher	n. of lessons	CFU
12/02/20	14/02/20	Open Day	Prof. Fulgione	3 (10 hours)	1.25
24/02/20	28/02/20	Qualitative Mathematical Modelling for Socio-Ecological Systems	Jeffrey Dambacher	5 (35 hours)	4.3
02/04/20	07/05/20	Gene expression in model systems	Prof.ssa Caterina Missero	7	2
20/04/20	06/05/20	System Biology	Prof. Giovanni Scala	6	2
12/05/20	12/05/20	Physiology	Prof. L. Cigliano, Prof. Caterina Missero	2	1
13/10/20	19/10/20	Experimental design and statistical inference	Dr. Tomàs Vega Fernàndez (SZN)	5 (15 hours)	1.87

Second Year

From	To	Course title	Primary teacher	n. of lessons	CFU
12/01/21	16/02/21	Programming and Algorithmic thinking	Prof. G. Scala	10	1.25
24/03/21	30/03/21	Fundamentals of synthetic biology	Prof.ssa Velia Siciliano	3 (6 hours)	1
15/10/21	18/10/21	Presentation Skills and CV writing	Prof. Lele Castello	2 (4 hours)	0.5

Third Year

From	To	Course title	Primary teacher	n. of lessons	CFU
21/03/22	23/03/22	AMAre PLUS GIS training course	Prof. Federica Foglini	3	2.25
28/04/22	26/05/22	Ecosystem-based management and Marine Spatial Planning in the Mediterranean Sea	Prof. Dania Abdul Malak	4	0.25

English course:

From	To	Course title	Primary teacher	n. of lessons	CFU
20/10/20	26/05/21	Cambridge CAE	Alessio Mirarchi	50 (100 hours)	12.5

Scientific seminars:

Date	Institution	Title	Speaker	Time
07/04/20	Aeolian Islands Preservation Fund	Perchè proteggere gli ambienti estremi delle Eolie? L'attuale ricerca sul COVID-19 e gli organismi estremofili	Prof. Donato Giovannelli	15:00-15:30
20/04/20	Aeolian Islands Preservation Fund	Le praterie di Posidonia dell'isola di Salina	Prof. Gabriele Procaccini	15:00-16:00
29/04/20	Dipartimento di Biologia – Università Federico II	"Evoluzione" Un viaggio lungo 4 miliardi di anni	Prof. Roberto Ligrone	10:00-12:30
29/04/20	Scuola di Agraria e Medicina Veterinaria – Università Federico II	COVID-19: una visione One-Health	Prof. Giuseppe Cringoli, Prof. Matteo Lorito, Prof. Gaetano Oliva	16:30-18:30
16/09/20	FISV	Virtual FISV Symposium on SARS-CoV-2 Biology and COVID-19: Current research and perspectives	Gennaro Ciliberto	8:30-17:30
03/10/20	Associazioni culturali MaDre, NSeaYet	Il Capitale Naturale		9:00-13:30
13/11/20	Dipartimento di Biologia – Università Federico II, Università di Pisa	Ecological experiments in the Anthropocene	Prof. L. Benedetti-Cecchi	10:00-12:00
17/11/20	Dipartimento di Biologia – Università Federico II, Università di Trieste	Subtle and disregarded human impacts on the functioning of marine systems	Prof. Antonio Terlizzi	14:00-16:00
23/11/20	Università degli Studi della Campania "Luigi Vanvitelli"	Fires and soil resources	Prof. F. Rutigliano	10:00-12:00
24/11/20	Università degli Studi di Salerno	Underground environments in an adaptative ecosystem management framework	Prof. D. Baldantoni	11:00-13:00
26/11/20	Università Federico II, University IUAV of Venice	A review of the combined effects of climate change and other human stressors on the marine environment	Dr. E. Gissi	14:00-16:00
27/11/20	Dipartimento di Biologia – Università Federico II	Experiments in ecology	Prof. G. Maisto	14:00-16:00
30/11/20	Università Federico II, Università del Salento	Experiments in ecology	Dr. G. Guarnieri	9:00-12:00

01/12/20	Università Federico II, Stazione Zoologica Anton Dohrn	Restoration ecology: from theory to costs	Dr. L. Musco	9:30-11:00
03/12/20	Università Federico II, Stazione Zoologica Anton Dohrn	The contribute of experimental ecology to the main concepts in ecology	Dr. L. Tamburello	14:00-16:00
21/12/20	Dipartimento di Biologia – Università Federico II	Good science and bad communication: design principles for scientific visualization	Prof. D. Giovannelli	10:00-12:00
22/01/21	SItE	Ecology is everywhere! La successione ecologica dei biodeteriogeni delle pergamene antiche	Prof. L. Migliore	15:00-16:00
23/04/2021	SItE	Ecologia quantitativa: dai dati ai modelli	Prof. M. Gatto	15:00-16:00
17/12/2021	SItE	Cambiamenti Climatici: la risposta della Natura e gli effetti sulla Società Umana	Prof. A. Fano	15:00-16:00
18/02/2022	SItE	Ecological Restoration: challenges and perspectives	Prof. S. Fraschetti	15:00-16:00
22/02/2022	Dipartimento di Biologia – Università Federico II	Multi-omics approaches and modeling of disease pathogenesis	Prof. C. Missero	16:00-17:30
01/03/2022	Dipartimento di Biologia – Università Federico II	Citizen-science, community engagement e biotecnologie entomologiche per la lotta ecosostenibile alla zanzara tigre asiatica Aedes albopictus	Prof. M. Salvemini	16:00-17:30
18/02/2022	SItE	Ecologia delle malattie e COVID-19: una panoramica su dati, meccanismi e modelli	Prof. M. Gatto	16:00-17:30