Università
Ca'Foscari
Venezia

PhD in Science and
Management of Climate Change

34° Cycle (PhD defence: 2023)

Research dissertation

Analysis of marine ecosystem functioning and services in a changing climate

SSD: BIO/07, GEO/12, SPS/07

PhD coordinator
Prof. Enrica De Cian

Supervisor

Prof. Fabio Pranovi

Co-Supervisor

Dr. Fabien Moullec

PhD candidate

Laura Basconi
ID **956409**

A mia nonna Lena, esempio di

eleganza d'animo,

amore incondizionato e

costanza

It's not what happens to you, but how you react to it that matters

Epictetus

TABLE OF CONTENTS

SUMMARY

English

The present doctoral thesis aims to create knowledge around two main caveats (i) understanding climate change-related impacts on marine ecosystem functioning and, (ii) unrevealing the spatial relationship between ecosystem functioning and ecosystem services. The doctoral thesis is constituted of three peer-reviewed scientific papers. The first seeks a better understanding of the impact of climate change on trophodynamics (i.e., speed and proportion of biomass flowing in a food web) as a proxy of ecosystem functioning in the Mediterranean Sea. Results projected a general Mediterranean Sea reduction in ecosystem functioning with some areas that shows a resilient response to climate change. The second work reports a multiple coastal and marine ecosystem services assessment using as the study area the Northern-Central Adriatic Sea. It resulted in the mismanagement of ecosystem services across the area which should be corrected to maintain the long-term supply of coastal and marine ecosystem services for the benefit of all. The third part puts the other two into perspective, using the Adriatic Sea as study-case, to explore whether it is possible to infer the impact of climate change on ecosystem functioning to services, studying the spatial relationships among these two facets of ecology. Results show no spatial coherence among areas of high ecosystem services supply and high values of ecosystem functioning indicators, supporting that the linkage between ecosystem functioning and service could be highly non-linear and therefore push for further studies. The joint analysis of ecosystem functioning, and services often treated as two different branches of science is of paramount importance in the view of marine Ecosystem-based Management (EbA) and sustainability.

Italian

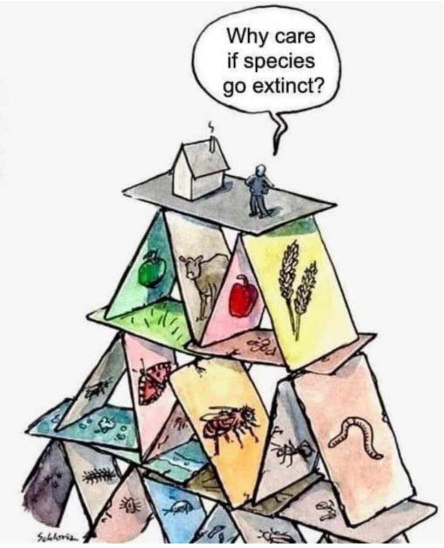
La presente tesi di dottorato mira a rispondere a due pressanti questioni in ecologia: l’impatto del cambiamento climatico sul funzionamento ecosistemico marino e la relazione spaziale tra funzionamento e servizi ecosistemici. La tesi dottorale è per tanto costituita da tre articoli scientifici. Il primo ha l’obiettivo di comprendere e misurare l’impatto del cambiamento climatico, a scala Mediterranea. In questo lavoro sono stati impiegati indicatori di dinamica trofica (misurazione di velocità e proporzione del flusso di biomassa lungo la catena alimentare marina). I risultati constano di una generale riduzione del funzionamento ecosistemico marino a causa del cambiamento climatico, mostrando tuttavia aree di resilienza dove gli indicatori, a seguito del cambiamento climatico, tendono addirittura ad aumentare rispetto alla situazione corrente. Il secondo lavoro è una misurazione multipla di sette servizi ecosistemici marini e costieri nel Mar Adriatico. I risultati sottolineano una malagestione dei servizi ecosistemici che deve essere ampiamente riveduta per garantire il loro mantenimento e utilizzo nel futuro. Il terzo lavoro unisce i due precedenti cercando di trovare un legame spaziale tra indicatori di funzionamento e quelli dei servizi ecosistemici, con l’obiettivo di inferire l’impatto del cambiamento climatico misurato sul funzionamento ai servizi. I risultati mostrano un’incoerenza spaziale tra le aree caratterizzate da alto funzionamento ecosistemico e alta generazione di servizi. Per questo motivo risulta vitale approcciare alla gestione delle risorse marine con una visione olistica che tenga conto sia delle mere caratteristiche ecologiche (funzionamento) e delle richieste socioeconomiche dello stesso.

French

L’objectif visé par cette thèse doctorale est de répondre à deux questions pressantes en écologie: l'impact du changement climatique sur le fonctionnement des écosystèmes marins et la relation spatiale entre le fonctionnement et les services des écosystèmes. La thèse de doctorat se compose donc de trois articles scientifiques. (I) Le premier a pour objectif de comprendre et de mesurer l'impact du changement climatique à l'échelle méditerranéenne. Dans ce travail, des indicateurs de dynamique trophique (c'est-à-dire la mesure du taux et de la proportion du flux de la biomasse à travers le réseau trophique marin) ont été utilisés. Les résultats consistent en une réduction générale du fonctionnement des écosystèmes marins due au changement climatique, montrant cependant des zones de résilience où les indicateurs, à la suite du changement climatique, ont même tendance à augmenter. (II) Le deuxième travail est une mesure multiple de sept services écosystémiques marins et côtiers dans la mer Adriatique. Les résultats soulignent une mauvaise gestion des services écosystémiques qui doivent être revus en profondeur pour assurer leur maintenance et leur utilisation à l'avenir. (III) Le troisième travail combine les deux précédents en essayant de trouver un lien spatial entre les indicateurs de fonctionnement et ceux des services écosystémiques, dans le but d'inférer l'impact du changement climatique mesuré sur le fonctionnement des services. Les résultats montrent une incohérence spatiale entre les zones caractérisées par un haut fonctionnement écosystémique et une forte génération de services. Pour cette raison, il est essentiel d'aborder de façon holistique la gestion des ressources marines, et de prendre en compte, à la fois, les simples caractéristiques écologiques (fonctionnement) et les exigences socio-économiques de celles-ci.

Popular Science

Ecosystems worldwide are subjected to unprecedented loss of biodiversity, largely resulting from a range of human impacts such as over-exploitation, habitat loss, and climate change. However, as humans, we depend on ecosystems, and this is quantified through the currently widely used metrics of ecosystem services. To understand the consequences of the ecological crisis humanity is facing on both land and Oceans, the scientists are trying to understand and quantify the relationship between ecosystem functioning (i.e., the ability of an ecosystem to perform and maintain a suite of key properties such as flux of energy and biomass trough the food chain) and ecosystem services (i.e., benefits that Nature provides to humans), ultimately trying to answer: how human-induced changes are affecting ecosystem functioning? How these impacts on ecosystem functions relate to ecosystem services?



“Nature and other species are the life support system for human lives”

An always positive relationship between ecosystem functioning and ecosystem services supply has been theoretically stated by different scientific publications. However, empirical evidence to demonstrate this is scattered. Practically, humans can carry on their activities, at least in the short term, taking advantage of ecosystem services even when the functioning of the natural system is low because of strong socio-economic drivers for that particular ESs delivery with consequent detachment to the ecological processes. Furthermore, in certain ecological conditions, the functioning could be high but the service low because of no alignment between ecological and socioeconomic needs. For instance, when considering the tropicalization of the Mediterranean Sea, species will move northward affected by increasing water temperature and tropical species from the Red Sea will colonize the southern Mediterranean Sea (as it is already occurring at the current time). The ecological community always adapts to environmental condition changes (i.e., ecosystem resilience) however is fishery (i.e., a provisioning ecosystem service) ready for tropical fishes? Is the marker ready for fishes different from the endemic ones? Another instance comes from a survey that was carried out among tourists and recreational boaters along the Adriatic coasts. Some tourists exchanged surrogates of Nature (e.g., urban parks, agricultural fields, swimming pools) as Nature itself leading to high quantifications of ecosystem services in the areas, even when the presence of nature which should support them (i.e., functioning of the ecosystem) was low. To address these knowledge gaps, the present doctoral thesis shows how climate change, as the widest human-induced impact, will impact ecosystem functioning by the end of the 21st century using the Mediterranean Sea as a study case, as both a hotspot for biodiversity and climate change. Successively, a complete assessment of coastal and marine ecosystem services in the Adriatic Sea is presented. In the same study area, the present thesis explores the spatial relationship between multiple indicators of ecosystem functioning and ecosystem services trying to understand if the linear positive relationship, often depicted in the theoretical approach, between ecosystem functioning and services holds.

Results demonstrated that climate change will impact ecosystem functioning in the Mediterranean Sea generally lowering it by the decade 2091-2100. However, resilient areas to climate change (i.e., climate refugia) are shown. The theoretically described positive relationship between the several ecosystem functions and services indicators does not emerge from empirical evidence. Eventually, the present thesis is an attempt to warn against the isolated use of ecosystem services as the only ecologically sound metrics in sustainable approaches. Enhancing ecosystem functioning conservation, especially in climate change scenarios, is highly supported by the results of the present thesis.

List of Figures

Figure 26. The flow of ES tourism as the number of tourist arrivals per km 2 year in the coastal municipalities or counties facing the Northern-Central Adriatic Sea.

Figure 27. The normalized capacity of the ES recreational boating corresponds to the spatial explicit indicators for the attractiveness potential of the Northern-Central Adriatic system (1x1Km resolution).

Figure 28. The flow of ES boating is reported as daily trips by leisure boaters of Italy and Croatia expressed as the number of boat passages per year per km 2.

Figure 29. Normalized coastal erosion prevention capacities (a) and coastal erosion prevention flow (b) per the whole set of 10km coastal segments.

Figure 30. Normalized carbon sequestration potential as tons of carbon dioxide (CO2) sequestered by the selected habitat per km 2 per year (1x1Km resolution).

Figure 31. Division of territorial waters of the four regions: Friuli-Venezia Giulia, Veneto, Emilia-Romagna and Marche. They have been used as boxes to run the eco-physiological model for the evaluation of the ES mussel aquaculture capacity.

Figure 32. Normalized mussel aquaculture capacity as a result of the RAC bio-energetic box model based on sea surface temperature and chlorophyll A data.

Figure 33. Normalized whitefish aquaculture capacity as the suitability of whitefish farming based on sea surface temperature, Significant Wave Height (SWH), distance to the harbor, current sea uses, and cumulative impacts data (Porporato et al., 2020).

Figure 34. Normalized mussel production in tons per km2 per year.

Figure 35. Normalized whitefish production in tons per km2 per year.

Figure 36. Normalized spatially explicit annual average of industrial fishing effort by trawling (2018).

Figure 37. Normalized (0-1) ES capacity and flow for each of the seven ESs considered in the Northern-Central Adriatic multiple assessments.

List of Tables

Table 13. Extracted value per municipality (IT) and county (HR) per each ES capacity with the included unit of measurement. The mussel and whitefish aquaculture capacities have been assessed just for the Italian side.

Table 14. Extracted value per municipality (IT) and county (HR) per each ES flow with the included unit of measurement.

List of Abbreviations

AR6: 6th Assessment Report of the IPCC

BRT: Biomass Residency Time

CBD: Convention of Biological Diversity

CICES: Common International Classification of Ecosystem Services

CMEMS: Copernicus Marine Environmental Monitoring Service

CMIP6: Coupled Model Intercomparison Project 6th phase

CSMs: Climate Suitability Models

EbA or EbM: Ecosystem-based Approach or Ecosystem-based Management

ECI: Efficency Cumulated Indicator

EF: Ecosystem functioning

ESs: Ecosystem Services

ESMs: Earth System Models

FishMIP: Model Intercomparison Project

GCM: General Circulation Models

GHG: Greenhouse Gas

GLORYS: Global Ocean reanalysis and Simulation

GSA: (Mediterranean Sea) Geographic Subareas

ICES: The International Council for the Exploration of the Sea

IPBES: Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

IPCC: Intergovernmental Panel on Climate Change

MSFD: Marine Strategy Framework Directive

MSP: Maritime Spatial Planning

POLCOM-ERSEM: Proudman Oceanographic Laboratory Coastal Ocean Modelling

RCPs: Representative Concentration Pathways

SROCC: The Ocean and Cryosphere in a Changing Climate (IPCC Special Report, 2019)

SSPs: Shared Socio-economic Pathways

STECF: Scientific, Technical and Economic Committee for Fisheries (European Commission)

TEEB: Economics of Ecosystem and Biodiversity

ZJ/yr: ZeptoJoule per year (measurement unit for Ocean warming)

CHAPTER I

INTRODUCTION

1.1. Ecosystem functioning

1.1.1 Definition

Ecosystems are defined as the species, processes, and abiotic variables which are present in a defined space and time. The biotic components include the ecological characteristics (i.e., species and processes) whereas the abiotic components are the physicochemical or environmental characteristics. Studing Nature at the ecosystem scale entails the analysis of: the ecological community (consisting of populations of different species), their interactions, the particular habitat they live in, and the interactions between species and habitat. Species assemblages or ecological communities in an ecosystem can be seen as complex networks, defined by nodes (species) and links or edges which are fluxes of matter and energy among the species (i.e., interspecific processes).

The list of species in a certain ecosystem is often described as species richness or biodiversity (whether considering, as in this thesis, the diversity of the species pool and, not the intraspecies genetic diversity – see Flynn et al., 2011 for an explanation of the difference between these two concepts), while fluxes of matter and energy (together with nutrient cycles, primary productivity, and decomposition) are part of the ecosystem processes or functions. The whole set of indicators of the ecosystem structures are species number, species biomass, populations abundance, functional groups, and even the spatial patterns exhibited by habitat mosaics and landscapes. The same is true for ecosystem functioning which can be measured by primary productivity, biomass flows, nutrient cycling, carbon flux, or nitrogen use. Ecosystem functioning can be measured even with species interconnections thought: (i) predator-prey interactions, (ii) mutualistic or symbiotic relations, (iii) competitive and (iv) facilitating relationships.

Eventually, in the ecosystem network processes (functions) shapes species fitness (structure) and the other way around. In other words, structure influences the function and *vice versa* as in other biological organizations. For instance, at another biological level when proteins maintain their structures they have the function while when proteins are unfolded or denatured (i.e., they do not keep their proper shape) they are no longer able to interact with other subunits to carry on their specific functions (Wu et al., 1995). Another instance from the marine environment can be found in the different algal morphologies which correlate with different levels of light and nutrient use, and resistance to herbivory (Steneck & Dethier 1994) therefore connecting the structure to the ecosystem functions.

Vice versa, ecosystem functions influence the structure such as nutrient cycling, the flux of biomass across trophic levels, and many more instances as the Gaia Hypothesis suggests (Lovelock, 1972). To better navigate these concepts, it is vital to think holistically about structure and functioning and embrace these two concepts together.

1.1.2. Different facets of ecosystem functioning

Ecosystem functions are based on stocks and relative stability of energy and materials fluxes in an ecosystem (Pacala and Kinzig 2001). To simplify, ecosystem functioning can be defined through three basic cycles of matter and energy: extraspecific cycles (biogeochemical cycles), intraspecific cycles (life cycles and histories), and interspecific cycles (food webs) (Fig. 1) (Boero 1999).

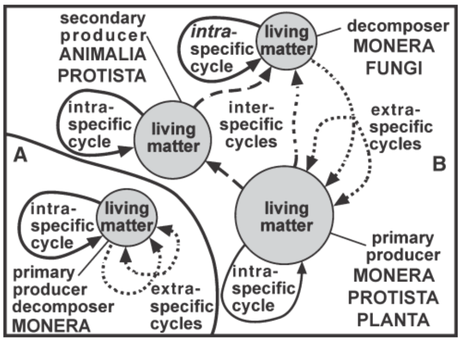


Figure 1: Type of ecosystem functioning: extraspecific cycles are biogeochemical cycles, intraspecific cycles are life cycles and histories allowing the persistence of species, and

interspecific cycles occur through food webs and are mainly, but not solely, trophic interactions ("who-eats-whom")

1.1.2.1 Extraspecific cycle: biogeochemical cycles

Biogeochemical cycles are extraspecific functions caused by interactions between the physical environment and the biological compartments of the ecosystem (Boero & Bonsdorff, 2007). One primary biogeochemical function is photosynthesis transforming inorganic resources (i.e., light and nutrients) into living biomass. On one hand, primary production constitute the basis of marine food webs. On the other hand, all individuals respire, an in doing so, lose energy which sums up metabolic losses in the ecosystem (Snelgrove et al., 2014; Strong et al., 2015). This dead organic matter namely detritus will either sink and create a flow of carbon exported to the depth of the Ocean (Oguz et al., 2001; Snelgrove et al., 2014; Kiørboe et al.,2018) or be recycled via the microbial loop and re-enter the pelagic food web (Kiørboe et al., 2018; Strong et al., 2015). In a nutshell, biogeochemistry accounts for the production and availability of the building blocks of life deriving from primary production or decomposition (Boero & Bonsdorff, 2007). Biogeochemical cycles being at the basis of life are often quantified as metrics of ecosystem function (Loreau 1998, Naeem & Wright 2003) in fact one of the currently accepted definitions of ecosystem functioning is the efficiency of biogeochemical cycles. However, looking solely at biogeochemical cycles considers mostly the abiotic side of ecosystem functioning, disregarding the intricate biological patterns and processes that make it possible, resulting in a reductive view which does not account for the complexity of communities and ecosystems. For instance, many are examples of situations in which biogeochemical fluxes have been kept constant, but the ecosystems have still decreased its functioning (e.g., alien species, overgrazing, species becoming dominant etc,). In these environmental conditions the regime shifts (Lewontine, 1969) was possibly needed to maintain the ecosystem functioning.

1.1.2.2 Intra-specific cycles: life-history and functional traits

Each species has its characteristics such as feeding habits, growth and fecundity rates, and longevity. These characteristics are called biological traits and the combinations of them form the species life history strategy. On species traits and eco-physiology depends the species responses to environmental variation. The ecological niche has been first described as the habitat in which the species lives and its accompanying behavioral adaptations (Grinnell, 1917). The definition of ecological or species niche has been completed by Hutchinson (1957) who described the ecological niche as follows, the n-dimensional space of resources available to and used by a species (Hutchinson, 1959). The ecological niche informs how species respond to climate conditions, biotic competition, and distribution of resources among other forcing factors. Hence, the species niche can be influenced by both abiotic and biotic factors of an ecosystem.

The term trait refers to the various aspects of the biology of an organism (e.g., physiology and behavior) that characterize its population responses to environmental changes or its role in ecosystem processes (Diaz & Cabido, 2001; Violle et al., 2007). One of the intraspecific functions is the average growth efficiency, quantifying the efficiency of an organism at transforming its food into growth increasing species fitness (Kerr, 1971; Andersen et al., 2009; Barker 2009). The species fitness can downstream K-strategy and r-strategy (Pianka, 1970) constituting ecosystem functioning as well. Besides, trait-based approaches comprise an arsenal of methods using biological traits to describe ecological functioning (e.g., Naeem & Wright, 2003; Bremner et al., 2006; Dolbeth et al., 2016; Pecuchet et al., 2017; Villéger et al., 2017). As different species may sometimes serve the same functions, at similar of different rates (Duffy et al., 2008), an approach where species' functioning is measured using traits could be one of the best choice, especially in areas characterized by high biodiversity.

1.1.2.3 Interspecific cycles: food web

Marine species live in community networks which are called food webs. Food webs account for the whole set of trophic interactions (“who-eats-whom”) that occur in an ecological community. The efficient circulation of matter and energy among species passes through various levels of biological organization, involving primary, secondary (and higher) production and decomposition. Many are the processes considered and therefore the indicators. The ICES community has defined a suite of 60 potential food-web indicators that embrace, the primary production required to sustain fishery, the productivity of seabirds (or charismatic megafauna), zooplankton indicators, integrated trophic indicators, biomass flows, and the biomass of trophic guilds (Tam et al., 2017). Among them what has been explored in the present thesis is the dynamics behind the energy fluxes which is called **trophodynamics**. To date, several have been the modeling efforts to digitally represent food webs to study how boundary condition changes can alter them to better informe the management of marine resources (e.g., Ecopath with Ecosim, Atlantis, OSMOSE models, and others). However, the complexity of natural systems does not allow for food web precise description. Equation-based modeling could fail to represent the huge amount of information on ‘who does what’, ‘when’, and ‘how’ in an ecosystem. Nevertheless, it is often said that modeling helps to disentangle the complexity of the modeled systems and that, step after step, the approximation of early models will lead to increasingly refined models, eventually allowing almost complete understanding - and therefore predictions - of the future conditions of the modeled systems.

To summarize, whether the aim is to account for ecosystem health it is surely insufficient to look solely at one of the metrics descrived above, because the functioning of the whole ecological system is based on the efficiency of the all three types of cycles (i.e., extra-, intra-, and interspecific cycles). In fact, the whole set of processes and patterns are essential for the functioning of ecosystems and are deeply interconnected (Bonsdorff et al., 1995) guaranteeing the continuity of ecosystem stability and resilience. A real holistic study must take into consideration all of them to better understand the state of an ecosystem (Boero & Bonsdorff, 2007). However, data availability, computer memory, and time are the limiting factors that sometimes oblige researchers to use one of the three cycles or even a sole indicator of a cycle to canvass ecosystem functioning as a whole. In particular, in the present thesis uses trophodynamics as an indicator of food web functioning. Furthermore, the intraspecific life history of species (i.e., ecological niche and biological variable) has been considered trying to complement the interspecific trophodynamic aspects (CHAPTER II *of the present thesis).*

1.2. Ecosystem services

1.2.1 Definition

Ecosystem services (ESs) are defined as the contributions of ecosystem structure and functioning – in combination with other inputs – to human well-being (Costanza et al., 1997; Berkes et al., 2000; Burkhard et al., 2012). Hence, the ES conceptual framework, and the associated concept of Nature's contributions to people (Díaz et al., 2018), lies in the value attribution of Nature for the benefits it provides to humans. In the early days, ES has been defined by environmental economists (Costanza et al., 1997, 1998) and just then embraced by ecologists and social scientists (Holmlund & Hammer 1999; Luck et al., 2003; Chee 2004 and many others). ESs have been thought to give a monetary value to common goods otherwise external to the market (i.e., externalities) and allow Nature to be easily linked to socio-economic aspects (Haines- Young and Potschin, 2010; Jones & Paramor, 2010) . Consequently, since the first definition, the ES concept has received increasing importance in the field of sustainability science and environmental management (MA 2005; de Groot et al., 2010; Seppelt et al., 2011; Burkhard et al., 2012).

1.2.2 Ecosystem service cascade: from structure to ecosystem service values

The elements that make up the link between ecosystems and human well-being are often described by means of a “cascade”. Where the biophysical structure and functioning are linked to the benefits (and values) they provide through a series of intermediate steps (Haines-Young and Potschin, 2010; Potschin and Haines-Young, 2011) (Fig. 2). The diagram makes a distinction between ecological structures and processes created or generated by living organisms and the benefits that people derive. For instance, the presence of many herbivores (i.e. biodiversity-ecosystem structure) and an efficient trophic transfer of matter and enegy (i.e. trophic transfer efficiency- ecosystem function) in the ecosystem can provide the capacity of the natural system to sustain carnivores or top predators that are the main target for fishery and are therefore eaten by humans. The ecosystem features underlying the ES fishery, in this case, can have the potential to modify the nutrient uptake of human diets. Notwithstanding that society will value this function of ecosystem differently in different places, and at different times, the foundation idea that the cascade concept highlights is that services do not exist in isolation from people’s needs. As the authors of the cascade approach suggest, the ES exists if one can identify the specific human beneficiary (Haines-Young and Potschin, 2010; Potschin and Haines-Young, 2011).

The cascade thus stresses the role of society as the beneficiary of ES, but on the other hand, it does not provide the complete picture because all the other many ESs that serves Nature itself are not considered. It considers solely the ‘final’ outputs or products from ecological systems to humans. This topic is further discussed as part of the present thesis in CHAPTER IV.

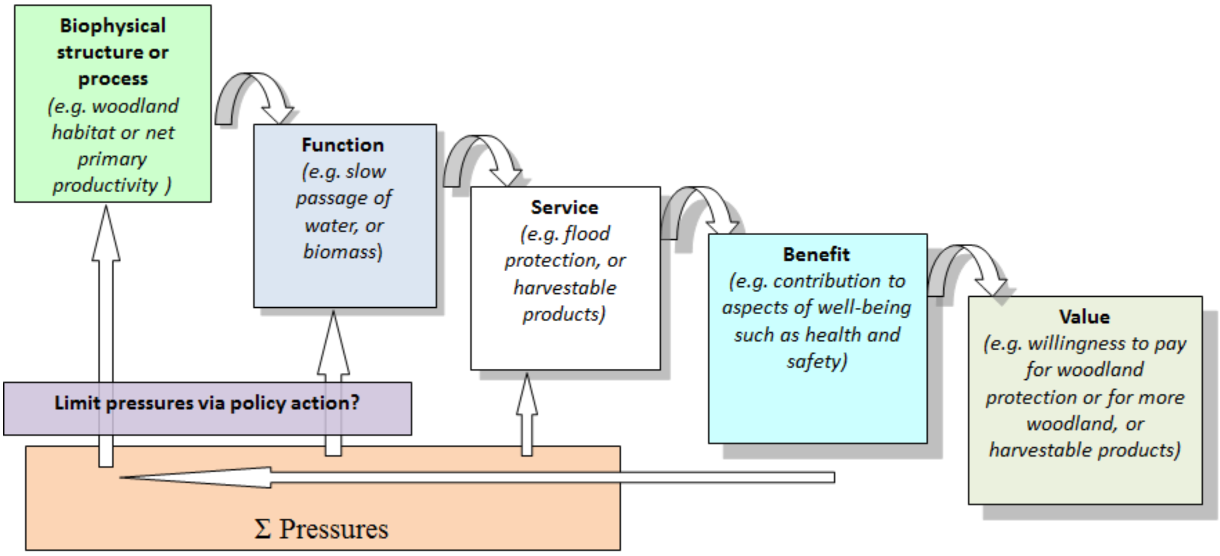


Figure 2: The ecosystem service cascade model initially proposed in Haines-Young & Potschin, 2010 modified to separate benefits and values in De Groot et al. (2010).

1.2.3 Ecosystem services assessment

Ecosystem service (ES) assessment is the mean of measurement of some or all ESs supplied by a given ecosystem. The main advantage of ES is being a transdisciplinary metric, therefore, ES assessments can be performed by using different assessment methods – biophysical, social, or economic. As reported in the previous paragraph, the intervention of some anthropogenic factors in ES delivery is a key aspect, already highlighted by several authors (Andersson et al., 2007; Bohnke-Henrichs et al., 2013; Burkhard et al., 2014; Fischer & Eastwood, 2016; Jones et al., 2016; Queiroz et al., 2015). Particularly, Fischer et al., (2009) specify that forms of capital other than natural are required to realize benefits from ESs. These “additional inputs” (*sensu* Burkhard et al., 2014) refer to the anthropogenic contributions to ES, which are recognized to be hardly separable from the ecosystem-based contributions in many human-influenced systems, and that constitutes an addiction to the complexity of ES assessments (Burkhard et al., 2014). Hence, the transdisciplinary metric of ES need to consider all the features of an ecologic socio-economic system. Notwithstanding qualitative assessments where the ES concept is used as metaphors or heuristics ideas for explaining and/or structuring a problem, **quantitative ES assessments** are the most used. They are spatial or non-spatial explicit and they can be divided into evaluation (i.e., using proper biophysical or social indicators for each ES) or valuation (i.e., using proper economical indicators such as the monetary value for each ES) assessments.

The ecosystem, due to its intrinsical multifunctionality, supplies humans with multiple ESs altogether therefore a comprehensive ES assessment considers all or a set of ESs (Klain et al., 2014; UNEP, 2014). The MA (2005) defined the ES classification system, proposing four main ecosystem categories: supporting, provisioning, cultural, and regulating services. **Supporting services** are ESs that are necessary for the production of all other ESs. Some examples include biomass production, production of atmospheric oxygen, soil formation and retention, nutrient cycling, water cycling, and provisioning of habitat. **Provisioning services** are the products, assets, or goods obtained from ecosystems, including, for example, genetic resources, food and fiber, and freshwater. **Cultural services** are the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, knowledge systems, social relations, and aesthetic values. Eventually, **regulating services** are the benefits obtained from the regulation of ecosystem processes, including, for example, the regulation of climate, water, and some human diseases. The most used framework to define ESs after the MA (2005) are: The Economics of Ecosystem and Biodiversity (TEEB, 2010), and the Common International Classification of Ecosystem Services (CICES, 2009 revised in 2013). While the former considers the same ES categories as the MA, in the latter the supporting services are treated as part of the underlying ecosystem structure and functioning that characterize the ecosystem. The explanation behind this is that 'supporting services' are only indirectly consumed by humans not aligning with the definition of ES in which a human beneficiary should always exist. The present thesis approach to the ES follows the second viewpoint (*see* CICES, 2013).

1.2.3 Ecosystem services for natural resources management

The use of ES in environmental management, especially in the context of the Ecosystem-based Approach (EbA) or Management (EbM), has the crucial advantage to provide a policy-relevant , recognized, and operation indicator of the sustainability of human use of ecosystems (McLeod & Reeve 2005; de Groot et al., 2010; Schröter et al., 2014). Being inclusive and transdisciplinary, multiple ES management faces the challenge to harmonize the provision and use of multiple ES in a way to minimize trade-offs so that the relative use of each is carried out sustainably. In particular, to convey messages of sustainability through ESs one of the main helpful definitions comes with the distinction of ES capacity and ES flow (*sensu* Villamagna et al., 2013). ES capacity is the bunches of ecosystem processes that produce benefits potentially useful to people and the flow is the actual use of the service. This distinction is a great help when deciding on the ES indicators because the capacity is tightly linked with Nature itself (i.e., ecosystem functioning) and thus the indicator could easily be biophysical , while the flow accounts more for the human uses of that natural assets or resources (i.e., socio-economy). This distinction helps even when comparing ES with EF as it has been done in CHAPTER IV to see differences in relationships among ES capacities, ES flows and ecosystem functioning metrics.

The management perspective behind the ES assessment aims to implement strategies for maintaining the benefits provided by nature in the long term. As suggested as outcomes of the Operationalisation of Ecosystem Services and Natural Capital - OpeNESS project (Dick et al., 2018; Jax et al., 2018), the complete management of ESs inland or in seascapes should follow the steps of (1) defining the problem, (2) defining the relevant ESs, (3) assessing and valuing the ESs, (4) suggesting solutions to decision-makers, (5) adopting and implementing the solution, (6) monitoring and evaluating the effect of the solution, (7) recommencing the cycle once again to assess whether any further adjustment is required. Notwithstanding, the huge support in the past decades of international bodies (notably European Commission;

IPBES; UNEP) to include ESs in decision-making at all levels, often multiple ES assessments do not arrive on the real round table of management remaining a limited scientific effort (i.e., reaching the step 4 of the above presented list). However, conceptual work could end up being nothing more than theoretical speculation if it is not operationalized. Linking conceptual work with empirical work in real-world situations and problems is the core of the usefulness of concepts of ES. Minding as highlighted by Jax et al., 2018, ES assessment can be carried out in an enormous range of ecological and socio-economic contexts therefore even the operationalization must be studied *ad hoc*. Furthermore, multiple paths and methodologies for tackling a problem could exist, but they should be chosen depending on the specific ecological, social, economic and political contexts. Moreover, the ES concept may give rise to (alternative) solutions that may compete with more conventional ways of dealing with problems, such as engineered or technological solutions.

Real-world problems are often “wicked” problems, which at first are seldom clear-cut and well-defined, but often rather complex and subject to differing interpretations and interests. That complexity must be accounted for, emphasizing that there is no simple way to approach complex system problems but rather the need to consider equal different expert voices.

1.3. Ocean-Climate nexus

1.3.1 Climate Change definition and causes

The Intergovernmental Panel on Climate Change (IPCC), the United Nations body for assessing the Science related to climate change, has reported that each of the last four decades has been successively warmer than any decade that preceded it since 1850. The global surface temperature was 1.09 (0.95 to 1.20) °C higher in 2011–

2020 than 1850–1900, with larger increases over land, 1.59 (1.34-1.83) °C than over the Ocean, 0.88 (0.68 to 1.01) °C. The last IPCC report (AR6 - April 2022) defined climate change as a mega-trend that impacts all facets of the environmental, social, and economic spheres. It has stated unequivocally that human influence has warmed the Atmosphere and the Ocean through the use of fossil fuels (i.e., oil, coal, natural gas) released into the atmosphere as greenhouse gases (GHG) (CMIP6-IPCC). Since around 1750 increases in GHG (i.e., Carbon dioxide (CO_2), Methane (CH_4), Nitrogen dioxide (N_2O), and chlorofluorocarbons (CFC)) concentrations have been observed. These gas concentrations have continued to increase in the atmosphere, reaching annual averages of 410 parts per million (ppm) for carbon dioxide (CO_2), 1866 parts per billion (ppb) for methane (CH4), and 332 ppb for nitrous oxide (N_2O) in 2019. The Land and the Ocean have taken up a near-constant proportion (globally about 56% per year) of CO_2 emissions from human activities over the past six decades, with regional differences. Widespread, rapid, and highly non-linear changes in the Atmosphere, Ocean, Cryosphere, and Biosphere have occurred with climate feedbacks and thresholds of abrupt change. These changes over decades to millennia will not be avoided if the thresholds are overcome leading to irreversible mutation of the human life-support system.

We are already witnessing climate change effects, consequently the following paragraphs will be divided into, **observed changes** linkable to climate change at **present** and **projected** climate-induced **changes** in the **future**. Generally, to project climate change impacts in the future, an ensemble of General Circulation Models (GCMs) and, Earth System Models (ESMs) is used. Mechanisms of the Atmosphere, Ocean, Cryosphere, and Land are susceptible to experiments, even if not impossible, therefore the changes in a changing climate are tested with numerical models. The General Circulation Models (GCMs) typically simulate all the compartments of Earth (i.e., Atmosphere, Ocean, and Land). Moreover, there are climate models such as the Earth System Models (ESMs) which may comprehend the carbon cycle, atmospheric chemistry, ice sheets, and vegetation dynamics, nitrogen cycle but also ecosystems, urban or crop systems. The models are numerical approximations of the Earth system that, support the attribution of observed changes to specific forcings, and are the best available information for assessing future climate change impacts. These models could receive as inputs different Shared Socioeconomic Pathways (SSPs) which will lead to different **Representative Concentration Pathways (RCPs)** as concentrations of GHG in the Atmosphere and Ocean which will cause a different warming capacity of the system. The systematic set of models used by the IPCC uses an ensemble of the global-scale model (including both GCMs and ESMs), in a scientific effort called Coupled Model Intercomparison Project (CMIP). At the time of this thesis, the most updated CMIP is the 6th (runned in 2018) which is the one used for the analysis presented in CHAPTER II of the present thesis. The ensemble is used because model outputs can be biased due to uncertainties in their physical equations or parameterizations, specification of initial conditions, knowledge of external forcing factors, and unaccounted processes and feedback (Hawkins & Sutton, 2009; Deser et al., 2012; Gupta et al., 2013). Hence, it is safer to withdraw conclusions based on different model results.

1.3.2 Climate change impacts the physics of the Ocean

Heat, water, and biogeochemically relevant gases (e.g., oxygen (O2), carbon dioxide (CO2)), exchange at the air-sea interface, and Ocean currents and mixing caused by winds, tides, wave dynamics, density differences, and turbulence redistribute these throughout the global Ocean. The Ocean is a fundamental climate regulator on seasonal to millennial time scales since seawater has a heat capacity four times larger than air. Accordingly, the Ocean has taken up about 93% of the anthropogenic excess heat in the climate system (Resplandy et al., 2019).

1.3.2.1 Observed

It is measured that the global Ocean has warmed unabated since 1950 (Fig. 3), and that since 1993, the rate of Ocean warming has more than doubled. **Warming** has ranged from 3.22 ± 1.61 ZeptoJoule per year (ZJ/yr) (surface–700 meters depth) and 0.97 ± 0.64 ZJ/yr (700–2000 meters depth) between 1969 and 1993, to 6.28 ± 0.48

ZJ yr–1 (surface–700 meters depth) and 3.86 ± 2.09 ZJ yr–1 (700–2000 meters depths) between 1993 and 2017. The deep Ocean below 2000 m has warmed since 1992, especially in the Southern Ocean. To date, globally the Southern Ocean accounted for 35–43% of the total heat gain in the upper 2000 meters between 1970 and 2017 with an increase to 45–62% between 2005 and 2017.

This observed surface Ocean warming and high latitude in addition to freshwater inputs, due to land ice melting, are making the surface Ocean less dense relative to deeper parts of the Ocean. It inhibits the mixing between the surface and deeper waters leading to **stratification**. The mean stratification of the upper 200 meters of the Ocean has increased by 2.3 ± 0.1% from the 1971–1990 average to the 1998–2017 average.

Consequently, **a loss of oxygen** has occurred from the surface to 1000 meters depths. Datasets spanning from 1970 to 2010 show that the open Ocean has already lost oxygen by a very likely range of 0.5–3.3% over the upper 1000 meters, alongside a likely expansion of the volume of oxygen minimum zones by 3–8% (IPCC reports this data in SROCC,2019 special report with medium confidence). This oxygen loss is primarily due to increasing Ocean stratification but even as an effect of changing ventilation and biogeochemistry.

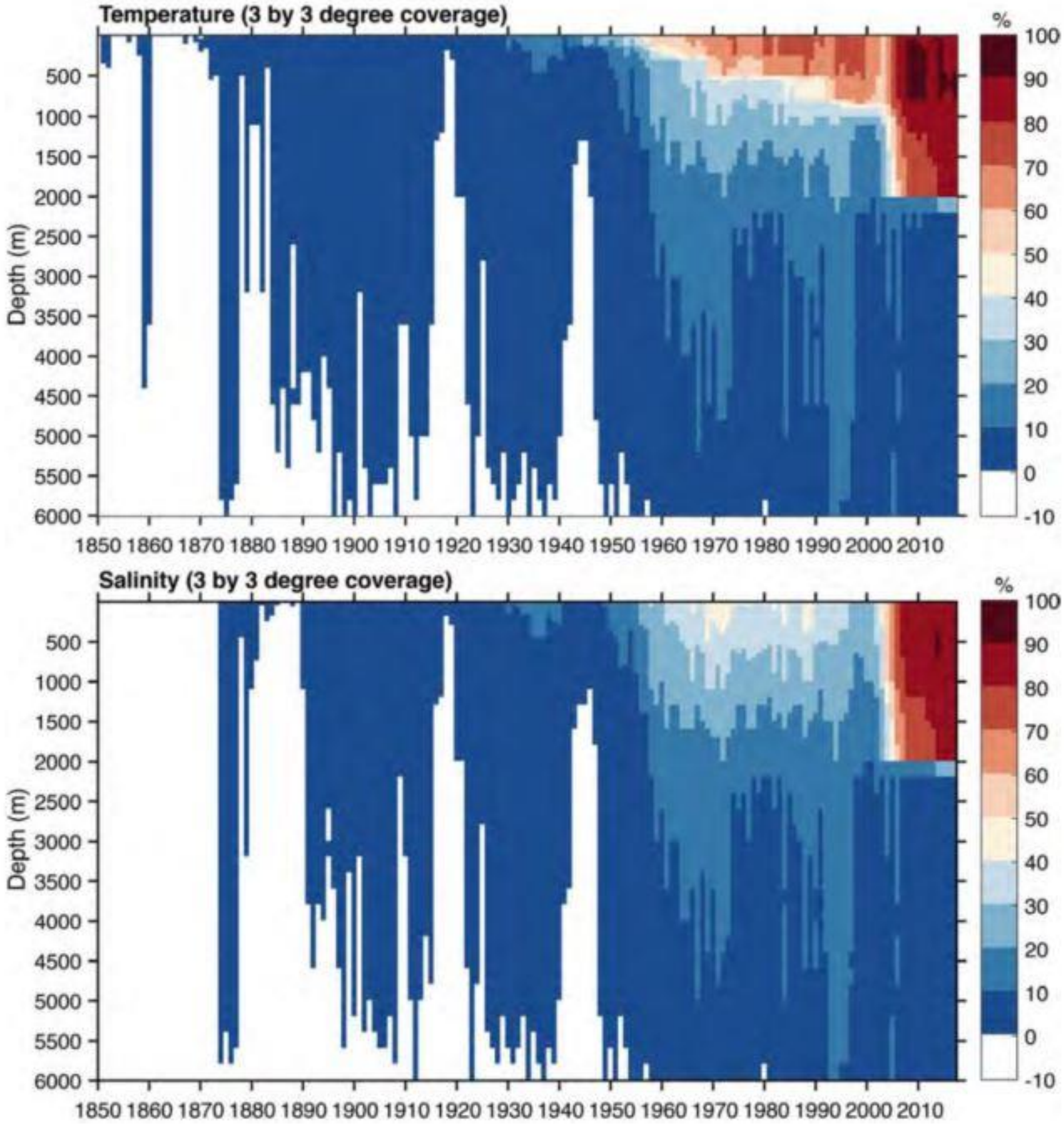


Figure 3: Global changes as a percentage (%) of Temperature (above) and Salinity (below) from 1850 to 2020 following a profile of Ocean depth from the surface to 6000 meters (SROCC, 2019).

The Ocean has already taken up between 20–30% of total anthropogenic CO2 emissions since the 1980s. It caused the Ocean to undergo **increasing acidification**. Open Ocean surface pH has declined by a very likely range of 0.017–0.027 pH units per decade since the late 1980s, with the decline in surface Ocean pH very likely to have already emerged from background natural variability for more than 95% of the Ocean surface area. **Global mean sea level** (GMSL) **is rising**, with acceleration in recent decades due to increasing rates of ice loss from the Greenland and Antarctic ice sheets, as well as continued glacier mass loss and **Ocean thermal expansion**. Together with relative sea-level rise marine heatwaves, increases in tropical cyclone winds, and rainfall, exacerbated extreme sea-level events and increased coastal hazards.

Marine heatwaves defined when the daily sea surface temperature exceeds the local 99th percentile over the period 1982 to 2016 (Hobday et al., 2016), have doubled in frequency since 1982 and are increasing in intensity. Globally, marine heat-related events have become longer-lasting and more extensive. It is very likely that between 84–90% of marine heatwaves that occurred between 2006 and 2015 are attributable to the anthropogenic temperature increase.

1.3.2.2 Projected

The warming and surface acidification already emerged in the historical period will exacerbate in the future (Fig. 4). An increase in Ocean temperature, over the 21st century, is projected at unprecedented conditions. By 2100, the top 2000 m of the Ocean are projected to **take up 5–7 times more heat** under RCP8.5 (or 2–4 times more under RCP2.6) than the observed accumulated Ocean heat uptake since 1970. Changes are expected in the upper Ocean stratification, as well. The annual mean density **stratification** of the top 200 m (averaged between 60ºS and 60ºN latitude) **is projected to increase** by 12–30% for RCP8.5 and 1–9% for RCP2.6, for the period 2081 to 2100 relative to the period 1986–2005, inhibiting even more vertical nutrient, carbon and oxygen fluxes. SROCC IPCC special resport stateted with respectively medium, medium low and medium confidence that by 2081–2100 under RCP8.5, **Ocean oxygen content, upper Ocean nitrate content, net primary production, and carbon export** are projected to **decline** globally by very likely ranges of 3–4%, 9–14%, 4–11%, and 9–16% respectively, relative to 2006–2015. Oxygen loss between 100 and 600 m depth is projected to emerge over 59–80% of the Ocean area by 2031–2050 under RCP8.5.

Continued carbon uptake by the Ocean by 2100 is virtually certain to exacerbate Ocean acidification. **Open Ocean surface pH is projected to decrease** by around 0.3 pH units by 2081–2100, relative to 2006–2015, under RCP8.5. For RCP8.5, there are elevated risks for keystone aragonite shell-forming species due to crossing an aragonite stability threshold year-round, in particular in the Polar and sub-Polar Oceans by 2081–2100. For RCP2.6, these conditions will be avoided this century, but some eastern boundary upwelling systems are projected to remain highly vulnerable.

Marine heatwaves are projected to become more frequent, durable, increase in extent and intensity (maximum temperature reached). Climate models project increases in the frequency of marine heatwaves by 2081–2100, relative to 1850–1900, by approximately 50 times under RCP8.5 and 20 times under RCP2.6.

Sea level will continue to rise at an increasing rate. Extreme sea level events, that are historically rare (once per centuray in the recent past) are projected to occur frequently (at least once per year) at many locations by 2050 in all RCP scenarios, especially in trophical region. This increase will continue after the end of the 21st century under all RCP scenarios, with projected changes in waves and tides which will vary locally in whether they amplify or ameliorate the sea level rise related hazards. The global mean sea level (GMSL) rise is projected to be in 2100 of 0.43 meter (0.29–0.59 meters) more with respect to 1986–2001 for the RCP2.6 and 0.84 meter (0.61–1.10 m, likely range) for the RCP8.5. The rate of global mean sea level rise is projected to reach **15 mm/yr** (10–20 mm/yr) under RCP8.5 in 2100, and to exceed several centimetres per year in the 22nd century.

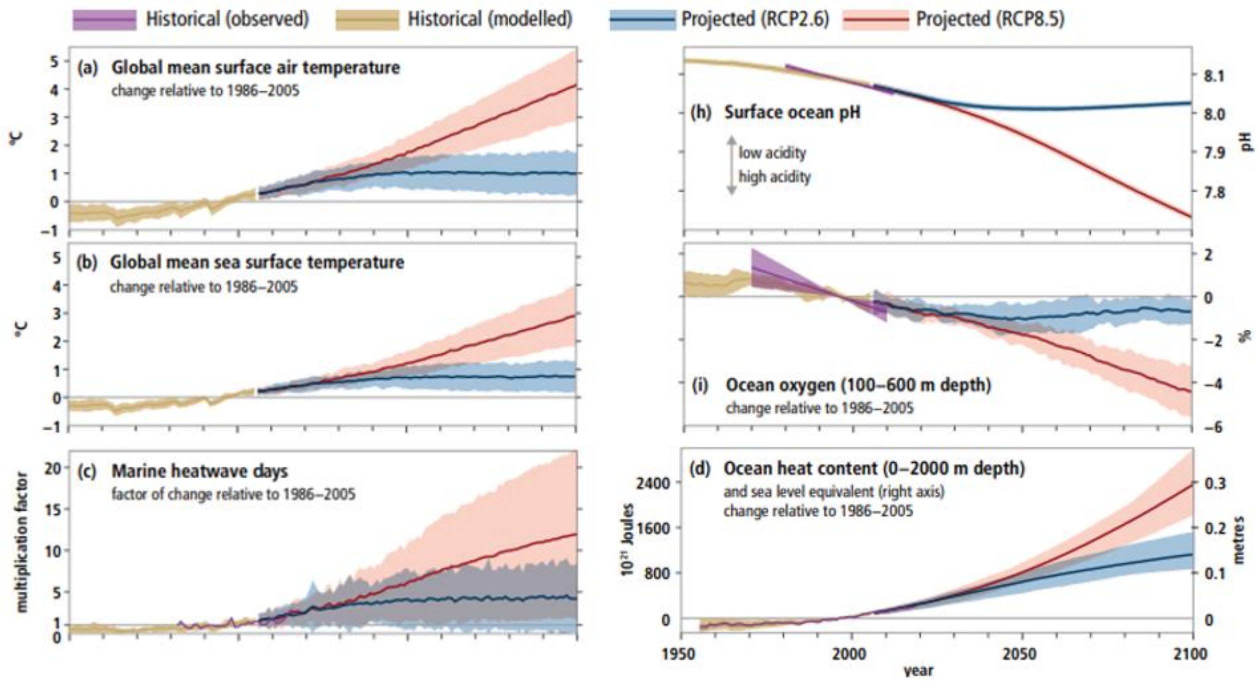


Figure 4: Historical and Projected climate change impacts on the physics of the Ocean (1950-2100)

1.3.3 Climate change ecological impacts on the Ocean

1.3.3.1 Observed

Traditional ecology developed concepts, such as ecological succession (the process by which the structure of a biological community evolves over time) and climax (the dyamic state of apparent termination of ecological successions), to indicate that, differences in environment and climactic states drive ecological communities to change in dynamic equilibria (Connell & Slatyer 1977). When abiotic conditions are relatively stabel in space and time representing an optimum condition, the most desirable, ecological communities reach the climax but are the transient (non-stable) states that enance biodiversity over time and allow the systsem to change. Often ecology is approached with human tendency of having everything under controll and crystalize and statis in its status. However, species adapt to new environmental conditions, if the time of changes align with time of adaptation (i.e., physiological adaptation, turnover of species, natural selection) therefore change in space and time. However, differently from the terrestrial ecosystems the marine ecosystem is mainly constitued by ectotherms and alignment to climate change consequences can be difficult for (i) the speed at which changes are occurring, (ii) because of the high dependency of the animal body temperature to the outer environment in which they live. To adapt to climate change, since the 1950 **many marine species across various taxonomic groups have undergone shifts in biogeography** (i.e., changes in distribution of species in geographic space through time), and and changes in seasonal activities. As the definition of ecological niche (Hutchinson 1959) species moved where temperature for instance is suistable for them, **expanded their spatial ranges or moving when is colder and these have altered ecosystem structure and functioning**. Into the bargain, species responding to climate change are shifting their climatic niche along three non-exclusive axes. The three axes are: time (i.e., phenology) (Root et al., 2003; Edwards & Richardson 2004; Parmesan 2006), space (i.e., range shifts – which have already been observed in more than 1000 species) (Walther et al., 2002; Parmesan 2006) and self (i.e., physiological alterations – that allows tolerance to warmer or drier conditions and behavioural changes of diets, activities, energy budgets) (Bellard et al. 2012; Poloczanska et al., 2013). Considering the latter which is the one explored in the present thesis (CHAPTER II), the rate and direction of observed shifts in distributions are shaped by local temperature, salinity, oxygen, and Ocean currents across depth, latitudinal and longitudinal gradients. At golabl scale, poleward shifts in distributions across different marine species since the 1950s are of 52 ± 33 km per decade and 29 ± 16 km per decade for organisms in the epipelagic (upper 200 m from sea surface) and seafloor ecosystems, respectively. Moreover, **climate change has a multi-levels effect from individuals to populations, from single species to ecological networks**. In fact, the various effects on populations are likely to modify the 'web of interactions' at the community level (Gilman et al., 2010; Walther 2010) having the potential of altering biological communities, ecosystems functioning, and their associated services (ESs) to humans (Parmesan & Yohe 2003; Kortsch et al., 2015). For instance, multiple climate-related drivers on low trophic level such as phyto and zooplankton have affected food web structure and function, and therefore fisheries (ES). Impacts of climate change affect not solely pelagic but even benthic species (i.e., species living close to the sea floor), in particulat, warm-water coral reefs and rocky shores dominated by immobile (which cannot migrate poleward or deepward), calcifying (e.g., shell and skeleton producing) organisms such as corals, barnacles and mussels, are currently impacted by both extreme water temperatures and Ocean acidification. Prolonged periods of high environmental temperature and dehydration of the organisms pose high risk to intertidal ecosystems. On the coastal areas, vegetated ecosystems which help buffer the impacts of sea level rise are disappearing. In response to warming, distribution ranges of seagrass meadows and kelp forests are expanding at high latitudes and contracting at low latitudes since the late 1970s, and in some areas episodic losses occur following heatwaves. Nearly 50% of coastal wetlands have been lost over the last 100 years, as a result of the combined effects of localised human pressures, sea level rise, warming and extreme climate events.

1.3.3.2 Projected

Ocean warming, acidification, oxygen loss, nitrate content and net primary production change will reach their treshold of impact on marine ecosystems all prior to 2100 for over 60% of the Ocean area under RCP8.5 and over 30% under RCP2.6. Projected Ocean warming reveals **changes in net primary production** (with areas of increase and areas of decrease) altering biomass, secondary production and community structure of marine ecosystems. Due to changes in net primary production the sinking flux of organic matter from the upper Ocean is projected to decrease largely leading to a **decrease** in global **biomass of marine animal communities** and **a shift in species composition are projected** over the 21st century in Ocean ecosystems from the surface to the deep seafloor **under all emission scenarios**. Highly-confidence projections have been made for the global-scale biomass of marine animals across the foodweb is projected to decrease by 15.0 ± 5.9% by the end of the 21st century relative to 1986–2005 reference period under RCP8.5 (Fish-MIP effort under CMIP6 scenario - Tittensor et al., 2021).

In the benthic realm, ocean warming, oxygen loss, acidification, and a decrease in flux of organic carbon from the surface to the deep Ocean are projected to harm habitat-forming cold-water corals, which support high biodiversity, partly through decreased calcification, increased dissolution of skeletons, and bioerosion. As a result, 95% or more of the deep sea (3000–6000 meters depth), seafloor area and cold-water coral ecosystems are projected to experience declines in benthic biomass under non-mitigation policy scenario (RCP8.5).

Vulnerability and risks are the highest where and when temperature and oxygen conditions both reach values outside species' tolerance ranges because they can even trigger species extinctions. Over and above, the majority of predictive models indicate rather alarming projections on how climate change will affect ecosystems, with the worst-case scenario leading to extinction rates that would qualify as the sixth mass extinction in the history of the Earth (Barnosky et al., 2011).

Climate change impacts on marine ecosystems are generally similar of above descriptions, but the magnitude of change is rather different among biomes (Fig. 5). For instance, the temperate water of the semi-enclosed Mediterreanean Sea basin, as the study area of the reported in CHAPTER II of the present thesis, is defined as hotspot of biodiversity as well as hotspot for climate change effects (Coll et al., 2010; Marbà et al. 2015). Here, impacts of warming are already evident on from phytoplankton (Molinero et al., 2005; El Hourany et al., 2021) to marine vegetation (Diaz-Almela et al., 2007; Monserrat et al., 2022), invertebrates (Calvo et al., 2011; Martín-Vélez & Abellán 2022), and vertebrates (Gambaiani et al., 2009; Lasram et al., 2010; Albouy et al., 2012,2013,2014; Azzurro et al., 2019; D'Amen & Azzurro 2020). The impacts range from impacts on growth, survival, fertility, invasion from trophical species (mainly from the Red Sea), migration and phenology of pelagic and benthic organisms. Overall, 50% of biological impacts in the Mediterranean Sea occur at summer with surface temperature anomaly of about 4.5°C (Marbà et al., 2015). The Mediterranean Sea is warming at two to three times the rate for the global ocean (Vargas-Yanez et al., 2008), showing an increase in the occurrence of hot extremes by 200-500% throughout the region (Diffenbaugh et al., 2007).

Eventually, non-climatic human activities in the Mediterranean Sea such as fishing and marine pollution have the potential to exacerbate climate change induced ecosystem impacts.

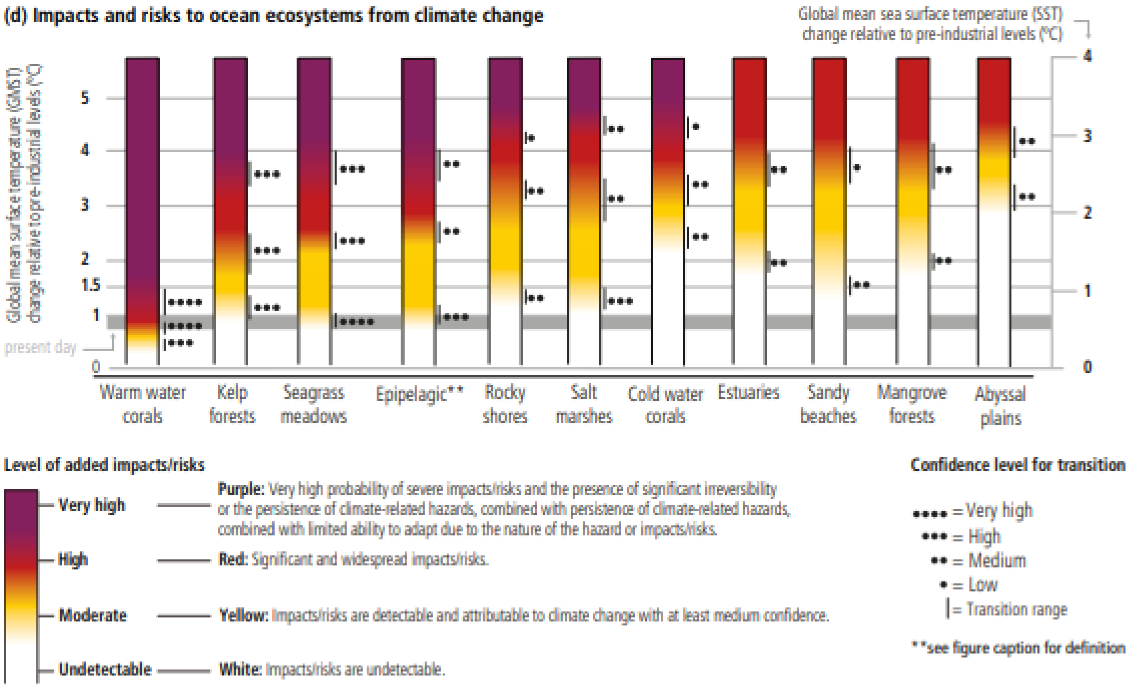


Figure 5: Projected changes, impacts and risks for marine ecosystems. Warm water corals: corals which form coral reefs in global trophic water being at the base of the most productive ecosystem on Earth. Kelp forests: typical habitat former in subtrophical areas. Seagrasses meadows, Epipelagic, Rocky shores, Salt marshes, Cold water corals (deep corals), Estuaries, Sandy beaches, Mangrove forests, Abyssal plains.

1.3.4 Observed impact on ecosystem services and the human society

The Ocean covers 71% of the Earth surface. All people depend directly or indiretly on the Ocean. For food, regulation of climate, recreational activities, renewable energy, trade, transport and benefits for health and well-being (UN, 2017 –

declaration for the Ocean Decade (2020-2030)). The 40% of human population worldwide live within 100 Km from the coastline making the Ocean health our health. These social communities in close connection with coastal environments as well as Small Island Developing States (SIDS), home for 65 million people, or polar areas are particularly exposed to Ocean changes. Other communities further from the coast are also indirectly exposed to changes in the Ocean, such as trough extreme weather events, food provision and recreational activities.

Changes in the Ocean have impacted marine ecosystems and ecosystem services (ESs) with regionally diverse outcomes, challenging their governance. For instance, municipalities and industry are beginning to address infrastructure failures associated with coastal flooding and some coastal communities have planned for relocation (Ogie et al., 2018; Habel et al., 2020; Shimamura & Mizunoya 2020). In past decades, exposure of people and infrastructure to natural hazards have increased due to growing population, tourism and socio- infrastructure to natural hazards have increased due to growing population, tourism and socio- economic development. In the future, global mean sea level rise will cause the frequency of extreme sea level events at most locations to increase. Local sea levels that historically occurred once per century are projected to occur at least annually at most locations by 2100 under all RCP scenarios.

Many low-lying megacities and small islands are projected to experience historical centennial events at least annually by 2050 under RCP2.6, RCP4.5, and RCP8.5. Theimpacts on coastal and marine provided ESs have negative consequences for health and well-being especially for some conditions such as indigenous peoples and local communities which depend on fisheries. As explored in the previous chapters, warming-induced trophic changes modify the spatial distribution and abundance of some fish and shellfish stocks with locally dependent positive and negative impacts on catches, economic benefits, livelihoods, and local culture. Shifts in species distributions and abundance have challenged international and national Ocean and fisheries governance, in terms of regulating fishing to secure ecosystem integrity and sharing of resources between fishing entities. The maximum catch potential of fisheries, even if projected with medium confidence, will decrease by 20.5–24.1% by the end of the 21st century relative to 1986–2005 reference period under RCP8.5. These changes are projected to be very likely three to four times larger under RCP8.5 than RCP2.6 (Fig. 6).

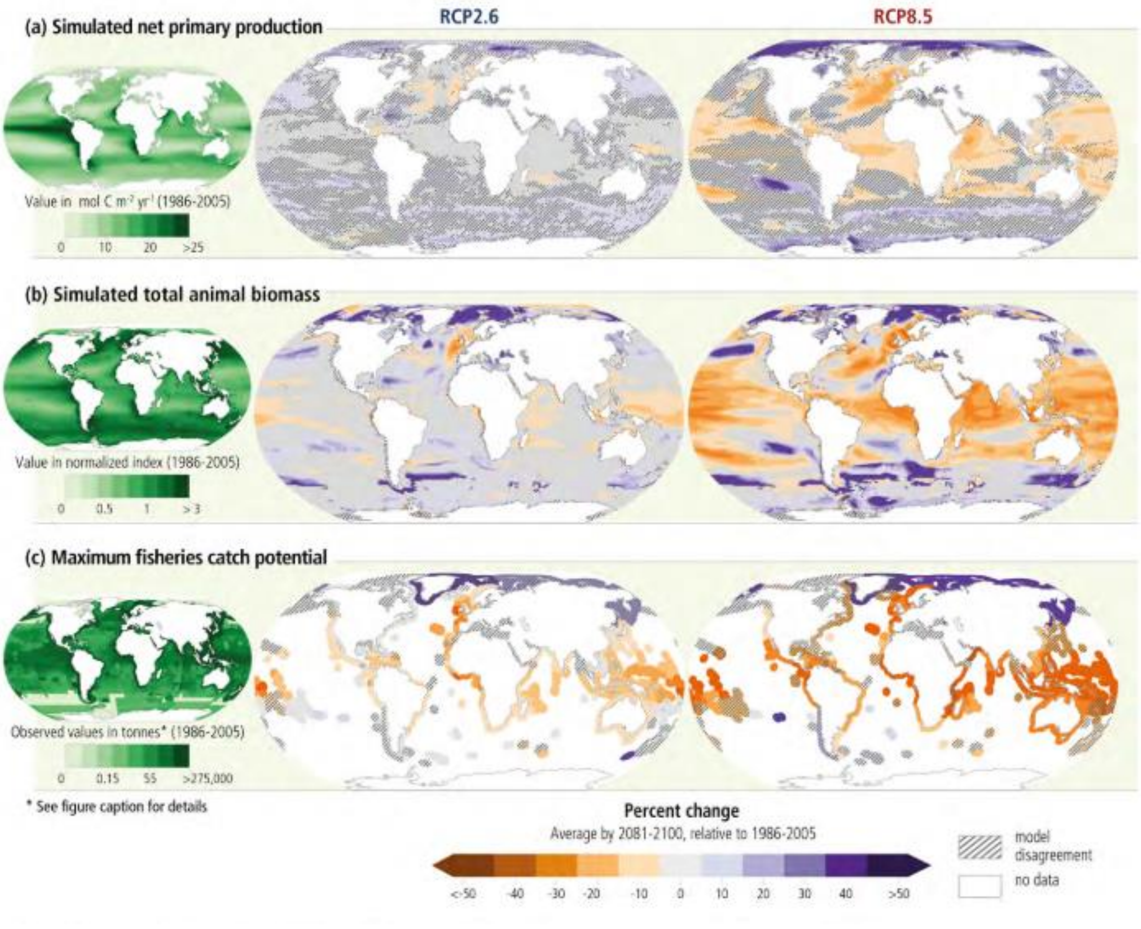


Figure 6: Trophic cascading from primary production (microbes) to fishes and fishery impact by climate change (SROCC-IPCC report, 2018).

Furthermore, in response to both climatic and non-climatic drivers (such as increased riverine nutrient run-off), harmful algal blooms are displaying range expansion and increased frequency in coastal areas since the 1980s. This will have negative impacts on food security, tourism, the local economy, and human health (Fig.7).

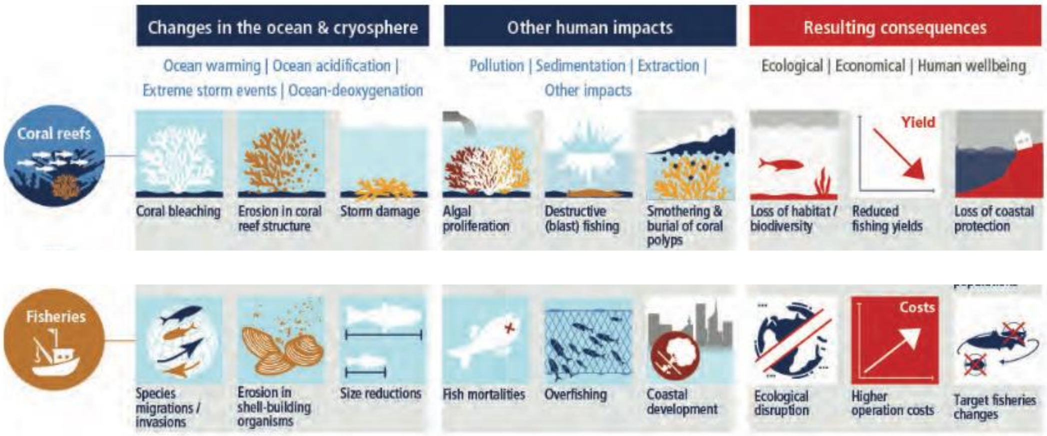


Figure 7: Summary schematic of the impacts and resulting consequences of climate change (e.g., warming, acidification, storminess and deoxygenation) and other human impacts, on coral reefs, and fisheries (SROCC- IPCC reports).

1.4. Motivations behind the research

The present doctoral project has been developed around two main research questions: (i) *How marine ecosystem functioning (EF)*[1] *will be impacted by climate change in the Mediterranean Sea?* (ii) *Do marine ecosystem functioning (EF) and ecosystem services (ESs)*[2] *share the same spatial patterns resulting in a high level of ESs where EF is high?*

In other words, the present project tackled climate change impacts on Mediterranean marine EF, seeking a better understanding of spatial relationships between EF and ESs and eventually providing management suggestions on how better conserve Nature, holistically including both EF and ESs.

1.4.1 Motivation behind the first research question

Ecosystems consist of all the biotic and abiotic components in a certain area that function together as a unit. This ensemble of stocks (structure) and fluxes or processes (functioning), both depend on the abiotic conditions (Duarte & Cebrian 1996). Hence, the mutation of climate inducing changes in the abiotic components affects both ecosystem structure and functioning. Notwithstanding the enormous importance of these impacts on ecosystem structure and the easy-to-use metrics to measure it (e.g., species richness) just recently the understanding of how climate change will impact the other component of the ecosystem namely functioning has increased. However, efforts in this direction have mainly been made on a global scale (Du pontavice et al., 2020, 2021; Tittensor al., 2021). More regional-scale studies on the climate change impacts on marine EF are needed to know , and therefore better manage the marine ecosystems in the face of climate change (Viitasalo et al., 2022).

---

[1] See paragraph 1.1 Ecosystem functioning for the definition of EF

[2] See paragraph 1.2 Ecosystem services for the definition of ESs

It is vital, not solely for the mere understanding of all possible impacts of climate change on ecosystems but even to start unrevealing feedback loops to biodiversity itself due to the interdependency between EF and species richness.

Massive is the amount of literature studying the Biodiversity and Ecosystem Functioning (BEF) relationship by a desire to understand how structure and ecosystem function (EF) are related (Worm et al., 2006; Stachowicz et al., 2007). The evidence suggests that there is a clear and direct relationship between key aspects of EF and various measures of biodiversity (e.g., richness, number of functional groups, evenness) (Balvanera et al., 2006). Balvanera et al., 2006 even suggested that the small number of negative relationships reported in the literature tend to be associated with studies that measured properties at the population level (i.e., individual species density, cover, or biomass), rather than those which looked at community-level characteristics (i.e, density, biomass, consumption). The structure-functioning positive relationship can assume three forms (Cardinale et al., 2006) (Fig. 8), Schwarts et al., 2000 suggested curves A and B while Cardinale et al., 2006 added the C relationship. The important difference between curves A and B is that in A, ecosystem function is highly sensitive to variations in biodiversity, while in B, there is a saturation pattern.

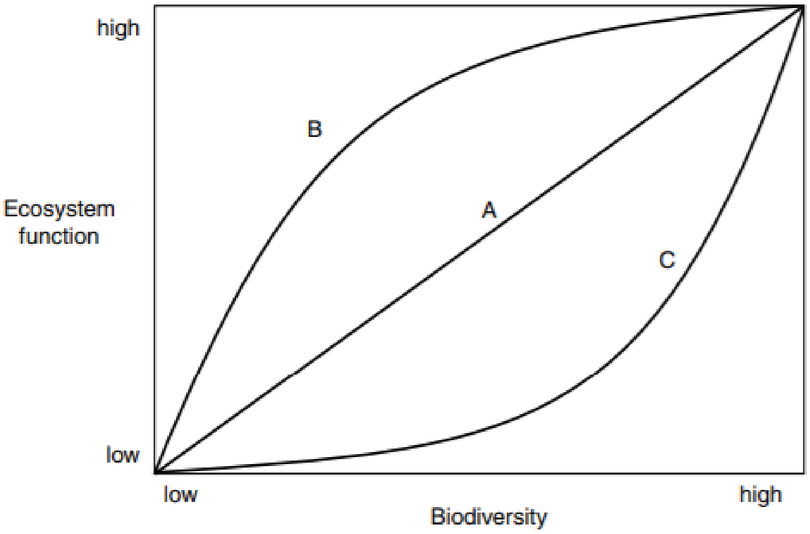


Figure 8: Potential relationships between biodiversity and ecosystem functioning (after Schwartz er al., 2000; Cardinale et al., 2006).

The latter (the B curve) is demonstrated to be the most reliable BEF relationship by a growing wealth of studies (Worm et al., 2006; Cardinale et al., 2006; Cardinale et al.,2007; Cadotte et al., 2008; Srivastava et al., 2009; Cardinale et al., 2011; Liang et al., 2016). Accordingly, ecosystem functioning reaches a plateau at a high level of biodiversity. This saturating pattern describes a decline of ecosystem functioning occurring much more rapidly at low levels of species richness than at higher ones. In this latter situations, ecosystem functioning appears to be buffered from the effects of biodiversity loss.

The accredited pattern of the BEF relationship (i.e., the B curve) highlights **the foremost importance of conducting climate change impact studies over EF and not solely on biodiversity (structure) to avoid possible misleading descriptions of the climate change effects on ecosystems.** Species always move and change if environmental conditions do, hence it is the functioning that should be conserved while the structure might change. Ecological communities adapt to physicochemical conditions either by genetical biodiversity evolution as the “red queen hypothesis” states (Van Vallen, 1973) or by changing set of suitable species that in the novel abiotic conditions survive (by mean of the natural selection, Origin of the Species, Darwin, 1909). Furthermore, being the structure and the function interconnected exploring the impact of climate change on functioning can make our understanding of indirect climate change impacts on structure (trough direct impacts on functioning) more complete.

Eventually, overlooking the impacts of changing abiotic conditions due to climate change on EF solely looking at biodiversity can lead to incomplete results. This doctoral thesis, as reported in CHAPTER II, tries to shed the light on the understanding of how climate change will impact two indicators of marine trophodynamics EF (i.e. Trophic Efficiency and Flow Kinetics) using the Mediterranean Sea as a case study.

1.4.2 Motivation behind the second research question

Several studies focused on the relationship between biodiversity and ES (Costanza et al., 1997, 1998; Worm et al., 2006; Quijas & Balvanera,2010; Mace et al., 2012; Schneiders et al., 2012; Winter et al., 2018; Balvanera et al., 2022) and some (Swift et al., 2004; Egoh et al., 2009; Lindegren et al., 2018; Birkhofer et al., 2018) have already seen that global patterns of biodiversity, ecosystem services (ESs), and human impacts are poorly correlated.

Natural ecosystem biodiversity, together with EF, underpin human health and wellbeing as different international frameworks report. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has proposed the vision of "One Health" for foraging the protection of ecosystem health to guarantee our health. Furthermore, good ecosystem status is the foundations on which our economy is built (Morgera, 2020). For these reasons, we should protect Nature (Fig. 9).

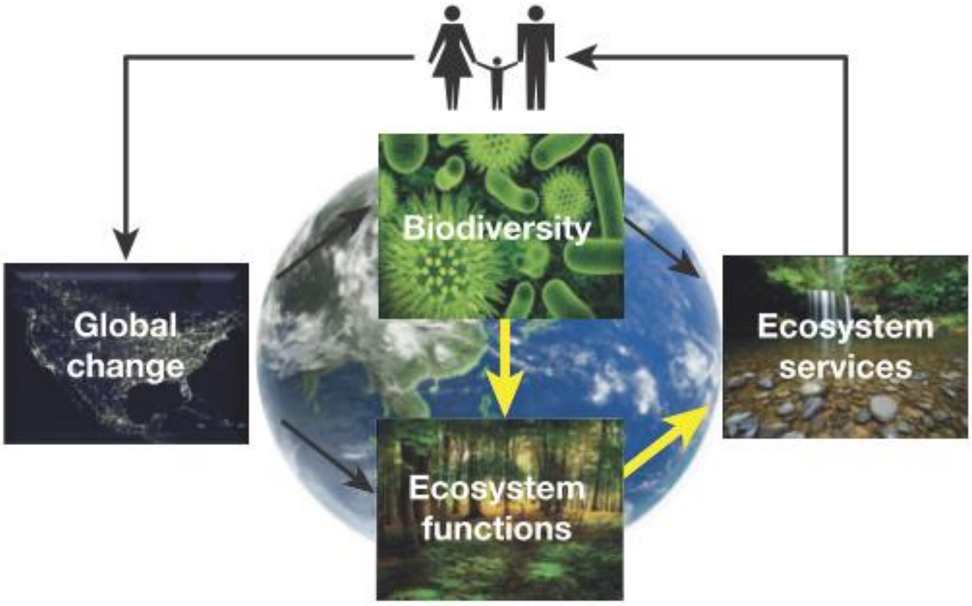


Figure 9: Biodiversity, EF, ESs are tightly linked to human well-being and survival (NASA and Shutterstock.com credits).

Notwithstanding the growing wealth of studies using the concept of the ES cascade (Fig. 2) (Haines-Young and Potschin, 2010; De Groot et al. 2010; La Notte et al., 2017; Rugani et al., 2019 and many more) and the usefulness of the ES cascade to make the ES concept more accessible to the non-science community (Potschin-Young et al., 2018) some authors (Balvanera et al., 2006; Cardinale et al., 2012) have already pointed out that ESs are not always positively related to an ecologically meaningful definition of EF, as the ES cascade instead describes (Peterson et al., 2010). Costanza et al., (2017) highlighted that the conceptualization of ES through the service cascade is for some aspects an oversimplification, as it does not capture the complex and dynamic connections occurring between the ecosystem structures, functioning, and the benefits we derive.

A key role in misconception in the ES cascade framework is played by the **antropocentrically defined concept of ecosystem function** as the capacity or capability of the ecosystem to do something potentially useful to people (Costanza et al., 1997; Daily 1997; de Groot et al., 2002; de Groot et al., 2010; Haines-Young and Potschin, 2010; Potschin and Haines-Young, 2011). However, as Jax (2005) stated, the term 'function' can mean several other things in ecology. It can mean ecosystem capability to resist or be resilient and include the whole set of processes that operate within an ecosystem (like nutrient cycling or predation) as Wallace (2007) and many other ecologists used it. In other words, EF in the ES cascade is considered only if a human beneficiary exists (Potschin & Haines-Young, 2011) even if **part of EF serves Nature itself and the other non-human.** These aspects, although of foremost importance, may not be positively directly related to human benefits as the ES cascade instead describes. Furthermore, any research endeavor advance should come through careful tests of the theory with the identification of empirical strengths or limitations (Bertness et al., 2014). Avoiding collecting empirical evidence of what is theoretically stated has the risk to create more confusion than help. Hence, the present thesis attempts to reveal some empirical evidence on whether and how EF and ESs are linked. To accomplish it, in CHAPTER IV, I have considered five EF indicators and eleven ES indicators assessed for the same year (2018) and the same area of study (Adriatic Sea).

Whether the two research questions above seem independent the *fil rouge* connecting them is that the study of the magnitude at which climate change will impact EF should be clarified to foresee a better projection, after knowing how EF and ES are connected, on how ES supply will be affected by climate change.

CHAPTER II

CLIMATE CHANGE IMPACTS MARINE TROPHODYNAMIC AS A PROXY OF ECOSYSTEM FUNCTIONING IN THE MEDITERRANEAN SEA

This Chapter has been submitted as a scientific paper in the peer-reviewed journal "Global Change Biology" (Impact Factor: 10.86). Bibliographic details: Basconi L., Moullec F., Drira S., Peck M., November 2022.

Abstract

Climate-driven changes in marine species abundance and diversity are well documented, still, their ecosystem-level implications are poorly understood. Understanding whether biodiversity change can maintain fundamental ecosystem functions represents a pressing challenge in an increasingly warmer world. In this study, we measured the effect of climate change (ssp1-RCP 2.6; ssp5-RCP 8.5) on the efficiency and speed of trophic transfer using the climate suitability models of 579 fish species. Results have shown a general reduction of trophic transfer efficiency and a slighter reduction of trophic transfer speed at the Mediterranean basin scale. Nonetheless, there are areas in the Mediterranean Sea which shown even an increase in both trophic efficiency and speed. These areas, being characterized by climate-resilient ecological conditions, could be identified as priority areas for conservation as well as for fishery adaptation in a changing climate.

Introduction

The Intergovernmental Panel on Climate Change (IPCC) special report on the Ocean and the Cryosphere in a changing climate (SROCC) (Pörtner et al., 2019) resume the immense pressure that climate change is exerting on marine ecosystems. It portrays an impactful future for most life in the Ocean and for the billions of people who depend upon it (Cheung et al., 2009; du Pontavice et al., 2020, 2021; Poloczanska et al., 2013; Pörtner et al., 2019; Tittensor et al., 2021). While climate change-related impacts on single species, taxa, or trophic levels have been explored, research at the ecosystem-level is still in its infancy (Montoya & Raffaelli, 2010). However, the first joint report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services IPBES and IPCC considers as imperative that the two crises of biodiversity loss and climate change are tackled together, as they are highly interrelated (Pörtner et al., 2021). In this context, the Mediterranean Sea is one of the best grounds to explore this interrelationship being both a hot spot of biodiversity (Coll et al., 2010) and climate change effects (*see* Marbà et al. 2015, for a review). The Mediterranean Sea is both the largest and deepest semi-enclosed sea in the world, and it is characterized by oligotrophic conditions (i.e., low nutrient content). It hosts about 17000 marine species (Coll et al., 2010) currently impacted by climate change. Evidence of climate change impacts on marine Mediterranean species ranges from viruses and bacteria (Danovaro et al., 2001, 2004, 2009), to habitat formers (Chefaoui et al., 2018; Marchini et al., 2019) passing through invertebrates, fishes, mammals, birds and reptiles (Sanz et al., 2003; Lasram et al., 2010; Rivetti et al., 2014; Gambaiani et al., 2022).

In the present article 579 endemic or established fish species in the Mediterranean Sea have been considered (Basconi et al., *in press* - data paper). Fishes together with the other components of the Mediterranean marine ecosystem (i.e. cephalopods, crustaceans, and bivalves) are the taxon mainly relevant for human food provisioning, constituting together with mammals and birds the foremost part of the higher levels of Mediterranean seafood web. Moreover, whether abiotic conditions mutate fishes can **move** changing their location and therefore the trophic interactions with other species and overall modifying the ecological community. To date, climate change effects on the Mediterranean fishes have been studied on their geographical distributions (Lasram et al., 2010 - 75 endemic fishes), species richness and mean body size (Albouy et al., 2012 – 288 coastal fishes), the number of feeding links and connectance (Albouy et al., 2013 - 288 coastal fishes), phylogenetics and functional diversity (Albouy et al., 2014 - 288 coastal fishes), life strategies (Koutsidi et al., 2020- 205 nektonic species). Climate change-induced effects at the ecosystem level has been already studied (Moullec et al., 2019; Schickele et al., 2021). However, organic matter and energy fluxes across the ecosystem (i.e., trophodynamics) in the Mediterranean Sea remain unexplored.

Nevertheless, studying the matter and energy fluxes across species, from small fishes up to top predators ($2 \leq TL \leq 4$) is an important proxy of ecosystem response to climate change (Gascuel et al., 2008; Maureaud et al., 2017; Eddy et al., 2020; du Pontavice et al., 2020, 2021) and it should be studied as complementary to climate change effects on biodiversity (Katsanevakis et al., 2020). Both terrestrial and marine studies report a Michaelis-Menten relationship between biodiversity and ecosystem functioning, at least when primary productivity is used as the ecosystem functioning indicator (Cardinale et al., 2011, Liang et al., 2016). Therefore, just looking at climate change-induced biodiversity shifts can overestimate climate change effects, especially in a species-rich ecosystem, such as the Mediterranean Sea (see ‘insurance’ effect - Yachi & Loreau, 1999). Furthermore, there is compelling evidence from experimental and theoretical studies (e.g., Balvanera et al., 2006; Cardinale et al., 2006; Naeem et al., 2009; Reiss et al., 2009) that the rate of ecosystem process change is much less pronounced across high levels of biodiversity, where there may be redundancy among species. This implies that ecosystem functioning could depict ecosystems as more resilient to climate change compared to biodiversity.

In this paper, we explored the effect of climate change on the Mediterranean trophodynamics as the interspecific ecosystem functioning (*sensu* Boero & Bonsdorff, 2007). The ecosystem functions considered are characterized by matter and energy flows (i.e., exchange of biomass between biological compartments or trophic levels). The use of trophodynamic indicators have been considered valuable to measure marine ecosystem functioning (ICES, Tam et al., 2017). Hence, in the present paper context, the food web is defined by the biomass flow from secondary producers (TL=2) to top-predators (TL=4) where the transfer of matter and energy occurs via consumption/predation (i.e., primary production is not included).

We explored in the Mediterranean Sea two key properties/processes in an ecosystem: trophic transfer efficiency and speed (*see* Lindeman, 1942; Gascuel et al., 2008 for details). They have been measured through the Efficiency Cumulated Indicator (ECI) and Biomass Residency Time (BRT). ECI represents the trophic transfer efficiency as the efficiency of energy and organic matter transfers occurring in food web. In other words, ECI is the indicator of the ratio between the production rates of two adjacent trophic levels and cumulated determine how much organic matter is transferred up to the food web (Lindeman, 1942; Jenning et al., 2002; Andersen et al., 2009). BRT is the cumulative measure of the inverse of flow kinetics or speed of trophic transfer (Gascuel et al., 2008; Schramski et al., 2015). It shows how fast species eat and are eaten in the food web. ECI and BRT have never been explored in the regional context of the Mediterranean Sea while the interest lay mainly on global studies (Maureaud et al., 2017; du Pontavice et al., 2020, 2021).

The present paper aim is to project how climate change will alter the Mediterranean marine ecosystem functioning at end of the century (2100) in respect of the reference current period under two Shared Socio-Economic Pathways 1- Representative Concentration Pathways 2.6 (ssp1-RCP 2.6) and the Shared Socio-Economic Pathways 5- Representative Concentration Pathways 8.5 (ssp5-RCP 8.5).

Materials and methods

Species modeled

To calculate ecosystem scale indicators, all Mediterranean enedemic or established Osteichthyes (Actinopterygii - Cope, 1887) and Chondrichthyes (Elasmobranch) species were considered. The species list was compiled from, FISHMED (the most updated list of fish species in the Mediterranean Sea), the Mediterranean CIESM Atlas of establishing fishes (i.e., fish described in the CIESM list as “E”), fish species reported in Coll et al., 2010, SeaAroundUs (i.e., list of commercially targeted species by fishery), and FishBase (all the fishes downloaded for the large marine ecosystem, Mediterranean Sea). Eventually, we obtain 771 fish species as residents in the Mediterranean Sea (Basconi et al., *in press* – data paper). Check for synonyms was done using Worms(https://www.marinespecies.org/aphia.php?p=taxdetails&id=129881) and the Eschemeyer’s website (R package – taxize was used for species “ID” downloading).

The global occurrences of the 771 were at first downloaded from public data repositories such as OBIS (Ocean Biogeographic Information System), GBIF (Global Biodiversity Information Facility), iNaturalist (A Community for Naturalists), VertNet (vertebrate biodiversity networks), Ecoengine (UC Berkeley’s Natural History Data) for the period 1993 to 2018. Species with at least 20 occurrences were retained decreasing the number of species used for the present work from 771 to 579.

Environmental data

Secondly, we downloaded and averaged temperature (T) and salinity (S) at the global scale. In marine ecological modeling, environmental variables are frequently treated as flat using 2-dimensional (2D) surface data (e.g., Sea Surface Temperature (SST), Sea Surface Salinity (SSS)) despite the Ocean

3-dimensional (3D) spatial component. The disparity of environmental conditions at the surface and depth is evident (SROCC IPCC, 2019) and it must be accounted for. Hence, to characterize the species' thermal and salinity tolerance limits (critical thermal for survival: minimum and maximum) the 579 fish species were divided based on their vertical habitats in, pelagic-neritic, pelagic oceanic, general pelagic, benthopelagic, demersal and benthic (as defined in FishBase (www.fishbase.org; Froese & Pauly, 2017). The water column data of temperature and salinity from the global oceanic physics reanalysis 001-030 - GLORYS12V1 per each pixel/model cell, month, and year (1993-2018) have been averaged as follows: surface-50m (general pelagic), surface-100m (pelagic neritic), surface-400m (pelagic oceanic), bottom-up to 100m (benthopelagic), bottom-up to 50 m (benthic, demersal, bathydemersal), bottom to -100m from the surface (bathypelagic) (Table 1).

Species class	Averaging from … to …	Source	Examples of species belonging to this categories (as defined in FishBase)
General pelagic	Surface-50 m	Median of min-max depths of species belonging to the "pelagic" FishBase-based classification	*Scomberesox saurus, Esox lucius, Sardinella maderensis*
Pelagic neritic	Surface-100m	Median of min-max depths of species belonging to the "pelagic-oceanic" FishBase-based classification	*Alosa alosa, Exocoetus obtusirostris, Sarda sarda, Trachurus trachurus*
Pelagic oceanic	Surface-400m	Median of min-max depths of species belonging to the "pelagic-neritic" FishBase-based classification	*Acanthocybium solandri, Champsodon nudivittis, Mola mola, Thunnus alalunga*
Bathypelagic	Bottom to -100m	FishBase glossary definition of "Bathypelagic"	*Alepisaurus ferox, Lampanyctus intricarius, Stomias boa, Vinciguerria attenuata*
Benthic, Demersal, Bathydemersal	Bottom-50m	Sea Bottom Temperature (SBT) used for previous efforts of niche spatial distribution (Lasram et al., 2020)	*Atherina boyeri, Callionymus reticulatus, Epinephelus caninus, Gobius niger, Parablennius tentacularis, Scorpaena maderensis, Torpedo torpedo, Zeugopterus punctatus*
Benthopelagic	Bottom-100m	FishBase glossary definition of "Benthopelagic"	*Aetomylaeus bovinus, Diplodus puntazzo, Salmo salar, Trachurus picturatus*

Table 1: Classes of vertical habitat considered for temperature and salinity vertical averaging to characterize the species' thermal tolerance limits (critical thermal for survival: minimum and maximum) of the 579 fish species considered. Temperature and Salinity data were obtained from GLORYS produces and distributes global ocean reanalyses at eddy-permitting (1/4°) resolution that aims to describe the mean and time-varying state of the ocean circulation, including a part of the mesoscale eddy field. GLORYS surface boundary conditions are derived from atmospheric ECMWF reanalyses. Assimilated observations are in-situ temperature and salinity profiles and satellite sea surface temperature. The data assimilation method is based on a reduced-order Kalman filter using the SEEK formulation (CMEMS, 2021).

Climate Suitability Models (CSM)

The 579 species' climatic niche distributions (i.e., climatic suitability) have been obtained using a 'model averaging' approach where the probability of climatic suitability is a function of temperature and salinity. The 'model averaging' allows us to account for model-based uncertainties in the modeling process, hence an ensemble forecasting approach (Araùjo & New, 2007) based on different statistical algorithms are highly reliable to apply. The seven algorithms used to calculate species Presence/Pseudo-absence (P/A) were: (1) Generalized Linear Models (GLM), (2) Generalized Additive Models (GAM), (3) Classification Tree Analysis (CTA), (4) Random Forests (RF), (5) Flexible discriminant analysis (FDA), (6) Artificial neural network (ANN) (7) Generalized boosting model (GBM). The analys was run thanks to the BIOMOD2 package (Thuiller et al., 2009) in the R statistical and programming environment (R Development Core Team, 2021). These correlative models have the advantage to use relevant environmental variables to infer the species geographical range when there is a lack of knowledge of the species physiology (Moullec et al., 2022). The present analysis assumes no dispersal limitation towards new climatically suitable areas because species invasions show that fish may reach a mean dispersal rate of 221 ± 5.4 km/yr on the northern side of the Mediterranean Sea (Lasram et al., 2008).

Calculation of ecosystem functioning indicators

Eventually, for calculating the Efficiency Cumulated Indicator (ECI) and Biomass Residency Time (BRT) between trophic levels 2 and 4 for the Mediterranean marine ecosystem, we collected the biological variables of the 579 fish species modeled. Trophic level estimates for each species when available have been extracted from a Mediterranean Review (Karachle & Stergiou, 2017) being the TL calculated already for the only Mediterranean, otherwise from FishBase (www.fishbase.org; Froese & Pauly, 2017). Other parameters required as input for empirical equations as growth parameters and ecological features were taken from FishBase (www.fishbase.org; Froese & Pauly, 2017). We used estimates of the needed parameters originating from the Mediterranean Sea when available, and average values from related taxa or global scale when not. Solely modified in the usage of the correct vertical temperature for the species (*see* Table 1) as Tj in the production/biomass (P/Q) ad consumption/biomass (Q/B) ratio calculations instead of Sea Surface Temperature (SST). To implement this trophodynamic approach, species-specific data were transformed into trophic class data, building a trophic spectrum using an established methodology (Gascuel et al., 2005; Gascuel et al., 2011) implemented in the R package EcoTroph (Colléter et al., 2013). The smoothing function distributed the value of catches, from the Scientific, Technical and Economic Committee for Fisheries (STECF) database (2014-2018), unique per each Geographical Subareas (GSA) for each species over a range of trophic classes, using classes with a width of 0.1 trophic level unit and a log-normal distribution. This assumes to mimic the within-species variability of trophic levels. Eventually, the species-specific values were then aggregated geographically to calculate the ratios between Production/Biomass (P/B) and Production/Consumption (P/Q) per pixel (1/12° - 7x7Km horizontal resolution). Finally, these two parameters were raised from trophic classes to the entire food web, using two cumulated indicators, Efficiency Cumulated Indicator (ECI - adimensional) and Biomass Residency Time (BRT - years).

Climate change scenarios

The same procedure was used to calculate ECI and BRT at the end of the 21st century. Two Intergovernmental Panel on Climate Change (IPCC) scenarios, the Shared Socio-Economic Pathways 1- Representative Concentration Pathways 2.6 (ssp1-RCP 2.6) and the Shared Socio-Economic Pathways 5- Representative Concentration Pathways 8.5 (ssp5-RCP 8.5) for the end of the 21st century (2091–2100) was used to simulate potential future climate suitability of species (source: CMIP6 efforts - Earth System Models (ESMs)). To ensure consistency with the resolution and extent of the climate suitability results for the current period (1993-2018), we gridded the sea temperature rasters to a 1/12° (ca. 7x7Km horizontal resolution at Mediterranean latitudes), using bilinear interpolation. From the ESMs the two environmental predictors temperatures and salinity, at the correct mean vertical habitat of the species, were extracted and the average for the period 2091–2100. Eventually, we calculated the climate-driven range expansions projecting the potential future geographical location for each of the 579 species under the two changing climate scenarios (Basconi et al., in press – data paper). These spatially explicit climate suitability results have been the starting ground for the calculation of the ecosystem functioning indicators (i.e., ECI and BRT). Because the study focused on community-level changes, no variation in fishery catches was included in the empirical equations, the von Bertalanffy growth parameters were also assumed to remain unchanged. These assumptions allowed us to focus on the changes in trophodynamics solely due to changes in water column temperature and salinity as effects of climate change. Furthermore, we did not explicitly account for the effects of changes in diet or behavior on individual or population levels as well as for niche adaptability of single species in new environmental conditions.

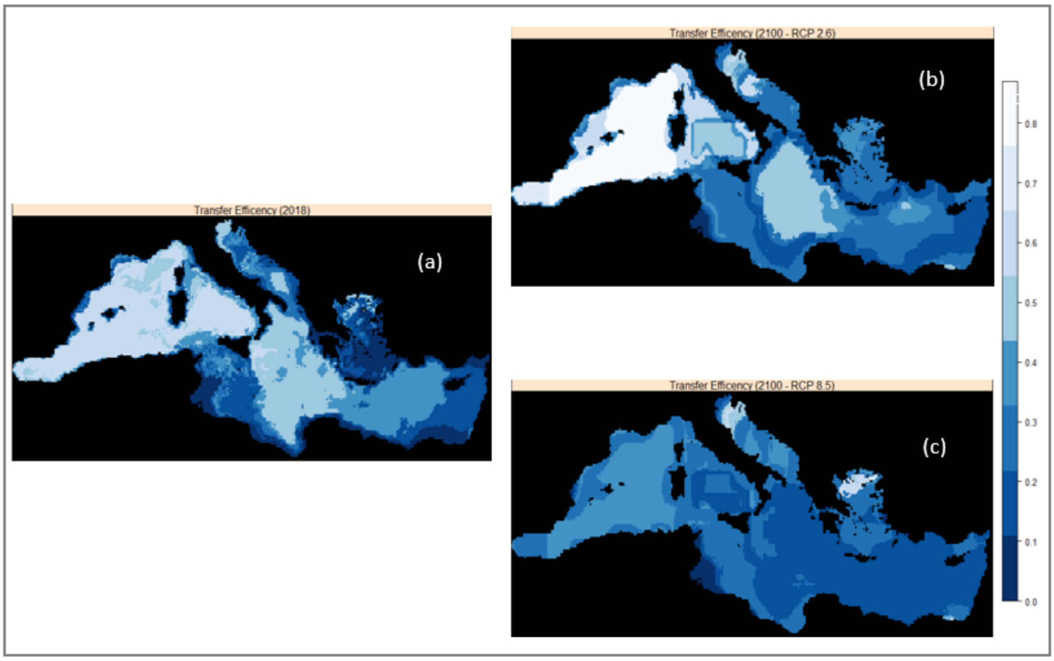
Eventually, the two trophodynamic indicators for 2091-2100 under spp5-RCP 8.5 climate change scenario were expressed as anomalies, in respect to the avarage 2009-2018, to provide an overview of temporal variation of ecosystem functioning within each GSA.

Results

Starting from the final dataset describing the extent of occurrences (i.e., climate suitability models) of 579 endemic or established fish species on a 1/12° (7x7Km horizontal resolution) grid covering the whole Mediterranean Sea (Basconi et al., *in press* data paper) trophic Efficency Cumulated Indicator (ECI) and Biomass Residency Time (BRT) have been calculated.

In the current period (2014-2018), the whole Mediterranean Sea shows a heterogeneous pattern both for ECI and BRT. ECI ranges from 0.022 to 0.646 with an average transformed in the percentage of 3.65 %. BRT ranges from 2.54 to 9.35 with an average of 6.1 years. For 2091-2100, under the 'effective mitigation' climate change scenario (spp1-RCP 2.6) in the Mediterranean Sea ECI is projected to range from 0.03 to 0.87 with a mean value of 4.16 %, and BRT is projected to range from 3.42 to 6.90 with an average of 6.1 years. BRT remains stable in respect to current conditions in scenario spp1-RCP 2.6. While under the 'no effective mitigation' climate change scenario (ssp5-RCP 8.5) ECI is projected to range from 0 to 0.68 with a mean value of 2.55% and BRT is projected to range from 3.4 to 6.8 with an average of 5.9 years.

Fig. 10a shows heterogeneous values of ECI in the current period (2014-2018). The general pattern is shaped in lower values on the continental shelves in comparison with open water values in the Mediterranean basin. Fig. 10b shows an increase in ECI for the Western basin and a slight decrease in the Eastern basin already under spp1-RCP 2.6. Nevertheless, the Western-Eastern dichotomic behaviors are more evident under spp1-RCP 2.6 it is consistent even under ssp5-RCP 8.5, where it decreases more in the Eastern than in the Western. In the 'no effective mitigation' scenario, a general decrease of ECI interests the whole Mediterranean basin besides some exceptions of increase (e.g., Adriatic Sea, Aegean Sea). BRT does not change as much as ECI, even if a general slight increase can be seen both in ssp1-RCP 2.6 and ssp5-RCP 8.5. There are areas, such as the Adriatic Sea, the Aegean Sea, and the most Eastern-South part of the basin that shows a decrease in BRT under 'no effective mitigation' scenario.



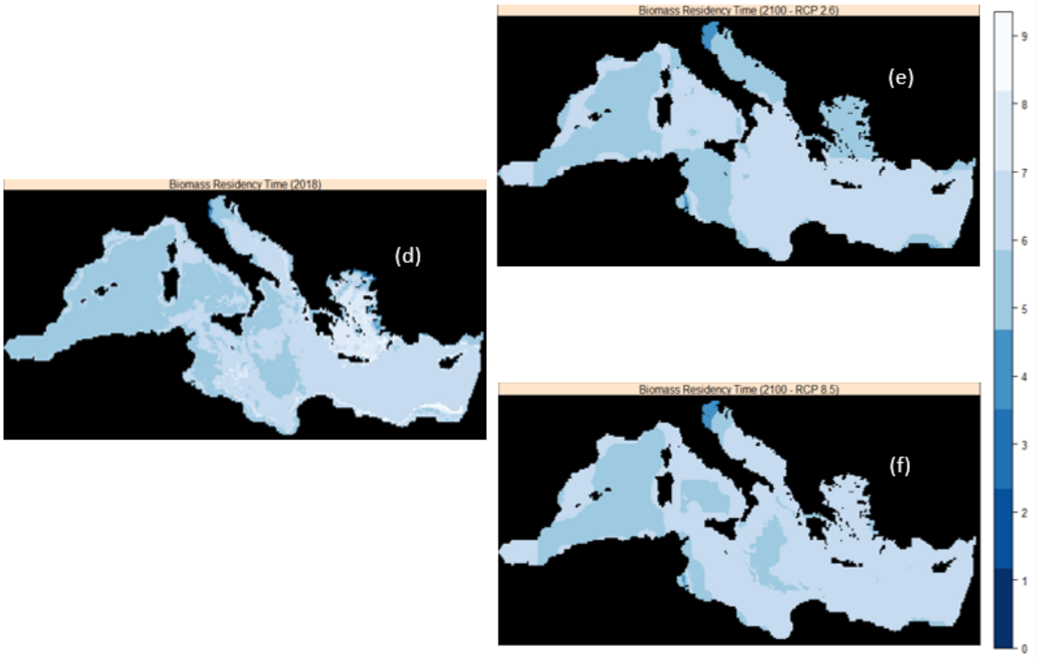


Figure 10: Mediterranean (a) ECI for the current period, (b) ECI in 2091-2100 under spp1 – RCP 2.6, (c) ECI in 2091-2100 under spp2 – RCP 8.5 and, (d) Biomass Residency Time (BRT) for the current period (e) BRT in 2091-2100 under spp1 – RCP 2.6, (f) BRT in 2091-2100 under spp2 – RCP 8.5

At the basin scale under the ssp5 – RCP 8.5 scenario ECI decreases as the main trend. This reduction starts in the Eastern basin as it is evident even under spp1- RCP 2.6 scenario (Fig. 20, *Appendix A*). GSA 15,16, GSA 17, and GSA 25 represent exceptions for the general trend (Fig. 11a). Under the same climate change scenario, BRT does not mutate much showing a trend invention at the GSA 13,14 (Fig. 11b) neither under spp1- RCP 2.6 scenario (Fig. 21. *Appendix A*).

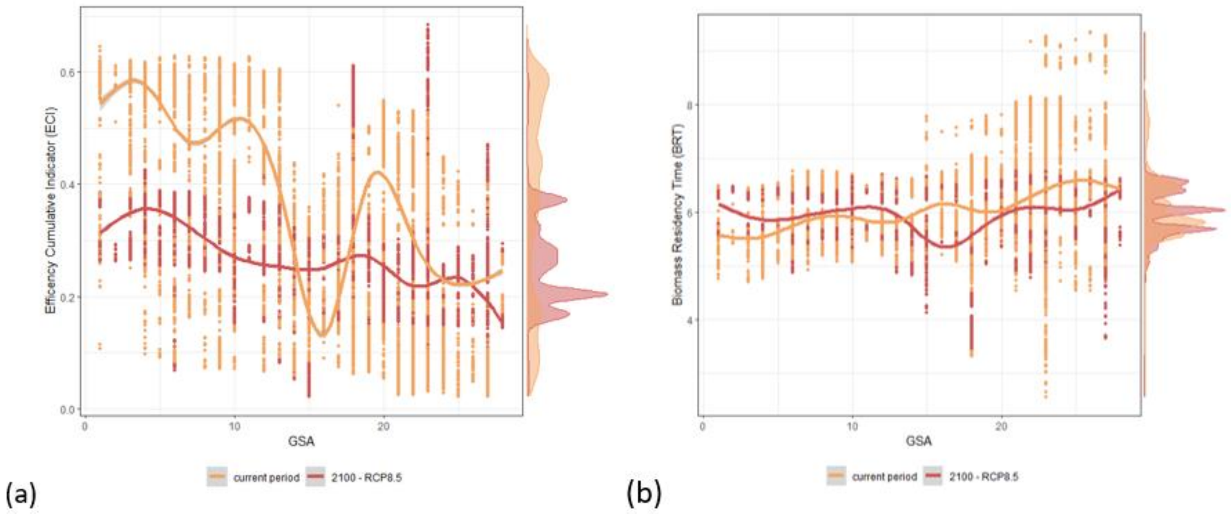


Figure 11: Mediterranean (a) Efficiency Cumulated Indicator (ECI) and (b) Biomass Residency Time (BRT) for the end of the 21st century in respect to the current avarage (2014-2018) as average per Geographical SubAreas (GSAs).

Fig. 12 reports ECI and BRT anomalies between the projected conditions for the end of the 21st century under ssp5-RCP 8.5 and the current condition (2014-2018). We confirm a more evident change in ECI compared with BRT. In Fig. 12b, BRT shows close to zero anomalies. At the GSA scale, it is possible to see patterns shaded at the basin scale analysis where it is clear the different behaviour at the GSA scale (Fig. 22 and Fig. 23, *Appendix A*). Fig. 12a shows some significant positive anomalies of ECI across the basin for 2091-2100 (ssp5 – RCP 8.5). Consistently, the same 'exceptional' areas, even if still less pronounced, show projected negative anomalies of BRT. Single GSA changes under the two climate change scenarios as well as for the current period are reported in This divergence between the general basin-scale ECI ad BRT trends and 'exceptional' areas are consistent both in spp1 -2.6 and spp5 – 8.5 (anomalies maps under spp1-RCP2.6 *see* Fig. 20 and Fig. 21 in *Appendix A*).

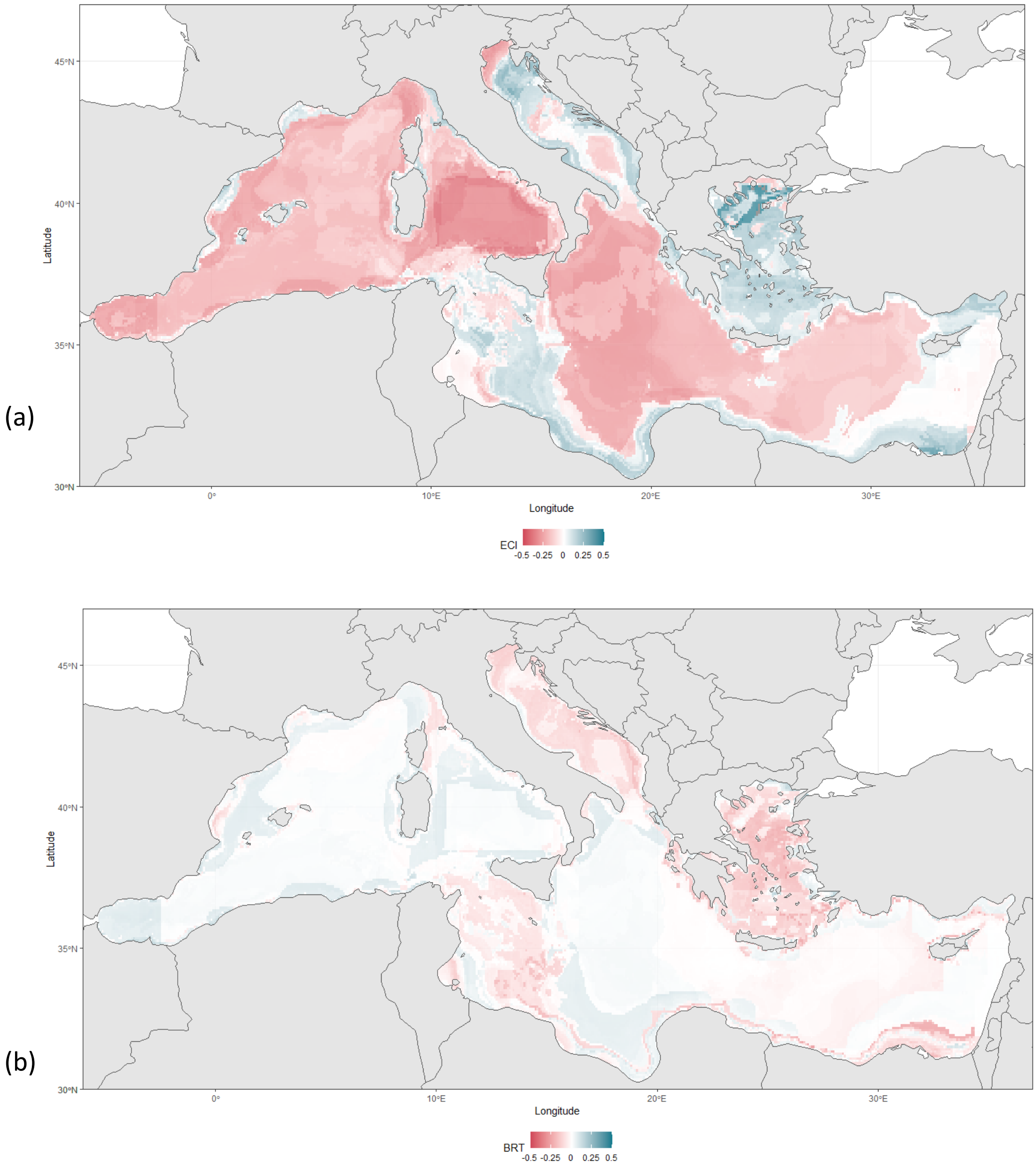


Figure 12: (a) Positive and Negative changes of Efficiency Cumulated Indicator (ECI) and, (b) Positive and Negative changes of Biomass Residency Time (BRT) projected for 2091-2100 under spp5 – RCP 8.5 (2014-2018 – current period)

Discussion

The estimates of trophic Efficiency Cumulated Indicator (ECI) and Biomass Residency Time (BRT) between trophic levels 2 and 4 fit in the global ranges reported in both Maureaud et al., 2017 and du Pontavice et al., 2020 for temperate ecosystems (i.e., 2%<ECI<10% and 1year<BRT<9year).

In the same historical (Maureaud et al., 2017) and future (du Pontavice et al., 2020) studies the used environmental variables (mainly sea surface temperature) seem to alter more trophic transfer efficiency than its speed. Globally for continental shelves ecosystems, from 1950 to 2010 trophic efficiency increased together with speed (Maureaud et al., 2017) while from 2010 to 2100 under the 'no effective mitigation' scenario (ssp5-RCP 8.5) the speed continues to increase but trophic efficiency decrease as results of climate change (du Pontavice et al., 2020, 2021). Notwithstanding the use of a different methodology, even in the present study climate change appears to be more impactful on trophic transfer efficiency (ECI) than on the trophic transfer speed (inverse of BRT) as all the graphical representations of the two indicator variabilities support (Fig. 10, 11, 12). BRT does not seem to change much even between ssp1- RCP 2.6 and ssp5- RCP 8.5, while it is clear the opposite behavior of ECI under the two climate change scenarios, especially in the Western basin. In particular, under the 'effective mitigation' scenario (ssp1-RCP 2.6) on the Western Mediterranean sea, there are estimations of an increase in trophic transfer efficiency that can be a proxy of still suitable conditions for the ecological community which in the optimum conditions tends to reach the dynamic equilibrium, namely climax (Cowles 1899, Whittaker et al., 1953). On the other hand, under the 'no effective mitigation' scenario (ssp5-RCP 8.5) there is a decrease in trophic efficiency supporting once more the strong link between ECI and climate variables (i.e., Temperature and Salinity in the present analysis) (Eddy et al., 2021). Given that, lower transfer efficiency leads to more biomass at low trophic levels, with a decrease in ECI due to climate change the biomass flowing to high trophic levels would decrease. As results of this work and others (i.e., du Pontavice et al., 2020, 2021; Eddy et al., 2021) climate change will cause an overall reduction in ecosystem functioning. From a socio-economic view, the decrease in the proportion of matter and energy passed through the food web will likely impact fishery which currently targets mainly high trophic level species (ICES, FAO).

While the climate drivers could be more tightly linked to the trophic transfer efficiency than the speed, the differences between values of the Western and Easter Mediterranean basins seem to be consistent for both indicators. Especially under ssp5- RCP 8.5 where trophic transfer efficiency is estimated to decrease more evidently (Fig. 11a) in the Eastern than the Western basin while the opposite even if still less evident, is true for BRT. The reasons behind this can be found in the different ecological characteristics as the Western basin as one of the most biodiverse areas in the Mediterranean Sea (Coll et al., 2010; De Juan & Lleonart, 2010, Mouillot et al., 2011, Chatzimentor et al., 2021). Furthermore, at current time, the rapid warming has already severely modified marine ecosystems in the Eastern basin (Bevilacqua et al., 2021) causing a different response in ecosystem functioning (Schwartz et al., 2000; Cardinale et al., 2006). Accordingly, the Eastern basin shows already under ssp1-RCP 2.6 a reduction in comparison with the Western basin that is even projected to overall increase its trophic efficiency at the end of the 21st century. The present findings support the role of the Western basin as climate refugia and encourage the proposed 4D conservation priorities for the Western basin as reported by Doxa et al., 2022.

The same consistency with 4D climate-smart conservation suggestions by Doxa et al., 2022 is shown for the Aegean Sea (GSA 22) and other areas of deep-water formation (i.e., Northern Adriatic Sea (GSA 17), and Gulf of Lion (GSA 7)) (Lascaratos et al., 1999). Fig. 12a and Fig. 12b show 'exceptional' areas in ecosystem functioning pattern where the general decrease in efficiency and the slight decrease in BRT completely reverse. These 'exceptional' areas are characterized by a projected increase in trophic transfer efficiency and an increase, even if consistently less pronounced, of speed (i.e., light blue areas in Fig. 12a, light red areas in Fig. 12b). This trends of increase of both indicators here considered as proxies of ecosystem functioning dominates at the global scale in historical times (Maureaud et al., 2017). Hence, this shows the possibility of having resilient to climate change areas in the Mediterranean Sea. In fact, from a theoretical point of view, a perturbed ecosystem should lower the stored cumulative biomass (Libralato et al., 2014; Pranovi et al., 2014) and therefore increase efficiency and speed. Making possible the consideration of these 'exeptional' areas as result of resilient responses of the ecological community to disturbances (Gascuel et al., 2008). These areas are already behaving differently in terms of ecosystem functioning in the current period (2014-2018), where for instance trophic efficiency is lower in these areas than in nearby ones (Fig. 10a). Besides areas of deep-water formation, the other 'exceptional' areas can be linked to the Mediterranean thermohaline "conveyor belt" circulation (Poulain et al., 2007). This current carries water from the Atlantic Ocean through the Mediterranean Sea, becoming Levantine Intermediate Water in the Eastern basin and inverts its transfer in the area where GSA 24, GSA 25, and GSA 26 are located. Across GSA 15, and GSA 16 at intermediated depth (between 200 and 500 meters) there is an encounter of the cold, less salty Atlantic water entering the Eastern basin and the warmer, saltier Levantine Intermediate Water that is carried to the Gibraltar Strait to be released back in the Atlantic Ocean. Overall, in these two areas, Temperature and Salinity (i.e., independent variables of the present study) are different along the water column in comparison with nearby areas. Furthermore, even if not accounted for here, deep cold water is rich in nutrients and has high oxygen content supporting even more the higher functioning of the marine community.

As stated by Young et al., 2015 marine hotspots besides being areas with high biodiversity or high concentration of a population or life history stage areas are even areas of high ecological functioning. Hotspots have high conservation needs (Hazen et al., 2013) and, representing in this case even the role of climate refugia (*sensu* Tzedakis et al., 2002) have an indisputable role in climate change adaptation. The 'exceptional' areas of higher functioning unrevealed by this study are in line with results reported in Doxa et al., 2022. Whether a higher ecosystem functioning is maintained in some Mediterranean areas under climate change these portions can still act as spillover in the nearby areas as Marine Protected Areas (MPAs) (Stobart et al., 2009; Colléter et al., 2014).

Still, this analysis does not include other than climate change-induced temperature and salinity alterations such as overfishing, pollution, marine heatwaves, increase in acidification, and other drivers equally important in shaping marine ecosystem functioning (Moullec et al., 2019; Garrabou et al., 2022). These multiple stressors conditions could still prevent foodwebs from responding resiliently in the areas highlighted by the present work. However, we argue that understanding these changes in flows of energy and matter across trophic levels is essential to formulate new conservation strategies and management approaches that consider, mitigate, and help to adapt to present and future climate change effects (Vergés et al., 2019).

Furthermore, observing this pattern and unrevealing 'exceptional' areas was possible thanks to the consideration of six vertical habitats in which the 579 modeled species were identified. Modeling marine species that can live in the whole water column (e.g., Bathypelagic, Benthopelagic organisms) or close to the bottom (e.g., Benthic and Demersal organisms) with Sea Surface Temperature (SST) is highly misleading (Duffy & Chown, 2017). Many scholars have already highlighted the importance of studying the Ocean in its 4th dimension (both on the axis of space and time) as the challenge for the 21st century (Rex et al., 2005; Vassallo et al., 2020; Doxa et al., 2022). The deep-water zones, although covering 40% of the Mediterranean Sea, is often underrepresented in traditional climate suitability and ecosystem functioning studies.

Furthermore, due to the nature of the global studies on climate change impacts on trophodynamics (Maureaud et al., 2017; du Pontavice et al., 2020,2021; Tittensor al., 2021) the areas of possible resilience of the Mediterranean marine ecosystem have never been shown before. Besides, they are fundamental to understanding the complete response, resiliency included, of marine communities in the Mediterranean Sea in a changing climate. Whether we only look at biodiversity we risk overestimations of climate change effects on Mediterranean fishes while the ecological intrinsic ability of the marine communities to respond in different ways to stressors is evident. In other words, environmental changes could alter the structure (i.e., biodiversity) but resilience communities even through what is called the 'insurance hypothesis' (Yachi & Loreau 1999) can buffer reduction in ecosystem functions maintaining their overall functioning even in climate change scenarios (Schwartz er al., 2000; Worm et al., 2006; Cardinale et al., 2006; Cardinale et al.,2007; Cadotte et al., 2008; Srivastava et al., 2009; Cardinale et al., 2011; Liang et al., 2016). Ecosystems could still function independently to what species remain, disappear, or colonize the community driving the intuitions of many scholars to focus on functions measurement than structural ones in ecosystem conservations under climate change (Katsanevakis et al., 2018; Rilov et al., 2019; Bevilacqua & Terlizzi, 2020; Katsanevakis et al., 2020).

The present study highlights areas in which the marine community could show a resilient response to climate change at the end of the 21st century. Furthermore, as the European Union's Marine Strategy Framework Directive (MSFD) and its overarching plan to reach and maintain Good Environmental Status (Rogers et al., 2010) suggest, holistic ecological indicators are essential to describe the marine community responses, especially under climate change (Viitasalo et al., 2022). Ecosystem structure (i.e., biodiversity) and functioning (i.e., ecological processes) should be explored in parallel since some of them might be more sensitive than others to different components of climate change. Furthermore, conservation should be farsighted considering climate change projections as just recently been included operationally (Brito-Morales et al., 2022), paving the path for a new era of climate-smart conservation initiatives (Doxa et al., 2022). The current network of marine protected areas in the Mediterranean Sea has low climatic stability and consider mainly coastal shallow areas (Abdulla et al., 2009; Kyprioti et al., 2021; Manea et al., 2020; Doxa et al., 2022). The need to go beyond traditional conservations strategy is pressing.

The same is true for fisheries management failing to fully consider the present and future changes in environmental conditions following climate change which could highly help to plan for climate change adaptation (Möllmann et al., 2021). Adaptive management is the way forward (e.g., preserving ecosystem functions in climate change hotspots, and identifying and targeting climate refugia areas for protection) using Marine Spatial Planning (MSP) as a framework for action, especially given the push for Blue Growth (Rilov et al., 2019).

The limitation of the present work seattled in the uncertainty link to the trophodynamics approach (Hilborn 2011), the solely use of environmental variables (i.e., temperature and salinity) assuming that species ranges are mainly driven by the abiotic environment. However, this is a reasonable hypothesis for marine species as the water temperature is commonly considered the main driver of fish geographic ranges (Sabatés et al., 2006; Ben Rais Lasram & Mouillot, 2009; Cheung et al., 2009; Ben Rais Lasram et al., 2010; Guisande et al., 2013) biotic drivers are here missing. Not considering interactions among species risks to underestimates indirect effects caused by interspecific changes (Tam et al., 2017; Moullec et al., 2022). Eventually, the cause-effect pathways of climate change and multiple human activities could be included, accounting for the combination of additive, synergistic and antagonistic impacts on ecosystems (Stelzenmüller et al., 2018).

Modern approaches to Ecosystem-based Management (EbM) and sustainable use of marine resources must account for the myriad of pressures (interspecies, human, and environmental) affecting marine ecosystem functioning.

Conclusion

Marine ecosystem description still often depends on ecosystem structure indicators (e.g., biodiversity, functional traits, species biomass) and fails to consider the full complexity of the ecological system. The present work measured the impact of climate change on trophodynamics ecosystem-scale indicators (flow of biomass between trophic levels 2 and 4) in the Mediterranean Sea as proxies of ecosystem functioning. Results show a general decrease in the proportion of biomass moved through the food web under the ‘no effective mitigation’ climate change scenario for the end of the 21st century. Projecting a reduction in biomass to a higher trophic level, mainly targets of fishery link these indicators to socio-economic impacts calling for a need for fishery adaptation to climate change. This work even unrevealed areas of the Mediterranean marine ecosystem that seem to be resilient to climate change and that should be protected as climate refugia, informing a better prioritization of sites for conservation.

In a view of both Ecosystem-based Management (EbM) and the precautionary approach to climate change, the presented analysis reinforces the need to consider environmental impacts on ecosystem functioning and not solely on biodiversity as our life support system maintenance.

CHAPTER III

ECOSYSTEM SERVICES FOR SUPPORTING COASTAL AND MARINE RESOURCES MANAGEMENT, AN EXAMPLE FROM THE ADRIATIC SEA (CENTRAL MEDITERRANEAN SEA)

This Chapter has been accepted as a scientific paper in the peer-reviewed journal “Ocean and Coastal Management” (Impact Factor: 4.33). Bibliographic details: Basconi L., Stocco A., Rova S., Pranovi F., October 2022.

Abstract

Ecosystem Services (ESs) assessment is increasingly being considered as the constitutive metric to embrace the social, ecological, and economic spheres. Spatially explicit ES assessments can integrate and standardize different types of information making them comparable. In this context, a multiple coastal and marine ES assessment in the Northern-Central Adriatic Sea was carried out, considering seven ESs. Two cultural (tourism and recreational boating), two regulating (carbon sequestration and coastal erosion prevention potential), and three provisioning (mussel, whitefish aquaculture, and industrial fishery) ESs have been measured. A spatial analysis was carried out to describe (un)sustainable human uses of ecosystems in the area. Both the ESs capacity-flow conceptualization together with synergies and tradeoffs among ESs were analyzed. Results show spatial agreement for capacities while contrasting results emerged from the analysis across

flows and of the capacity-flow balance. The presence of a geographical pattern and areas of high, medium, and low ability to provide ESs across the study area was revealed together with considerations of the natural resources management in place. Some coastal provinces maximize a single ES detrimenting others, and some other provinces-built surrogates of Nature. This information can be useful in the context of the Marine Strategy Framework Directive (MSFD), and for the implementation of the Maritime Spatial Planning (MSP) in the Northern-Central Adriatic Sea.

Introduction

Human-induced changes in the oceans and coasts are being experienced worldwide (Ruckelshaus et al., 2015; Bennett et al., 2016; Resplandy et al., 2018; Van de Waal et al., 2020; Sampaio & Rosa 2020; Sampaio et al., 2021). Hence, the United Nations has declared 2020-2030 to be the Decade for Ocean Sciences for creating improved conditions for sustainable development at the sea, especially in the foreseen Blue Growth. The Blue Growth is the European initiative representing a long-term ecologically sustainable strategy to support the maritime sector's rising economy, even if some applications do not seem to align with this aim (Niner et al., 2022). For this reason, sustainable management of coastal and marine resources requires an inter and transdisciplinary framework to embrace all the spheres of sustainability (i.e. social, ecological, and economic). In this context, Ecosystem Service (ES) assessment is increasing in importance, as it is an approach accounting for human-Nature interactions (Balvanera et al., 2001; Daily & Matson 2008; Maes et al., 2012) and highlighting the socio-economic dependences on the ecological life support systems (Bastian et al., 2012). Many ecosystems are Social-Ecological Systems (SES) (Ostrom, 2009), and evaluating ESs reveals vital information and benefits that would otherwise remain hidden in the decision-making process (ME Assessment, 2005; Troy & Wilson 2006). The foremost importance of the ES framework has even been supported by the Convention on Biological Diversity (CBD), where ESs assessments have been defined as complementary to the formal conservation-based biodiversity targets. Furthermore, the EU biodiversity strategy 2030 considers imperative the ESs protection in any terrestrial, freshwater, and marine systems. Specifically for the coastal and marine ecosystems, the Maritime Spatial Planning (MSP) directive (European Commission, 2013) has the objective to "halt the degradation of ecosystem services" (Article 5) and requires Member States to consider ESs in their coastal management strategies (Article 8). Additionally, the establishment of the Intergovernmental Panel for Biodiversity and Ecosystem Services (IPBES, 2012) has shed light on the importance of ES assessments and the need to unravel ES relationships as key priorities of nations.

From the Millennium Ecosystem Assessment (M.E. Assessment, 2005) - when services were assessed most individually - efforts have been spent to assess different ESs altogether (Klain et al., 2014; UNEP, 2014; UNEP-WCMC, 2014). Multiple ES assessments have different advantages (i) to systematically assess and quantify the holistic benefits in terms of services that the ecosystems give to humans, (ii) to embrace all the direct and indirect uses of Nature with opportunities for increasing cross-sectoral communication between sectors and stakeholders (Tallis & Polasky, 2011; Hölting et al., 2019), (iii) potentially reduce management costs as multiple ESs can be managed together by strategies focusing on the ESs provider (UNEP, 2014). Notwithstanding, the intensive work of data mining and the use of independent metrics and indicators to assess each ES, multiple ES assessment is the only way to potentially depict the real socio-ecological landscape or seascape of the area. Despite the efforts to assess the ES set altogether in a spatially explicit context, multiple ES spatial assessment is rare in the marine realm due to the scarcity of spatial data. However, maps are essential in identifying and framing problems (Hauck et al., 2012; Chen et al., 2019). Since maps help to identify conflicts and synergies between ESs or among ESs and other sea uses, they can straightforwardly indicate places or areas where particular ESs or aspects of ecosystem functioning and, biodiversity are threatened. Understanding spatial synergies among multiple ESs, spatial explicitness of tradeoffs, and mechanisms behind these relationships unlock our ability to depict how different ESs interact, and how the sustainable management of seascapes can be accomplished (Costanza, 2008; Fisher et al., 2009; Hein et al., 2006; Schröter et al., 2012). For this reason, exploring both the capacity and the flow (*sensu* Villamagna et al., 2013) is essential (De Groot et al., 2010; Haines-Young & Potschin, 2010; Schröter et al., 2014). On one hand, the ES capacity is the ability of the environment to do something potentially useful to people (de Groot et al., 2010; Haines-Young & Potschin, 2010; Potschin & Haines-Young, 2011), according to a definition that considers an ES only if a human beneficiary exists (Potschin & Haines-Young, 2011). On the other hand, the ES flow is the *de facto* used ES (Burkhard et al., 2014). Capacity and flow are here defined in slightly different meanings from either supply and demand (Burkhard et al., 2012; Schröter et al., 2012; Tallis et al., 2012) or ecosystem function and service (De Groot et al., 2002; Petz & van Oudenhoven, 2012). In the present paper, ESs are anthropocentric defined.

Hence, including both the capacity and the flow in the assessment is of foremost importance when aiming to find valuable areas for ecological socio-economic conservation (Chan et al., 2006; Naidoo & Rickets 2006). Acknowledging that the use of the ES metric detaches from the conventional conservation practices of protecting Nature *per se* (i.e., ES metric allows the protection of the sole Nature used by humans), it remains the best approach to embrace interdisciplinarity and to preserve natural assets, guaranteeing ESs supply in the long term (Kroll et al., 2012). In particular, shedding light on how capacities and flows spatially co-variate, namely (de)coupling (as used in Vergana et al., 2021), can give us a better understanding of the relationships behind the observable ESs seascape. The distinction between ES capacity and flow has the crucial advantage to provide a parsimonious, but policy-relevant and operational indicator of the sustainability of human use of ecosystems (Schröter et al., 2014).

The present study represents a comprehensive assessment of the coastal and marine ES set, in the Northern-Central Adriatic Sea, looking at different scales. This multiple ES assessment leads to considerations at the transnational as well as at the trans-provincial levels. The former is between Italian and Croatian coastal sides and the latter is between regional provinces (Italy) and counties (Croatia) as two comparable administrative divisions. The latter trans-provincial approach reaches the objective to be detailed without getting lost in too fine-scale measurements. Both the used scales are profitable for the inclusion of all the social, ecological, and economic interests into stakeholders' consideration and management. Especially for the considered study area where Italy ecological monitoring is decentralized from the national level to regional agencies (ARPAs), and similarly for Croatia which has regional administrations devoted to the collection, integration, processing of environmental data, and submission of reports (Pezdevšek Malovrh et al., 2019). Eventually, doing a quantification of the same ESs with the same indicators at the transnational scale, and then switching to a provincial focus, allows potential comparisons between the effects of different management strategies across the different spatial scales. The foreground area of good use of multiple ESs creates precedents of best practices that could be repeated guaranteeing the protection of Nature and the derived coastal and marine ES set.

The present paper aims to assess seven coastal and marine ESs in the Northern-Central part of the Adriatic Sea, understanding the main ES patterns to define (un)sustainable Nature management in place for the study area. Identification of coupling and decoupling among capacities and flows and, of spatial tradeoffs and synergies among different ESs are the keys for unraveling the potential sustainable usage of different ESs.To reach these goals the present analysis has followed four steps (1) assessing and mapping ESs conceptually distinguishing between the capacity to provide ESs and the actual ES flows; (2) considering the sum of ES flows and ES capacities to detect areas of high ability to either supply (capacity) or use (flow) multiple ESs; (3) measuring the relationships (synergies/tradeoffs) among multiple ES flows, multiple ES capacities, and among capacities and flows of the same ES to explore sustainable management strategies; (4) identifying geographical patterns in the unalike socio-ecological conditions across the Northern-Central Adriatic Sea.

Materials and Methods

Study area

The Adriatic Sea is a semi-enclosed basin, located in the Central Mediterranean Sea and characterized by different types of coasts, sandy beaches on the Westside, and rocky shores on the East. The Northern-Central Adriatic Sea is a shallow basin, besides some deeper features (i.e., Pomo's pit), with an average depth of about 80 meters and it represents the widest continental shelf in the Mediterranean Sea (Pinardi et al., 2006). The coastal-marine system of this area is highly exploited by tourism representing the Italian peninsula as one of the most visited coasts by national and international tourists. Furthermore, including Venice, the area represents one of the major touristic destinations worldwide. On the other side, Croatia stands for a renowned seaside destination. The Northern-Central Adriatic coastal marine system is highly devoted to recreational boating both on the Italian and the Croatian sides, plus many sailboats and large motorboats (i.e., yachts) cruise to vacation in Croatia, especially over summer. In contrast, there is a polarity between the highly vegetated coast of Croatia and the poorly vegetated one of Italy. In addition, the Italian part suffers from the presence of soft sediment which does not protect the inland from sea hydrodynamics being one of the most endangered areas for future sea-level rise (Bonaldo et al., 2019, Perini et al., 2017). The area is also highly exploited for aquaculture purposes, with mussel production mainly on the Italian side and seabream (*Sparus aurata*, Linnaeus 1758) and seabass (*Dicentrarchus labrax*, Linnaeus 1758) farming mostly on the Croatian one. The present study area is even of high interest for industrial fishery purposes, predominantly trawling, representing one of the most fished areas of the Mediterranean Sea (Colloca et al., 2017).

Multiple ES assessment and analysis

Overall, this information drove the present work to consider seven foremost important coastal and marine ESs for the area, tourism, recreational boating, carbon sequestration, coastal erosion prevention potential, mussel and whitefish aquaculture, and industrial fishery (Tab. 2). They have been selected based on the importance they have for the area, as reported above, and to cover rather equally the three categories of cultural, regulating, and provisioning ESs, evenly balancing them in terms of direct and mediated services (*sensu* Rova et al., 2017). In the present transnational (Italy and Croatia) spatial analysis, 21 administrative units have been used corresponding to the 14 Italian coastal provinces and the 7 Croatian coastal counties (Fig. 13). Each unit considers the coastal portion, as the first 10 km inland parallel to the coastline and the marine area within 12 nautical miles from the coastline, corresponding to the national marine boundaries (Fig. 13).

The seven ES capacities and flows have been quantitatively and spatially assessed, except for the industrial fishery capacity for which spatially explicit data were unavailable, and for mussel and whitefish aquaculture capacity for which it was assessed just the Italian side, due to data availability. The capacity and flow indicators used to assess each ES are reported in Table 2.

ES class	Specific ES considered	ES description	Brief methodology used to assess the ES capacity	Indicators for ES capacity	Brief methodology used to assess the ES flow	Indicators for ES flow
Cultural	Tourism	Provision of opportunities for tourist recreation linked with marine/ coastal ecosystems (visiting and bathing)	Questionnaires directed to seaside tourists have been used to give weight to different features of the coastal-marine environment included in the composite indicator of attractiveness potential	Adimentional attractiveness potential	The number of tourists was downloaded from the Italian National Institute of Statistics (ISTAT) and the Croatian Bureau of Statistics	Tourists/ Km2/ Year
	Recreational boating	Provision of opportunities for recreational navigation with leisure boats in marine/ coastal ecosystems	Questionnaires directed to recreational boaters have been used to give weight to different features of the coastal-marine environment included in the composite indicator of attractiveness potential	Adimentional attractiveness potential	The number of leisure boats by maritime compartment was retrieved from the Coast Guard registry ('Capitaneria di Porto') and the Croatian Bureau of Statistics then divided in the marinas along both coastal sides. Questionnaires were used to set the number of both trips.	N° of boat trip/ Km2/ Year

Regulating	Coastal erosion prevention potential	Contribution to the prevention of coastal erosion/sediment transport processes	Methodology defined by Liquete et al., 2013 was used. To adapt it for application at the Northern-Central Adriatic scale, the coastal area was divided into coastal segments of around 10km	Adimentional indicator of Nature capacity to protect the coastal area	Methodology defined by Liquete et al., 2013 was used. To adapt it for application at the Northern-Central Adriatic scale, tides were excluded from the equation as of low relevancy for the present study area	Natural defense from marine-coastal ecosystems against inundation and erosion from waves, storms and sea level rise
	Carbon sequestration	Contribution to the maintenance of a favorable global climate through the sequestration of climate-influencing substances	Important habitats for the sequestration of carbon in the Northern-Central Adriatic Sea have been mapped. The rate of sequestration per each habitat was retrieved from the literature. The habitat areas were multiplied by their carbon sequestration and estimate per each habitat have been summed together	Kg CO2/ Km2/ Year	Important habitats for the sequestration of carbon in the Northern-Central Adriatic Sea have been mapped. The rate of sequestration per each habitat was retrieved from the literature. The habitat areas were multiplied by their carbon sequestration and estimate per each habitat have been summed together	Kg CO2/ Km2/ Year

Provisioning	Mussel Aquaculture	Seafood (mussel, *Mytilus galloprovincialis*) produced by aquaculture	Assessed through the eco-physiological model (R package RAC, Baldan et al. 2018) calibrated for *Mytilus galloprovincialis*	Tons of mussel producible/ Km2/ Year	The calculation of the annual national/regional (Croatia and Itay respectively) mussel production has been divided by the areas of the aquaculture farms of the territorial water to obtain the spatialized production of mussel	Tons of mussel produced/ Km2/ Year
	Whitefish Aquaculture	Whitefish (seabream and seabass, *Sparus aurata* and *Dicentrarchus labrax*) from aquaculture	Bio-energetic box model results (Porporato et al., 2020) were used. For the Northern-Central Adriatic Sea the economic scenario was selected	0-1 suitability range of sites for whitefish aquaculture production	The calculation of the annual production of *Dicentrarchus labrax* and *Sparus aurata* have been divided by the areas of the farms (Italy) or by county territorial water (Croatia) to obtain the spatialized production of whitefish	Tons of whitefish produced/ Km2/ Year

Industrial fishery	Fishing efforts by OTB, Rapido trawl, and PTM fishing gears	NA	NA	Spatial explicit fishing effort was estimated by using the Automatic Identification System (AIS) (Russo et al., 2020)	Trawls fishing effort/km2/year

Table 2. The ecosystem services (ESs) assessed for the Northern-Central Adriatic sea. Name, definition, brief methodology used for the assessment and indicators. More information on the methodologies used can be found in *Appendix B* of the present thesis.

Subsequently, all the spatially explicit ES indicators have been scaled to a 0-1 range, rasterized on a regular grid (1x1 km), and mapped (details on the single ES assessment and map can be found in *Appendix B*). Firstly, the linear sum among ES capacities and flows separately was performed, representing the full ES set. The sum was meant to be linear not to give more importance to one service or the other since all contribute equally to the Northern-Central Adriatic ES seascape. Secondly, each ES has been averaged based on the 21 spatial subunits (Fig. 13). To analyze possible relationships between capacities and flows, among flows and, among capacities a Spearman pair-wise spatial correlation (alpha=0.05) was applied. The correlation analysis was devoted to defining spatial (de)coupling among ES capacities and flows. The analyses have been performed with the open-source software QGIS 3.16 and R 4.0.3.

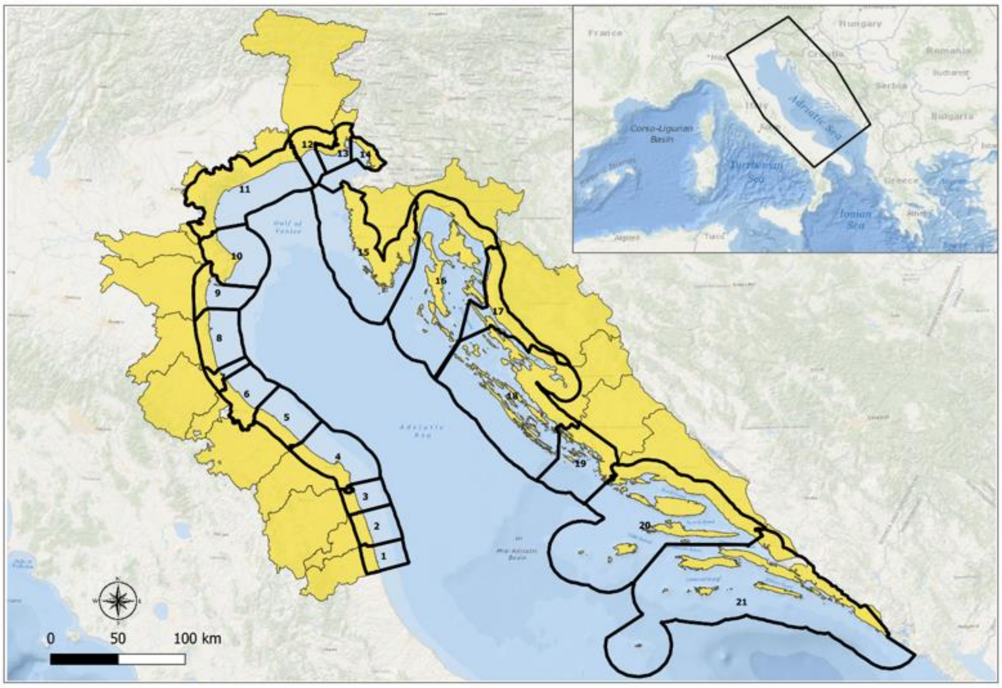


Figure 13: Study area with the spatial subunits as black boxes. Subunits are formed by the 10 km inland from the coastline and the 12 nm of territorial water. On the latitude, the division follows the administrative boundaries of provinces (Italy) and counties (Croatia) considered in the assessment. The fourteen Italian (1: Ascoli-Piceno, 2: Fermo, 3: Macerata, 4: Ancona, 5: Pesaro Urbino,

6: Rimini, 7: Forlì-Cesena, 8: Ravenna, 9: Ferrara, 10: Rovigo, 11: Venezia, 12: Udine, 13: Gorizia, 14: Trieste) and the seven Croatian subunits (15: Istra, 16: Primorje-Gorski Kotar, 17: Lika-Senj, 18: Zadarska, 19: Sibenik, 20: Split-Dalmatia, 21: Dubrovnik-Neretva).

In consideration of the spatial incompleteness of the three provisioning services (mussel, whitefish aquaculture, and industrial fishery) and their mere marine Nature they have not to be considered in the analysis of the geographical pattern. Using these ES capacities would lead to misleading and incomplete information for the study area. Especially for the industrial fishery ES for which besides ecologically sound information of stock-assessments (FAO-GFCM, 2018) and modeled Adriatic food web (Libralato et al., 2015) are present, spatially explicit information cannot be dragged at a scope useful for the present assessment. Therefore, the second part of the analysis of multiple ES considers just the two cultural (tourism, recreational boating) and two regulating services (carbon sequestration and coastal erosion prevention potential). Accordingly, a linear sum of four ES capacities, on one side, and of the four ES flows on the other side, were performed. These two linear sums namely multiple capacities and multiple flows were used to investigate how multiple ESs vary within different geographical/ecological conditions found in the study area. Eventually, the ESs composition and identity (which services contribute to what and in which magnitude) in the subunits of high, medium, and low ESs supply have been explored.

Results

Based on the current assessment, a total of 35.000.000 tourists and about 29.000 recreational boats annually visit the area. The coastal-marine habitats considered in the present assessment (broad-leaves forests, dunes, mixed forests, saltmarshes, coniferous forests, sclerophyllous plants, *Posidonia* *oceanica* (Delile, 1813), and sea-bottom burial rate) sequester annually a total of 1.243.577 tons of carbon dioxide (CO2). For coastal erosion prevention potential flow, values range from very low prevention ability for the Italian side to relatively high value for the Croatian coast. The ecosystem's ability to provide these two cultural and regulating ESs is higher on the Croatian side than the Italian one. About 29.000 and 1.800 tons of mussels and about 400 and 7.700 tons of whitefishes are produced annually for the Italian and the Croatian side, respectively. For the provisioning ESs it is not possible to have a transnational comparison of capacities because of an absence of spatial data. However, mussel aquaculture capacity is higher in the Veneto region than in other Italian regional coastal waters and whitefish aquaculture capacity increases going southward along the Italian stretch of coast. Original data ranges for each ES for the whole area of study are reported in Table 2 (ES values (means and sd) for each of the 21 administrative units can be found in Table 13 and Table 14 in the *Appendix B*).

ES	Capacity range	Capacity unit of measurement	Flow range	Flow unit of measurement
Tourism	0.14 (Forlì-Cesena)-1.06 (Primorje-Gorski Kotar)	adimensional	146 (Lika-Senj) - 16645 (Rimini)	tourists/km2/year
Recreational boating	0.003 (Macerata)-0.72 (Primorje-Gorski Kotar)	adimensional	2(Dubrovnik-Neretva)-46(Gorizia)	boat trips/km2/year
Coastal erosion prevention	0.10(Ferrara)-0.72(Istra)	adimensional	0.012(Forli-Cesena)-	adimensional

			0.78(Lika-Senj)	
Carbon sequestration	0 (Forlì-Cesena, Rimini)-30 (Lika-Senj)	Kg CO2/Km2/year	0 (Forlì-Cesena, Rimini)-30 (Lika-Senj)	Kg CO2/km2/year
Mussel aquaculture*	118(Gorizia)-189(Ferrara)	tons of mussel /km2/year	0-0.01	tons of mussel/km2/year
Whitefish aquaculture*	0.22(Rovigo) - 0.51 (Trieste)	tons of whitefish /km2/year	0.03(Split-Dalmacia)-0.73(Gorizia, Udine,Triest)	tons of whitefish /km2/year
Industrial fishery	NA	NA	0.14(Primorje-Gorski Kotar)-4.9(Macerata)	Trawls fishing effort/km2/year

Table 3. Ranges of the multiple ES capacity and flow per each ES evaluated for the Northern-Central Adriatic coastal-marine system. (*) Mussel and whitefish aquaculture have been assessed for their capacities just in the Italian territorial waters because of data scarcity.

The total capacity and flow maps (sum of all the seven ESs) reveal a complex spatial pattern that corresponds both to the environmental features and to the different human uses on the two coastal sides (Fig. 14). The high values in ES capacity are maintained along the Croatian coast, with hot spots in the Primorje-Gorski Kotar and the Lika-Senj counties.

The Italian side is poorer in total capacity along the coasts but rich in the marine realms. The highest marine values are present offshore of the province of Udine, Gorizia, Ascoli-Piceno, Fermo, Macerata, Ancona and Pesaro Urbino. The total ES flow in Croatia is high along the coasts almost everywhere (above all in the Primorje-Gorski Kotar county), whereas Italy has flow hot spots in the Gorizia, Udine, Trieste, and the Venezia provinces.

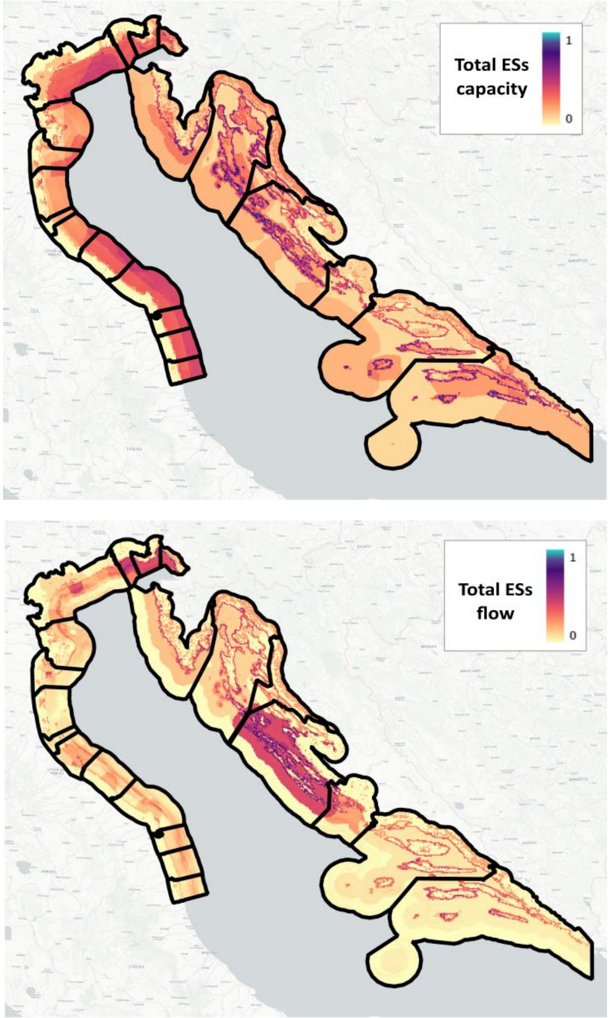


Figure 14: Maps of total ESs capacity and flow of the Northern-Central coastal-marine Adriatic system. The figure was created by the authors using the QGIS software, version 3.16 (https://qgis.org).

Results of the spatial analysis among ESs are reported in Fig. 15. Almost all the capacities showed a statistically significant (positive) correlation among themselves. On the contrary, just two statistically significant correlations have been detected among flows (industrial fishery versus coastal erosion prevention potential – negative – and whitefish aquaculture versus recreational boating – positive). Finally, the analysis of capacities versus flows showed many statistically significant – negative – correlations, as tourism, industrial fishery, mussel aquaculture, and whitefish aquaculture negatively correlated with their and others ES capacities, highlighting the presence of possible negative feedback between the use of flows over the capacities. In contrast, correlations – positive – have been detected for the coastal erosion prevention flow versus almost all the capacities.

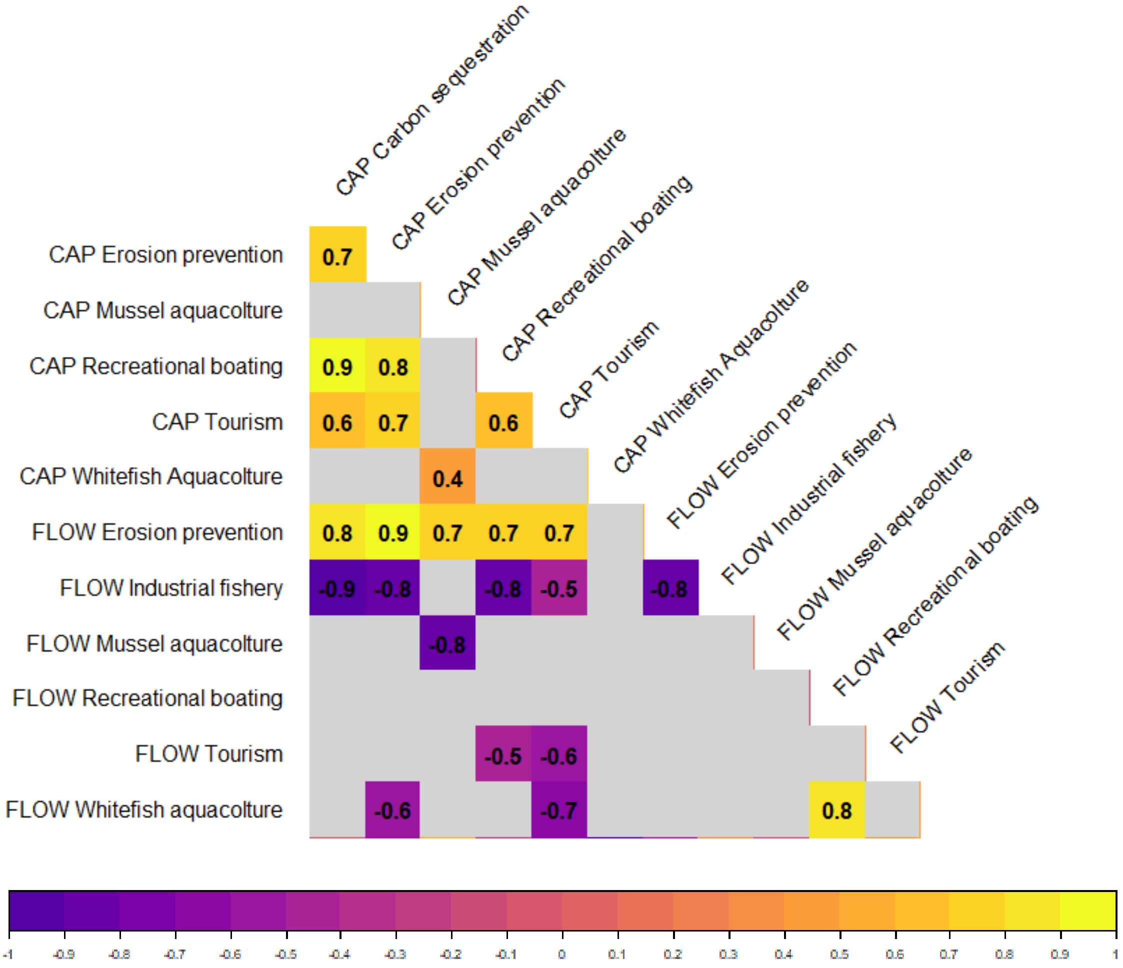


Figure 15: Correlogram representing spatial relationships among ESs.The picture represents coupling or decoupling (alpha=0.05) among ESs capacities and ESs flows

and the synergies and tradeoffs among ES flows and among -ES capacities. The figure was created by the authors using the R library: corrplot.

In the analysis of the sole four spatially complete ESs (tourism, recreational boating, carbon sequestration, and coastal erosion prevention potential) it was possible to identify five distinct groups across the study area (Fig. 16). A geographical gradient in ESs from South-West (low-low) to South-East (high-high) moving along the coast can be delineated, except for the subunit of Rimini. The five groups comprise, (1) subunits from Rovigo to Ascoli-Piceno, (2) subunits between the Venezia and Gorizia (including Rimini), (3) Trieste, (4) Istria, Primorje-Gorski Kotar, Lika-Senj, (5) subunits southern than Lika-Senj.

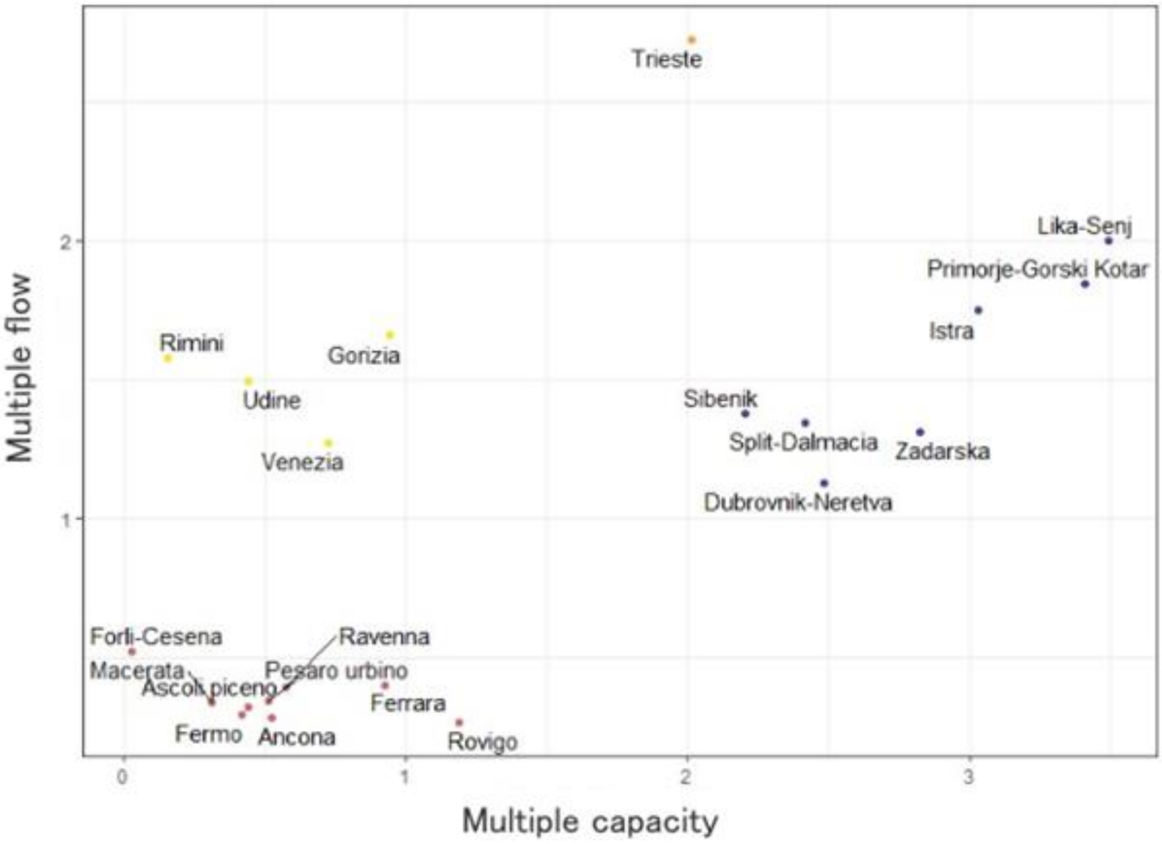


Figure 16: Relationship between the multiple ESs (tourism, recreational boating, carbon sequestration, and coastal erosion prevention potential) capacity and flow per subunit.

On one hand, moving from West to East the multiple ES capacity is increasingly made up of higher values of regulating services (carbon sequestration and coastal erosion prevention potential) (Fig. 17a). On the other hand, the multiple ESs flow does not show the same linear pattern having the "anomalous" subunits of Rimini, Venezia, Gorizia, and Udine, with high contributions by the two cultural services, tourism, and recreational boating, counterbalancing the low regulating service values (Fig. 17b).

Overall, there is a general coupling between capacities and flows over the geographical gradient finding a positive relationship between total ES capacity and flow. Except for the subunits of Trieste, Rimini, Venezia, Gorizia, and Udine for which a discussion of drivers is needed.

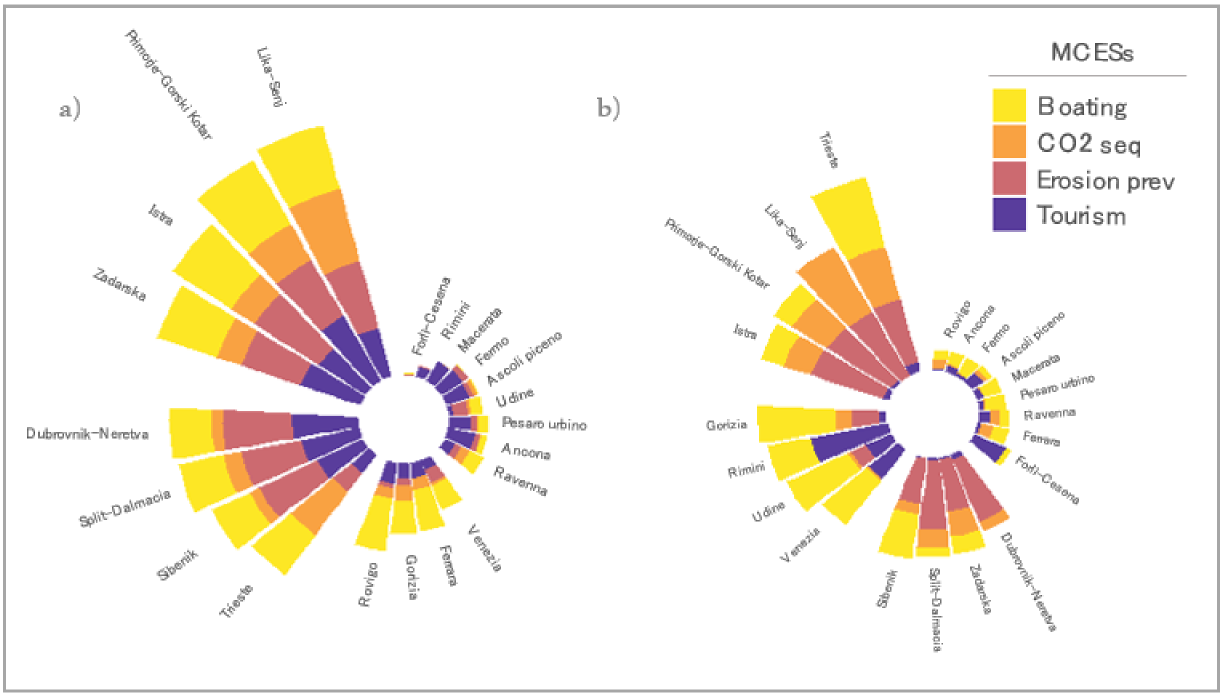


Figure 17: Identity and composition of multiple ES capacities (a) and flows (b) in the subunits in which ESs have been extracted. The subunits have been represented in the order of increasing multiple ES capacity (a) and increasing multiple ES flow (b)

Discussion

Multiple ES assessments can provide a logical and quantitative framework to link human well-being and activities to the coastal-marine system often overlooked in terms of human livelihood maintenance. The Northern-Central Adriatic coastal-marine system is a highly anthropized area (Zanetto & Soriani 1996, WTTC 2012). Industrial uses are massive from fishery to tourism (Pranovi et al., 2001, Jukic-Peladic et al., 2001, Coll et al., 2007) at the expense of environmental conservation (e.g., very few protected areas are present, covering about 11 % of the total Northern-Central Adriatic coastal-marine area). Furthermore, essential roles of regulating services (carbon sequestration and coastal erosion prevention potential), even if more difficult to quantify, will be lost if human society disregards Nature as the human life support system. Hence, to maintain socio-economic welfare related to Nature, sustainability roadmaps on how to properly use the ES set are vital. The present paper results showcased the distinction between capacity and flow (never made before for the study area) of ESs both in the coastal and in the marine realm (as highly suggested by the ICZM-2002/413/EC). Differently from what has been done by previous ES assessments in the area (Depellegrin et al., 2017; Manea et al., 2019; Farella et al., 2020) where coastal areas were not considered; the present inclusion of the coastal provinces surrounding the Adriatic basin drives the concept of local scale stewardship of the marine areas. We believe this is essential to ground and accomplish sustainable resource management together with the consideration of the present ES scenario in the area.

The unsustainability of single ES maximization

Our results found a coupling in regulating service capacities and flows (Fig. 15) as supported even by other authors (Burkhard et al., 2014). Accordingly, the flow of erosion prevention is coupled with almost all the other ES capacities, indicating that the conservation of Nature supporting this particular ES will likely lead to the co-benefits of the protection of other ESs. With regards to the cultural and provisioning services' capacity and flow, they are found to be often decoupled. Here, the commodification of Nature is driven by socioeconomic forces, creating strong tradeoffs (Schroter et al., 2018; Vergana et al., 2021). Hence, to define sustainable management strategies of provisioning and cultural service the inclusion of the economic, and social drivers has to be pursued. This inclusion avoids reductionism aiming at a version of sustainability at all levels, namely ecologic socio-economic sustainability. To better discuss the paradigm above some instances of different scenarios between capacity and flow revealed by the present study are presented. At the Northern-Central Adriatic basin scale, the provisioning ES industrial fishery is found to be decoupled with its and many of the other ES capacities (Fig. 15). The trawl fishing activity, which represents the main exploitation method in the area (Russo et al., 2020), is recognized as one of the most important stressors impacting the marine environment. It reduces both biodiversity and destroys the sea bottom (Jones 1992; Shephard et al., 2015; Pranovi et al., 2000) with the consequent reduction of the capacities of other ESs which rely on these natural features (Hopper et al., 2017). This finding shed light on the first tradeoff at the basin scale of the present assessment and the unsustainability of single-ES maximization which could prevent other services from being delivered. An analogous example is found in tourism along the Italian coast. At the transnational level, the tourism pressure registered in Italy (especially in summer) is higher than in Croatia regardless of the higher natural capacity in the latter area (Table 3)(Romano & Zullo 2014). This can potentially be explained by the economic will to build tourist infrastructures and devote the whole Italian stretch of the coast to almost solely tourism. However, this maximization of one ES has possibly led to the erosion of other ES capacities such as carbon sequestration and coastal erosion prevention potential, equally important for human well-being. An almost unique interest in tourism on the Italian side with the possible subsequent decrease of other ESs can warn Croatia, which is at present experiencing a fast increase in cultural services (Kozic & Mikulic, 2011; Croatian Bureau of Statistics, 2018). An extreme case of this maximization of a single ES comes from observing the smaller trans-provincial scale. Here, the Italian provinces from Rovigo to the southern Ascoli-Piceno have been subjected to enormous exploitation (Fig. 16) of Nature in the past leading to a decrease in ES flows which have been potentially dropped after the erosion of capacities (Fig. 17). Here neither socio-economic aspects nor other non-natural amenities or surrogates have helped the provinces to maintain single or multiple ES flows resulting in ESs poor provinces.

Overall, it is the relative proportions of different ES which determine the (un)sustainability of the observed management (Rova et al., 2019). Management focusing on single ES fails to capture the complexity of the system and can produce undesirable effects due to tradeoffs among ESs (Rova & Pranovi, 2017). Accordingly, the shortsighted use of single ES (maximization of one ES) will no longer guarantee the maintenance of the many other services equally important to human survival, such as regulating services (Cohen 1978; Bennet et al., 2009; Perini et al., 2017; Fennell & Cooper, 2020).

Artificial amenities to mimic the loss of Nature

Different strategies have been put in place by other Italian coastal provinces which have found a workaround through human management to counteract the loss of Nature and the related ES capacities. Few instances came with the high tourism flows reported in Rimini and Venice (Fig. 17b). These two provinces are for multiple ES capacity as poor as the provinces of the previous group but result in higher multiple ES flows (Fig. 17a,b). To comprehend the underneath phenomena, detachment from the rigorous ecological reasoning is highly needed in favor of the coastal provinces' social and economic assets (Costanza et al., 1998, Fisher et al., 2008, Gómez-Baggethun et al., 2010). The high ES multiple flows are given by the high cultural services contribution (tourism and recreational boating). Possible reasons for these high flows can be found in the attractiveness given by the non-natural compendium such as events (Rimini) and cultural heritage (Venice). In addition, tourism is possibly kept high because of the presence on the coast of natural surrogates (e.g., swimming pools) or other infrastructures (e.g., buildings with gardens or aquatic parks) that allow tourists to enjoy the outdoors, giving the impression of being immersed in Nature even when that is not the case. This led to unsustainable management of natural resources which should strongly be revised. The ESs which are maintained at present thanks to surrogates could drop in the future if the few Nature remained will be destroyed to build, for instance, other accommodation facilities.

Maintaining the supply of multiple services

Continuing to explore the trans-provincial scale, Trieste has the highest flow across the study area even if it has a lower multiple ES capacity in comparison with the Croatian set of counties (Fig. 17a,b). This can potentially be explained by the “hybrid” Nature of the province. Trieste represents both the natural and morphological coastal characteristics of Croatia and the Italian approach to tourism. This support once more, the importance to consider human influences in generating the ES flows (Potschin & Haines-Young, 2011; Rova et al., 2017) and including local-scale drivers’ dependency (Eastwood et al., 2016; Villamagna et al., 2013, de Groot et al., 2010).

Similar to Trieste, the high multiple ES flows of Udine e Gorizia are given by both cultural and regulating services. Trieste, Gorizia, and Udine in comparison with Rimini and Venice did not maximize the solely cultural services but conserved the regulating ones. It is straightforward, that when

Nature is well-conserved all the ES capacities are kept high, but the same is untrue for the ES flows for which other factors must be taken into account by managers, namely social and economic aspects. On the contrary, the highest flow would be present where the highest capacity is which is not what is here observed. Accordingly, the ability of an ecosystem to deliver multiple ESs does not solely need high capacity but even forms of capital other than natural (Fisher et al. 2009; Burkhard et al., 2014).

Best practices and management suggestions

Case-by-case reasoning, as done above, is imperative because of the high differences in the economic socio-ecologic drivers across the seascape (Eastwood et al., 2016; Villamagna et al., 2013, de Groot et al., 2010; Haines-Young & Potschin, 2010; Haines-Young & Potschin, 2011). Furthermore, each scale (e.g., basin-scale, transnational, trans-provincial) corresponds to different scenarios hence they all must be differently considered to take the right actions on the right grounds. Relationships among ESs highly depend on the scale of the study (Rodríguez-Loinaz et al., 2015; Zhang et al., 2020). Therefore, the coupling and decoupling between capacities and flows together with consideration of synergies and tradeoffs among ESs shed light on rights and responsibilities across space (Bennett et al., 2009; Deng et al., 2016; Meacham et al., 2016; Turkelboom et al., 2018; Vergara et al., 2021).

Marine and coastal management of the area should be strongly revised possibly considering the present analysis outcomes. For this and many other reasons, doing a multiple ES assessment plays a critical role in convincing stakeholder that the conservation of ecosystems are worthwhile. A new institutional arrangement for the inclusion of ES capacities and flows in the decision-making decisions process for the Adriatic marine and coastal area is highly needed.

A potential management strategy could be contextual and holistic planning in which the inherent complexities of the multiple ESs are recognized and included in an interdisciplinary view. Decision-makers at any scale should embrace quantitative and spatially explicit multiple ES assessments as a better-than-present decisional tool. This can show synergies and tradeoffs, for instance as shown by negative correlations in Fig.15, otherwise overlooked. Ecosystem based-management (EbM) and new policies should consider multiple ES assessments carried out by practitioners as compulsory for (i) minimizing ESs single-use, (ii) considering the spatial dependence between ES capacities and flows, (iii) including synergies and tradeoffs among ESs. This could define a better management strategy for conserving the multiple ESs provided by the coastal-marine system in the Adriatic area.

To conclude, elevated values of multiple ES flows must be correctly measured and not associated with areas where the mere high values of Nature are measured because socioeconomic drivers are as crucial as the environmental ones (Bennett et al., 2009; Foley, 2005; Raudsepp-Hearne et al., 2010a; Rodríguez et al., 2006). This highlights a concept of foremost importance; ES assessments must be always carried out because assigning areas with the highest natural compendium to the highest multiple ES flow can be strongly misleading.

Limitations and future work

Possible limitations of the present work dwell in the dependence of ES assessment on chosen indicators as already stressed in the literature (Heink et al., 2016; Van Oudenhoven et al., 2018). Accordingly, multiple ES assessments should be mapped with different indicators as well as at different scales and resolutions to have the least possible bias (Grêt-Regamey et al., 2014). For the present study, a new institutional framework must be created to embrace coastal and marine ESs to put in place suggested target management interventions (García-Nieto et al.,

2013) and to counteract the current mismanagement in place. Furthermore, in the future due to the rapid increase in populations and standards of living (Lamarque et al., 2011; Chan et al., 2012), the flow of several ESs is supposed to grow while the multiple coastal and marine ES capacity will be even more threatened by anthropogenic impacts and climate change (Pedrono et al., 2016). In this scenario not considering synergies and tradeoffs is a strong limitation because it can damage the long-term ability of Nature to provide ESs (Rosenberg et al., 2005, Peng et al., 2017, Farella et al., 2020). Finally, the real inclusion of multiple ES assessments in the Northern-Central Adriatic coastal-marine management should be tested empirically to ensure the relevance of the assessment (Schröter et al., 2012). Future implementations of the work could look forward including the assessment of different ecosystem functioning metrics (i.e., primary productivity, biogeochemical cycles, community trophodynamics, species life-history, and functional traits) for better exploring synergies and tradeoffs not solely dependent on ESs but also on other ecological features. This approach could lead to a better understanding of the dependency of coastal and marine ESs on ecosystem processes (ecosystem functions) and biodiversity (ecosystem structure).

Conclusion

The present assessment unraveled the spatial patterns of ES capacities and flows for the Northern-Central Adriatic coastal-marine system, essential to synthesize and plan the marine space as required by the European Maritime Spatial Planning (EU MSP), especially in the foreseen Blue Growth. It provides for the first time, an overview of the coupling and decoupling of ES capacities and flows together with synergies and tradeoffs among ESs. The presented analysis enabled us to highlight mismanagement never shown before, given the inclusion of regulating and provisioning services in the same assessment where cultural services are mapped.

We conclude the following, (i) Maximization of a single ES must be avoided for the preservation of other services equally important to human survival (ii) Surrogates of the natural compendium could prevent loss of single ES flow but not the maintenance of the whole ES set (iii) Extreme erosion of ES capacities put down multiple flows together resulting in a poor ES landscape where neither nature surrogates can help (iv) Both Italy and Croatia should better integrate the Science of ESs, including ESs synergies and tradeoffs in coastal and marine development plans with the aim of better delineating real sustainable management strategies.

Nature is indispensable for ESs and the socio-economic flows related to them, being at the basis of the ES concept (Costanza et al., 1997; Costanza et al., 1998). The Northern-Central Adriatic coastal-marine system-related economic flows could lower in the future if the few natural areas remaining will not be well conserved. Hence, the inclusion of Nature in the decision-making process in a view of the benefits of co-production is of paramount importance. Under the Ecosystem-based Management (EbM) concept (Christensen et al., 1996), and the 2030 Aichi targets (Convention of Biological Diversity (CBD)). Inter and transdisciplinary cooperation among ecological, social, and economic experts is the linchpin to matching sustainable governance of Nature and human well-being.

CHAPTER IV

FIRST EVIDENCE OF SPATIAL RELATIONSHIPS BETWEEN ECOSYSTEM FUNCTIONING AND SERVICES IN THE MARINE ENVIRONMENT

This Chapter has been pubblished as a scientific paper in the peer-reviewed journal "Journal of ecology and natural resources" (Impact Factor: 2.45). Bibliographic details: Basconi L., January 2023.

Abstract

The complexity of the marine system and the rate of anthropogenic impacts on ecosystem functioning demand a synthetic conceptual framework to organize the scientific knowledge needed to better conserve Nature and maintain ecosystem services supply. Currently, the most used conceptual framework is the cascade model which describes a tight positive link among biodiversity, ecosystem functioning and services. The present study explores the spatial relationship between ecosystem functioning and services discussing its heterogeneity. It warns the next generation of researchers on ecosystem services to quantify even ecosystem functioning in their spatial efforts, to better set real sustainable management strategies. The unified framework of ecosystem services and ecosystem functioning assessment will further the goal of protecting nature while humans use it.

Introduction

Since the pioneer paper of Costanza et al., 1997 ecosystem services (ESs), defined as the benefits that ecological systems provide to humans, are a widely used metric to measure Nature and its conservation status (e.g., Abunge et al., 2013; Veidemane et al., 2017; Nahuelhual et al., 2017; Manea et al., 2019; Friedrick et al., 2020; Hattam et al., 2021). Questioning the correctness of this metric as proxy of natural systems, several studies have focussed on the spatial relationship between biodiversity and ESs (Worm et al., 2006; Quijas & Balvanera,2010; Mace et al., 2012; Schneiders et al., 2012; Eastwood et al., 2016; Winter et al., 2018). Scattered and poor are the consideration about the spatial relationship between ecosystem functioning (EF) and services (Lindegren et al., 2017; Birkhofer et al., 2018). In ESs assessments EF measurement is often shaded using indirect metrics of it such as ecosystem structural components (i.e., habitats and their characteristics) (e.g., De Pellegrin et al., 2017; Elise et al., 2019). Still, works that empirically demonstrated whether and how ESs spatially relate with EF indicators are lacking**.** Theoretically, biodiversity and EF underpins ESs supply as explained by one of the most known theoretical frameworks: the cascade model (Haines-Young & Potschin, 2010; de Groot et al., 2010; Potschin & Haines-Young, 2011). Notwithstanding the wealth of studies using the concept of the ES cascade and, its usefulness for making the ES concept more accessible to the non-science community (Potschin-Young et al., 2018) some authors (Balvanera et al., 2006; Mumby et al., 2008; Peterson et al., 2010; Cardinale et al., 2012) have already pointed out the importance of explicitly presenting ESs as discreet and incomplete aspects of ecosystem functions to ensure the complementary valuation of EF indicator and biodiversity. Empirically, Naidoo et al., 2008 proved globally that regions selected to maximize biodiversity provide no more ecosystem services than regions chosen randomly. Furthermore, Costanza et al., (2017) highlighted that the conceptualization of ES through the cascade model is for some aspects an oversimplification, as it does not capture the complex and dynamic connections occurring between the ecosystem structures (i.e., biodiversity), EF, and ESs.

The main misconception of the ES cascade framework lies in the anthropocentric-defined concept of EF. In the cascade model, the functioning is described as the capacity or capability of the ecosystem to do something potentially useful to people (Costanza et al., 1997; Costanza et al., 1998; Daily 1997; de Groot et al., 2002; de Groot et al., 2010; Haines-Young & Potschin, 2010; Potschin & Haines-Young, 2011). However, as Jax (2005) stated, the term ‘function’ can mean several other things in ecology. It can mean capability, but it is often used more generally to refer to processes that operate within an ecosystem (e.g., nutrient cycling, the flow of energy and matter through predation, top-down control), and especially exists even when a human beneficiary is absent.

Accordingly, Wallace (2007) and many other ecologists describe EF as functions serving Nature *per se*, without being necessarily useful to people. Hence, these functional aspects may even not be positively directly related to human benefits, as the ES cascade instead describes.

Overall, to understand any system, empirical knowledge is needed (Bertness et al., 2014) as even the 'scientific method' simple states, in particular when the system described is complex such as the natural ecosystems. Unless we went back in time when for evaluating marine ecosystem status studying focal species or one indicator alone was acceptable, system-level phenomena should be the way forward in Ecosystem-based Management (EbM) (Cohen 2009; Rombouts et al., 2013; Klain et al., 2014; Tam et al., 2017; UNEP, 2014).

The present article seeks a better understanding of the spatial relationships between marine EF and ES indicators using the Adriatic Sea as the study case. To accomplish it I have searched for an evidence-based direct and positive spatial relationship among different EF and ESs indicators as the cascade model theoretically supports. Given that both EF and ESs quantifications are highly dependent on the indicator chosen (Jax 2005; Heink et al., 2016; Jax et al., 2018***;*** Van Oudenhoven et al., 2018) to reduce this possible biased the present article considers different ES assessments and EF indicators.

Materials and Methods

The quantification of spatial overlaps or correlations between five EF indicators (i.e., bottom-up forces, top-down forces, biodiversity, and trophodynamics) and 11 ES indicators (i.e., supporting, provisioning, cultural, regulating) for the same area of study (i.e., the Adriatic Sea) and, where possible for the same year (2018/2019) have been gathered (Table 4). First, these indicators have been extracted on a common grid of 20km horizontal resolution (i.e., OSMOSE model grid as the lowest resolution of the gathered data) to make them comparable. Second, all the indicators considered in the present analysis have been normalized through min-max normalization because of the different units of measurement. Eventually, pairwise relationships between all ES and EF indicators have been unrevealed by the average correlation between Kendal, Spearman and Pearson methods (alpha=0.05). Normality is not an assumption and transformation (square root or log) did not improve skewness values therefore untransformed data were used.

Ecosystem functioning indicators

Spatial explicit EF indicators have been gathered from different sources (Table 4). Pelagic bottom-up control has been extracted from the POLCOM-ERSEM model as the concentration of Chlorophyll-A (mg/cm3), using it as a proxy of primary productivity. Top-down control has been measured by the biomass of the 100 high trophic-level species modeled by OSMOSE-MED (Moullec et al., 2019). Transfer of energy and matter across trophic levels namely trophodynamics has been measured by Efficiency Cumulated Indicator (ECI) and, Biomass Residency Time (BRT) (*sensu* Maureaud et al., 2017). Biodiversity even if more measure of structure is complementary to EF, therefore, it has been included here. Biodiversity has been calculated after a climate suitability approach based on the 579 fish species endemic or if aliens established in the Mediterranean Sea (Basconi et al., *under review* (CHAPTER II of the present thesis).

Ecosystem services indicators

Spatial explicit ES indicators have been extracted from three different assessments carried out in the Adriatic Sea (De Pellegrin et al., 2017; Manea et al., 2019; Basconi et al., *in press,* CHAPTER III of the present thesis). The whole set of assessments approached with snapshot representations of Adriatic marine ESs, using among them different approaches, data sources, and analysis (reported briefly in Table 4). Used ESs have been, three maps of supporting ESs specifically related to the marine domains (i.e., surface, water column, bottom) (Manea et al., 2019), a general ability of the Adriatic marine area to deliver ESs (De Pellegrin et al., 2019), and two provisioning services (i.e, mussel aquaculture, fishery), two cultural services (i.e., tourism and recreational boating), and one regulating services (i.e., carbon sequestration). Furthermore, the disservice of eutrophication measured by the TRIX regional index was included.

EF or ES	Indicator	Description	Unit of measurement	Source	Reference
Primary productivity	Total chlorophyll-A	Chlorophyll-A is the as proxy of primary production	Kg/m3	POLCOM-ERSEM model (~ 12Km of horizontal resolution)	CMEMS
Top Down Control	Biomass of 100 High Trophic Level species	Biomass of species exerting a top-down control on the community	Tons/pixel	OSMOSE-MED (~ 20Km horizontal resolution)	Moullec et al., 2019
Trophodynamics	Efficiency Cumulated Indicator (ECI)	The process linked with the flow of matter and energy across trophic levels in an ecosystem. In particular ECI measures how efficent is the trophic transfer	Unitless	Calculated with biological variables of the species present by climate suitability in the Adriatic Sea (~ 7Km horizontal resolution)	Basconi et al., *under review (Chapter II of the present thesis)*
Biodiversity	Species richness	Presence/Absence over the grid was calculated with climate suitability models for 579 fish species	Species richness	Modeled with Temperature and Salinity (BIOMOD2) (~ 7Km horizontal resolution)	Basconi et al., *under review (Chapter II of the present thesis)*
Trophodynamics	Biomass Residency Time (BRT)	The process linked with the flow of matter and energy across trophic levels in an ecosystem. In particular BRT measures how efficent is the transfer of matter and energy.	Unitless	Calculated with biological variables of the species present by climate suitability in the Adriatic Sea (~ 7Km horizontal resolution)	Basconi et al., *under review (Chapter II of the present thesis)*

Carbon sequestration capacity & flow	Carbon sequestered by the marine habitats	Important habitats for the sequestration of carbon in the Northern-Central Adriatic Sea have been mapped. The rate of sequestration per each habitat was retrieved from the literature. The habitat areas were multiplied by their carbon sequestration and estimate per each habitat have been summed together	Kg CO2/ Km2/ Year	Calculated for Basconi et al., *in press*	Basconi et al., 2023 *(Chapter III of the present thesis)*
Tourism capacity	Attractiveness potential	Questionnaires directed to seaside tourists have been used to give weight to different features of the coastal-marine environment included in the composite indicator of attractiveness potential	Unitless	Calculated for Basconi et al., *in press*	Basconi et al., 2023 *(Chapter III of the present thesis)*
Recreational boating capacity	Attractiveness potential	Questionnaires directed to recreational boaters have been used to give weight to different features of the coastal-marine environment included in the composite indicator of attractiveness potential	Unitless	Calculated for Basconi et al., *in press*	Basconi et al., 2023 *(Chapter III of the present thesis)*
Recreational boating flow	Usage of the marine space by recreational boaters	The number of leisure boats by maritime compartment was retrieved from the Coast Guard registry ('Capitaneria di Porto') and the Croatian Bureau of Statistics then divided in the marinas along both coastal sides. Questionnaires were used to set the number of both trips.	N° of boat trip/ Km2/ Year	Calculated for Basconi et al., *in press*	Basconi et al., 2023 *(Chapter III of the present thesis)*

Mussel aquaculture capacity	Tons of mussel producible/ Km2/ Year	Assessed through the eco-physiological model (R package RAC, Baldan et al. 2018) calibrated for *Mytilus galloprovincialis*	Tons of mussel producible/ Km2/ Year	Calculated for Basconi et al., *in press*	Basconi et al., 2023 *(Chapter III of the present thesis)*
Industrial fishery flow	Measurement of fishing efforts	Spatial explicit fishing effort was estimated by using the Automatic Identification System (AIS)	Trawls fishing effort/km2/year	Extracted from Russo et al., 2020	Basconi et al., 2023 *(Chapter III of the present thesis)*
ESs total capacity	The capacity of marine habitats to provide marine ecosystem services	ESs have been measured from EUNIS seabed habitats (EmodNet website)	Unitless	Modeled by a matrix approach	De Pellegrin et al., 2017
Supporting ESs sea bottom	Quantification of the supporting ESs at the sea bottom	ESs delivery was mapped starting from spatial explicit marine components specific to the sea bottom	Unitless	Modeled by richness and hot spot analysis	Manea et al., 2019
Supporting ESs water column	Quantification of the supporting ESs in the water column	ESs delivery was mapped starting from spatial explicit marine components specific in the water column	Unitless	Modeled by richness and hot spot analysis	Manea et al., 2019
Supporting ESs sea surface	Quantification of the supporting ESs at the surface	ESs delivery was mapped starting from spatial explicit marine components specific to the surface	Unitless	Modeled by richness and hot spot analysis	Manea et al., 2019
Eutrophication	TRIX	Trophic index (considering dissolved oxygen, ChlA, total Phosphorous and Nitrogen)	Unitless	TRIX modeled for the Adriatic (~ 2Km)	ARPA regional agency (IT); Fiori

		et al., 2016

Table 4. Indicators of Ecosystem Functioning (EF) and Ecosystem Services (ESs) measured or modeled in the Adriatic Sea, and gathered in the spatial explicit analysis.

Results

Many are the direct or indirect theoretical connections found by the author through expert opinion (Fig. 18) among EF and ESs indicators. However, they do not seem to subsist in the spatial analysis results (Fig. 19).

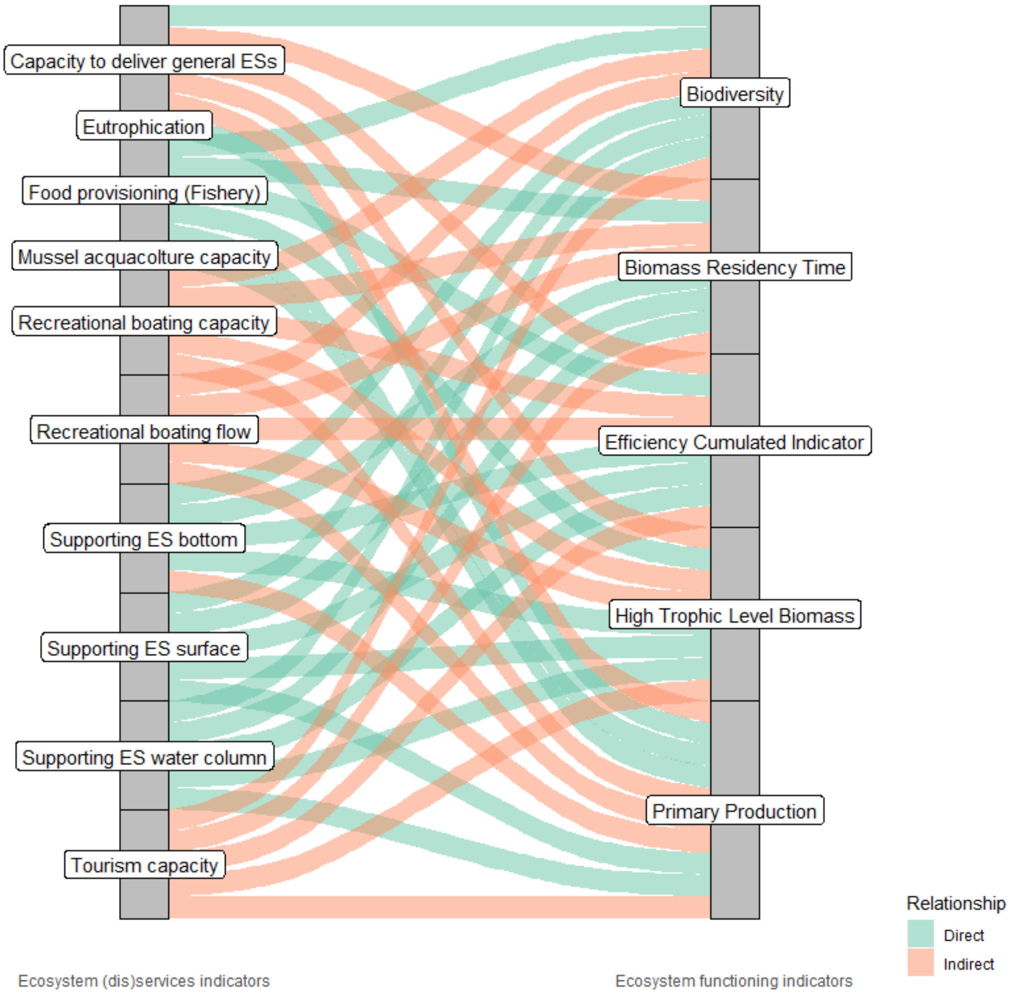


Figure 18: Theoretical links between ecosystem services and functioning indicators.

Overall, there is not a unique strictly positive or strictly negative trend describing the spatial dependency of ESs on EF indicators in the Adriatic Sea (Fig. 19). To effectively find the bigger picture of the spatial relationship between ESs and EF indicators solely strong correlations (i.e., higher than 0.5 or lower than – 0.5) have been reported as results. Among them, BRT negatively links with primary productivity (corr= - 0.71) and accordingly with eutrophication (corr= - 0.74). On the contrary, ECI positively correlates with primary production (corr= 0.51) and eutrophication (corr= 0.63). ECI negatively correlates with the capacity of mussel aquaculture (corr= - 0.73). The disservice of eutrophicated water is negatively correlated with both capacities of tourism (corr= - 0.70) and mussel aquaculture (corr= - 0.57) while possibly positively correlated (corr= 0.55) with the provisioning service of the fishery. There is even a correlation among services with positive correlations between supporting ESs in the water column and supporting ESs at the surface (corr= 0.53).

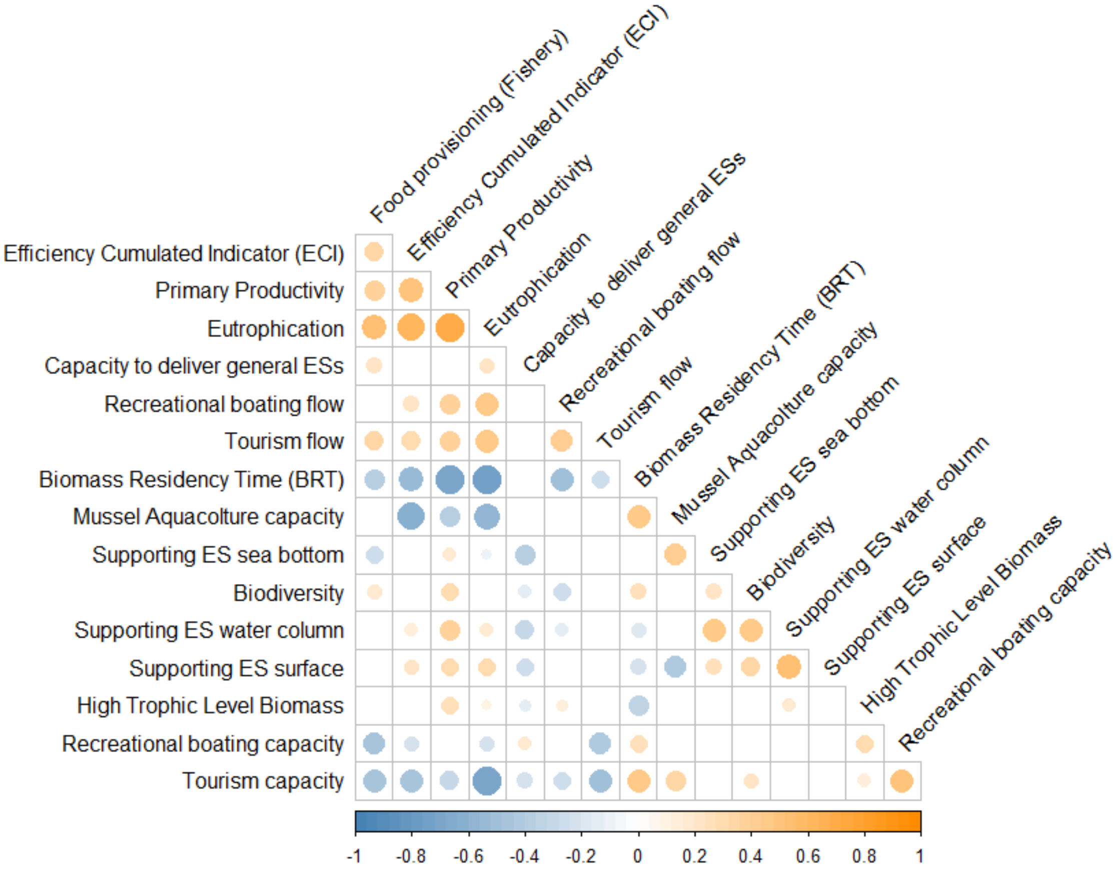


Figure 19: Spatial relationships (alpha=0.05) between eleven ecosystem services and five ecosystem functioning indicators, measured in the Adriatic sea.

Discussion

Rarely ecosystem processes and functions been empirically linked to human well-being and activities. According to the cascade framework (Haines-Young and Potschin, 2010; Potschin and Haines-Young, 2011) high values of ecosystem functioning correspond to high values of ecosystem services. Theoretically, without ecosystems that function well, ESs are not delivered but still, the spatial relationship can be highly non-linear or even disappear in certain contexts. Notwithstanding the static linkages made for crafting Fig. 18, the correlations between ESs and EF are highly dynamic and context-dependent as the marine system is. There are examples in which these two facets of ecology could exist even in opposite trends in the same spatial unit. For instance, in the study area considered for the present study (i.e., the Adriatic Sea) together with other authors (Basconi et al., *in press*, CHAPTER III *of this thesis*) a survey was prepared for users of the Adriatic marine space. 250 questionnaires were answered by recreational boaters and around half of them was showing a low interest in the health of the marine ecosystem. In fact, to the question "How many boat trips will you do in detrimental water quality?" (explaining what ecologically is meant for it) 50% of respondents replied with an "unchanged number of trips".

A second instance, include the present and future condition of Ocean warming. In fact, the Mediterranean has been projected to tropicalize over time (Bianchi & Morri, 2003; Bianchi 2007). The opening of the Suez Canal is and will further allow species to migrate to the Mediterranean Sea. Whether the Mediterranean Sea assumes tropical abiotic conditions, tropical species will settle in the empty ecological niches (e.g., see species reported by CIESM Atlas, Moschella et al., 2008). Besides invasive species that could disrupt native ecosystems, the species adaptable to new climate change-driven conditions are the one that functions in those peculiar environmental conditions. In other words, ecological communities genetically adapt (it is the natural selection that maintain the most adaptable species) to physicochemical conditions as the "red queen hypothesis" states (Van Vallen, 1973). The real caveat is how much the market will be adaptable to these new species. It has been already proven that alien or invasive species could be a problem for the Blue Economy in the Mediterranean Sea (Yildirim & Kaplan 2022).

Moreover, straightforward examples deal with the 'anthropization' of ESs which are linked to nature (as described by ES definition itself, Costanza et al., 1997) but for which socio-economic drivers are strong enough to maintain the services even when ecosystems are detrimental conditions.

Examples in the coastal and marine ecosystem are coastal erosion prevention by artificial protections as well as tourism by the creation of nature surrogates to maintain the delivery of ESs (Fisher et al., 2008, Gómez-Baggethun et al., 2010; Basconi et al., 2023). These are examples of ESs that might be largely independent of the level of functioning of ecosystems and therefore remain high (at least in the short term) even when EF is low. These instances are important examples of spatial detachment between ecosystem functioning and certain services (i.e., cultural service – recreational boating and tourism, provisioning service – fishery supply, regulating service – coastal erosion prevention) which should be always kept in mind by ES assessment practitioners.

Accordingly, the results of the present paper do not show any evident (i.e., correlation coefficent>0.5 or < - 0.5) spatial correlation between the used ESs and EF indicators. Neither between ecosystem functioning indicators and the ES capacities (e.g., tourism, recreational boating and supporting services by Manea et al., 2019) where it was expected by definition of ES capacity (*sensu* Villamagna et al., 2013) Whether they could the relationship will be non-linear and therefore not detected by the coefficients of correlation used. The only presented evident correlation between EF and ESs indicators is the Efficiency Cumulated Indicator and the capacity of the ecosystem to produce mussels in aquaculture farming but it is negative. All the other relevant correlations are among EF indicators (e.g., between primary productivity and Efficiency Cumulated Indicator). From an ecosystem functioning point of view, Biomass Residency Time shows the opposite correlation of the Efficiency Cumulated Indicator. The former represents the inverse of trophic transfer speed and it is stated in the literature (Gascuel et a., 2008; Maureaud et al., 2017; Gascuel et al., 2011) that ecosystem functions well when both trophic transfer efficiency and speed are high. In other words, this is the reason behind the opposite correlations between the two in a snapshot analysis such as the one reported in the present paper.

Some correlations shown in Fig. 19 serve to possibly prove that the analysis is sensitive in representing ecologically known spatial patterns in the marine realm. For instance, the positive relationship between primary productivity (extracted from a biogeochemical model) and eutrophication (coastally measured through the TRIX indicators by ARPAs and modeled at the whole basin scale). It could prove that the present analysis can show spatial relationships whether present.

The reported results challenge the assumption that a high level of biodiversity and a high rate of ecosystem functioning will always promote high ESs or that these high values will enhance ES supply in the same space. Other research carried out globally (Lindegren et al., 2018; Swift et al., 2004) and in the terrestrial realm (Egoh et al., 2009; Birkhofer et al., 2018) has proven EF and ESs indicators to be spatially uncorrelated. The present paper's results support the explicit use of ESs as discreet and incomplete aspects of EF enables the social and political changes required to ensure the use ESs metric in an ecologically meaningful manner (Peterson et al., 2010). A critical step in sustainable strategies is to not only agree on indicators that are compelling, intuitive, understandable, and defensible to all stakeholders but also capture the whole complexity of the system. Measuring critical states and processes that underlie the complexity of the ecosystem dynamics (i.e., Functioning—energy flows, Resilience—ability to recover from perturbation, Structure—species organization) is essential to understand the status of conservation or management. An agreement exists about the need to move towards more holistic but not simplistic indicators that recognize the full array of interactions within an ecosystem while measuring ecosystem services (Link 2005; Devictor et al., 2010; Schulp et al., 2012; Mayer et al., 2021). This is particularly true in a fast-changing benchmark as in the climate change era, in an already multiple stressors scenarios impacting the marine biodiversity and EF (Bellwood & Meyer 2004; Lasram et al., 2010; Rossi et al., 2017; Newbold et al., 2020). Whilst the spatial relationships between EF and ESs should be further confirmed with studies in other marine areas as well as in long-term studies, a precautionary approach to ecosystem conservation would seem prudent in the meanwhile including EF metrics in multiple ESs assessments.

Conclusion

Empirical knowledge demonstrating the spatial relationships among ecosystem functioning and services is rare. The current study presents for the first time in the marine realm a lack of spatial coherence between ecosystem functioning and services paving the road for a new coming branch of research seeking a better understanding between EF and ESs spatial relationships.

The present results suggest an urgent need to fuel interdisciplinary science bridging together the different metrics of (i) ecosystem structure (e.g., species richness, abundance, biomass), (ii) ecosystem functioning (e.g., primary productivity, trophodynamics, high trophic level top-down control and, (iii) ecosystem services (e.g., biophysical measurements, quantitative measurements, stakeholder preferences). Destructuring complex system into complicated ones, avoiding the pitfall of oversimplification, is the way forward for both natural resource management and conservation.

CHAPTER V

GENERAL CONCLUSIONS

The present thesis explored some of the interfaces or shades of ecology often overlooked in modern studies. Usually, the study of climate change impacts focuses on biodiversity changes, alien species establishments, and habitat or species biomass changes. Seldom academic studies, especially at the regional scale, concentrate on climate change impacts on ecosystem processes which altogether constitute the functioning of ecosystems. This is possibly due to the ease of measuring biodiversity (i.e., number of species) while ecosystem functioning is multi-facets (e.g., primary productivity, trophodynamics, nutrient recycling) and needs indicators to quantify these ecological processes otherwise difficult to measure in the field. Notwithstanding the added complexity of quantifying ecosystem functioning instead of biodiversity, measuring its changes will be particularly relevant in the context of climate change, especially in highly biodiverse ecosystems such as the Mediterranean Sea. In fact, with climate change empty and/or novel ecological niches will appear and a change in the biotic components will be unavoidable. However, is the functioning of the ecosystem that should be maintained and perhaps even helped with human management. Whether we only focus on biodiversity impacts, we risk overestimating climate change effects on the Mediterranean marine ecosystem (*see* 1.4.1 Motivation behind the research, CHAPTER I, Fig. 8).

For this reason, the present thesis showcases the first attempt to project how climate change will mutate ecosystem functioning in the Mediterranean Sea using as a proxy of the ecological community, 579 species of fish. Eventually, CHAPTER II reports a basin-scale decrease in both the speed and efficiency of the trophic transfer due to climate change by the end of the 21st century. These findings unrevealed areas in the Mediterranean Sea of possible resiliency, where ecosystem functioning indicators even increase by the end of the 21st century under climate change scenarios. In other words, whether climate change impacts on biodiversity could be depicted as a general loss of endemic in favor of alien species, the functioning of the whole ecosystems shows even some spots of resiliency.

Heterogeneity in ecology has always been seen as positive because it allows different responses from Nature and diverse species to survive in different contexts (Origin of the species, Darwin). The role of the areas projected to be more efficient and faster in their ecosystem trophic transfers (i.e., higher in ecosystem functioning) can have the role of spillover for the nearby areas even in a future shaped by climate change. Whether the complete responses of ecosystems are not inclusively studied, the precautionary approach suggested for climate change adaptation could fail, as we will not be able to center the objective of protecting Nature not in its stillness but in its dynamic equilibrium.

The second part of the thesis focused on including another layer of complexity in the ahead-described picture. It includes the use of Nature by humans in the context of socio-ecological systems through the use of the ecosystem services metric. A better understanding of how ecosystem services supply respond to changes in ecosystem functioning due to climate change is vital to maintaining human well-being in the long term. Nonetheless, before inferring how the ecosystem services supply will be impacted by climate change following the future changes in ecosystem functioning, there is a need to know how these two facets of ecology are spatially distributed and linked. Theoretically, a high level of ecosystem functioning should result in a high level of ecosystem services and vice versa (*see* 1.2 Ecosystem services, CHAPTER I, Fig. 2). Although, the latter exists if and only a human beneficiary is present. On the other hand, ecosystem functioning exists for Nature per se even when the human beneficiary is absent. Using a highly anthropized area such as the Adriatic Sea the present thesis reported an absence of spatial coherence between marine ecosystem functioning and service indicators. Notwithstanding the possibilities of still having a non-linear link between ecosystem functioning and services apparently, the picture depicted theoretically with the ecosystem services cascade needs to be implemented, becoming more flexible to case-by-case considerations.

Management and conservation suggestions of the present thesis support a shift toward ecosystem functioning measures in a changing climate more than the historical biodiversity monitoring, as other scholars have already proposed. Meanwhile, the spatial linkages between ecosystem functioning and services need further research to be described empirically and not just theoretically. Eventually, the present thesis warns about reductionism and oversimplification of complex systems, such as marine ecosystems, from both the functioning and the socio-economic perspectives through ecosystem services. Further research is vital to better understand the novelties reported in the present thesis and an interdisciplinary approach is the way forward to discover what is still hidden in the complexity and wonder of marine ecosystems.

Appendix A

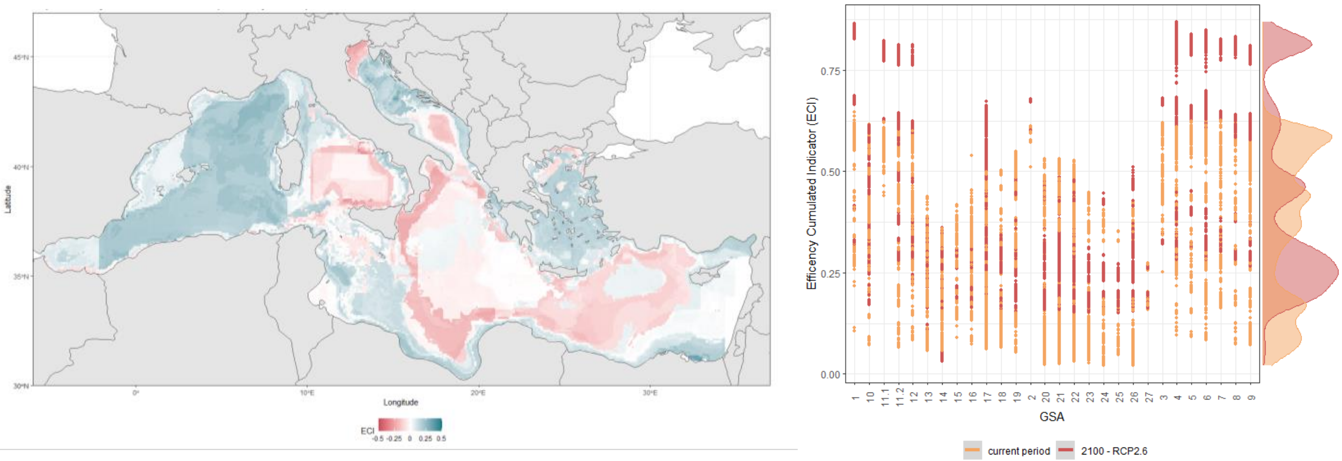


Figure 20: Efficiency Cumulated Indicator (ECI) under ssp21-RCP 2.6 anomalies compared to the current period (reference year: 2018)

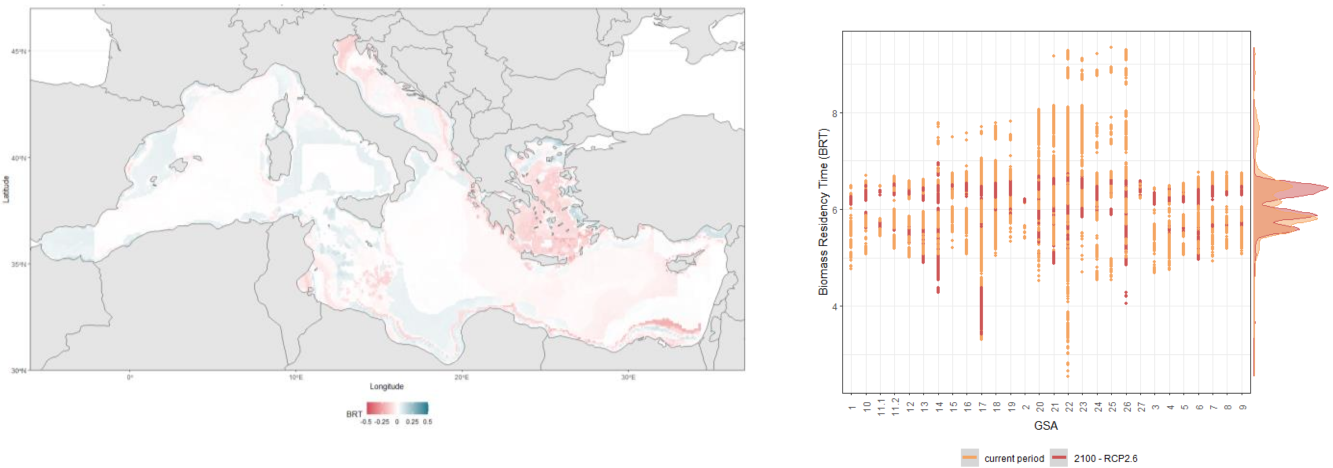


Figure 21: Biomass Residency Time (BRT) under ssp21-RCP 2.6 anomalies compared to the current period (reference year: 2018)

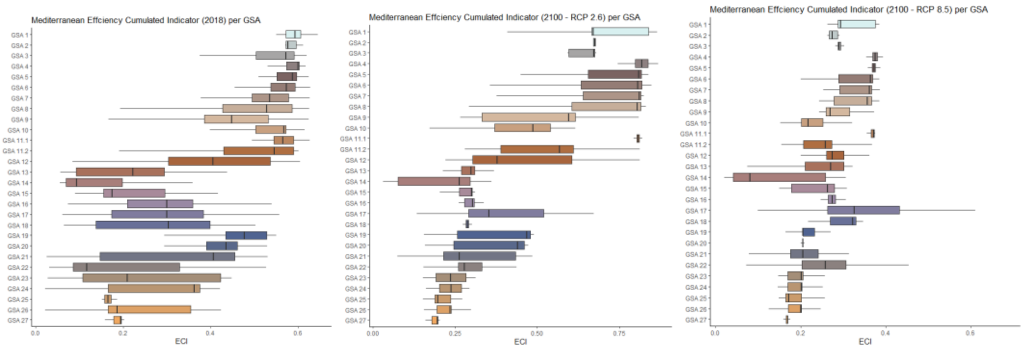


Figure 22: Efficiency Cumulated Indicator (ECI) per Geographical SubAreas (GSA) in current period, ssp1-RCP 2.6 and ssp5-RCP 8.5, respectively.

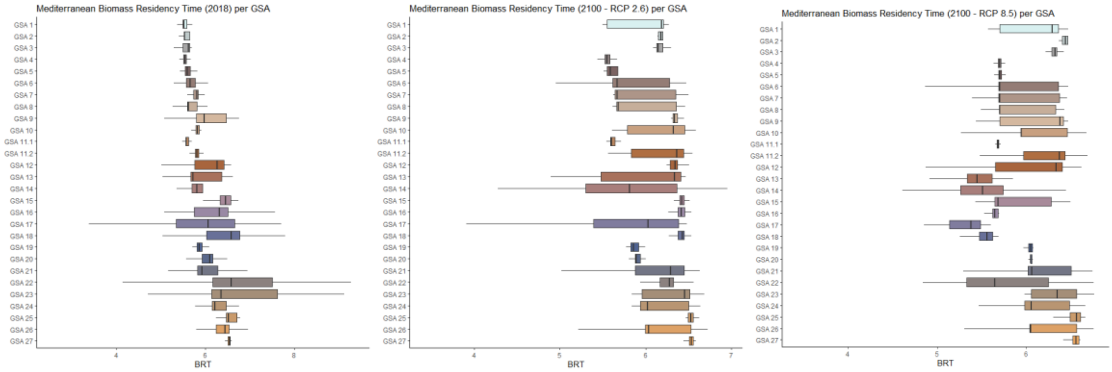


Figure 23: Biomass Residency Time (BRT) per GSA per Geographical SubAreas (GSA) in current period, ssp1-RCP 2.6 and ssp5-RCP 8.5, respectively.

Appendix B

In the following pages the workflow followed to spatially measure each ecosystem services (ESs) capacity and flow in the Northern-Central Adriatic coastal-marine system done for the Chapter III of the present thesis is reported.

Tourism capacity

A total of around 424 questionnaires directed to seaside tourists have been responded in the 2019 summer along the coasts of the study area. The questionnaires had the aim to (i) understand if tourists of this highly anthropized area considered Nature, among other characteristics of the place in their destination selection decision-making process (ii) to give weight to different features of the coastal-marine environment thus obtaining a composite indicator. To respond to the former question users have been asked to give importance to different destination features (i.e., cultural heritage, natural environment, local food and wine, tour operator offers, and events) rating them among, very important, moderately important, slightly important, not important. These have been quantified from 0 to 3. In Fig. 1 are summarized the results of the former question. The parallel coordinate plot (Fig. 1) shows heterogeneous responses for almost all the characteristics besides for "natural environment" where nodes in the chart are more concentrated in the class "moderately important" and "very important" (in the black rectangle). This makes meaningful to consider tourism in our ES assessment, whether people did not give importance to Nature of the area in their recreational activities then considering tourism as an ES would be wrong.

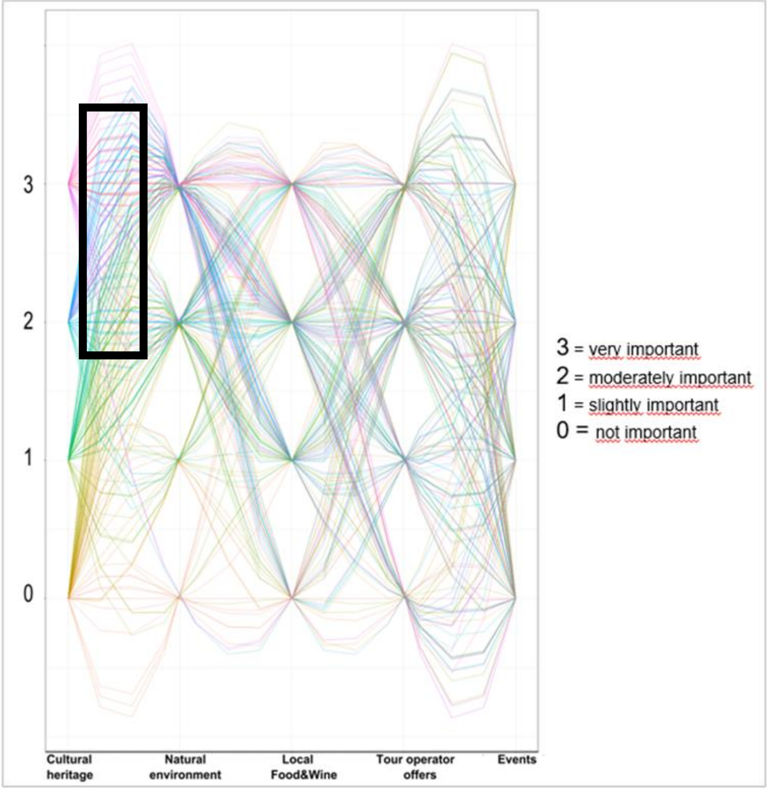


Figure 24: Summarised questionnaire responses of the around 424 seaside tourists of the study area collected in the summer of 2019. Response for "natural environment" is highly clustered on the value giving the highest importance (*black rectangle*).

The latter question was more precise: “How much importance have you given to the following characteristics of the natural environment when you decided to visit this area?” To respond to this question users have been asked to give importance to different natural features (i.e., water quality, protected area, terrestrial habitats, marine habitats, and cetaceans) rating among: very important, moderately important, slightly important, not important. These have been quantified from 0 to 3 and normalized from 0 to 1 (Table 5). They were included as weight (Wn) in the following linear model:

Natural attractiveness potential for tourism = W1*water quality + W2*protected areas + W3*terrestrial coastal habitat + W4*marine submerged habitat + W5*cetacens

MCES (ecosystem services)	Environmental features	Mean qualitative weight	Mean quantitative weight	Normalized weights value (W_n)
Tourism	Water quality	Moderately/very important	2.3	0.77
	Protected areas	Slighly/moderately important	1.7	0.56
	Terrestrial coastal habitats	Slighly/moderately important	1.9	0.65
	Marine submerged habitats	Slighly/moderately important	1.5	0.49
	Cetaceans	Not/Slightly important	0.6	0.22

Table 5. Values of different environmental features considered in the assessment of tourism capacity.

Each of these environmental features has been mined as spatial data from different sources (Table 2), weighted in the linear model and mapped (Fig. 25). Almost all the features considered had a categorical value of 0/1 (presence/absence). Exceptions are made for the marine habitat where the data came from the suitability model for *Posidonia oceanica* which results were treated using a 0.5 cut-off, to obtain presence/absence. Besides, the TRIX (measuring water quality) had four values ranging in a 0-1 scale, where 0=bad, 0.33=mediocre, 0.66=good, and 1=elevate trophic state, according to the reference values from Rinaldi & Giovanardi (2011).

Factor of environmental attractiveness	Mapping method, data source, and unit of measurement
Water quality	Trophic Index (TRIX, Vollenweider et al., 1998) modelled for the Adriatic Sea by Fiori et al., (2016)
Presence of natural terrestrial habitats	The proportion of natural habitats within segments of the coast of about 10 km long and 10 km wide. Natural habitats have been extracted from Corine Land Cover 2018 (CLC [2]), corresponding the following land-cover types: all forest types (CLC code 31*); all scrub and/or herbaceous vegetation associations (CLC code 32*); among open spaces with little or no vegetation (CLC code 33*), the classes beach dunes, and sand, and sparsely vegetated areas (CLC codes 331 and 332 respectively); all wetlands, both inland, and marine (CLC code 4**) (categorical P/A, adimentional)
Presence of marine habitats and fauna	Suitability model derived distribution of *Posidonia oceanica* was retrived from EMODnet [3] (Scardi et al., 2013). This species has been chosen to represent this factor of attractiveness because it constitutes one of the most important shallow habitats in the Mediterranean Sea, associated with high biodiversity and high environmental quality (categorical P/A, adimentional)

Opportunity to watch cetaceans	Distribution of bottlenose dolphins, one of the most common species of cetaceans in the study area, based on aerial surveys (Fortuna et al., 2018) (categorical P/A, adimentional)
Presence of protected or valuable natural areas	Marine protected areas, Ramsar sites and Natura2000 sites, retrieved from MAPAMed[4], and a marine site of community importance currently being established for the protection of dolphins and sea turtles, facing the coast of Veneto and Emilia Romagna[5] (categorical P/A, adimentional)

---

[1] Copernicus Land Monitoring Service: https://land.copernicus.eu/pan-european/corine-land-cover

[2] EMODnet (European Marine Observation and Data Network): https://www.emodnet.eu/emodnet-maps-catalogue

[3] MArine Protected Areas in the MEDiterranean: https://www.mapamed.org/

[4] https://ambiente.regione.emilia-romagna.it/it/notizie/attualita/2019/febbraio/accordo-tra-le-tre-regioni-del-distretto-alto-adriatico-per-unarea-marina-a-tutela-di-delfini-e-tartarughe

Table 6. Environmental features, mapping methods, data sources, and unit of measurements considered in the calculation of natural attractiveness potential for the tourism capacity.

Figure 25: Normalized capacity of the ESs tourism corresponding to the spatial explicit indicators for attractiveness potential of the Northern-Central Adriatic system (1x1Km resolution).

Tourism flow

The ES tourism flow has been measured as follows. The indicator used to represent the flow of tourism is the number of visitors/km2/year in the coastal area. This indicator has been quantified using official statistics of tourist arrivals (n° of visitors staying overnight) relative to the year 2018. Concerning the Italian coast, data at the municipality level were retrieved from the Italian National Institute of Statistics (ISTAT), and include all the coastal municipalities in the regions Friuli-Venezia-Giulia, Veneto, Emilia-Romagna, and Marche (Table 8). Concerning the Croatian coast, data at the county level was retrieved from the Croatian Bureau of Statistics and includes all the coastal counties of Adriatic Croatia (Primorje-Gorski Kotar, Lika-Senj, Zadar, Šibenik-Knin, Split-Dalmatia, Istra, and Dubrovnik-Neretva) (Table 7). To make comparable areas of different extent, the decision to standardize over square kilometers was made. Indeed, the assessed coastal surface considered has been calculated differently for Italy and Croatia, due to the different input data used. For Italy, the surface of each municipality has been used. For Croatia, although the data provided is relative to the coastal counties, it has been assumed that tourists are mainly located in the coastal part of these counties, corresponding to the coastal municipalities which have been used to map these data (Fig. 26).

Region	Tourist arrival (ISTAT, 2018)
Friuli-Venezia Giulia	1 824 884
Veneto	9 139 107
Emilia-Romagna	6 102 285
Marche	1 472 171
Total	18 238 447

Table 7. Number of tourist arrivals in the coastal municipalities of the four Italian regions facing the northern-central Adriatic Sea (data from ISTAT, the year 2018).

County	Tourist arrivals (Croatian Bureau of Statistics, 2018)
Istra	4 332 752
Primorje-Gorski kotar	2 909 914
Lika-Senj	789 330
Zadar	1 664 467
Šibenik-Knin	965 203
Split-Dalmatia	3 474 145
Dubrovnik-Neretva	2 734 000
Total	16 869 811

Table 8. The number of tourist arrivals in the counties of Adriatic Croatia (data from Croatian Bureau of Statistics, the year 2018).

Figure 26: Flow of ES tourism as the number of tourist arrivals per km 2 year in the coastal municipalities or counties facing Northern-Central Adriatic Sea.

Recreational boating capacity:

A total of around 220 questionnaires (though *ad hoc* and therefore different from the one for tourists) have been responded to by recreational boaters along the coasts of the study area over summer 2019. The questionnaires had the aim to respond to the question: how much importance did you give to the following characteristics of the natural environment when you decided to visit this area? As above, users have been asked to give an important ratio to different natural features (protected area, terrestrial habitats, marine habitats, and cetaceans). Here, in comparison which above water quality was not included because when one buys a boat because lives there cannot decide for the water quality of the place. Plus, from our questionnaires, the quality of water was not considered very much important for recreational boaters as reported by the responses at the question: How would your number of trips change in detrimental water conditions? All the other features have been rated among: very important, moderately important, slightly important, not important. These have been quantified from 0 to 3 and normalized from 0 to 1 (Table 9). They were included as weight (Wn) in the following linear model:

Natural attractiveness potential for recreational boaters = W6*protected area + W7*terrestrial coastal habitat + W8*marine submerged habitat + W9*cetacens

MCES (ecosystem services)	Environmental features	Mean qualitative weight	Mean quantitative weight	Normalized weights value (W_n)
Recreational boating	Protected areas	Moderately/very important	2.2	0.75
	Terrestrial coastal habitats	Moderately/very important	2.2	0.75
	Marine submerged habitats	Moderately important	2	0.68
	Cetaceans	Slightly/moderately important	1.8	0.60

Table 9. Values of different environmental features considered in the assessment of recreational boating capacity

All the environmental features have been mined as spatial data from different sources which are the same used to calculate the tourism capacity (Table 6) except for what concerns “water quality” which, as has been already said, was not considered for the assessment of recreational boating capacity. Fig. 27 reports the map result of the presented methodology.

Figure 27: Normalized capacity of the ES recreational boating corresponding to the spatial explicit indicators for attractiveness potential of the Northern-Central Adriatic system (1x1Km resolution).

Recreational boating flow

The indicator chosen for the flow of recreational boating is the number of boat trips per square kilometer per year, which represents the use of the coastal-marine ecosystem by boaters. This indicator has been estimated by combining information on the individual behavior of boaters, collected through questionnaires (the same used for the capacity), and information on the number and spatial distribution of leisure boats in the study area. The questionnaires asked for information regarding the port of departure, the main destinations, and the average distance traveled. Based on this information, the average frequency of the boat trips and the average distance covered during the trips were calculated (just for daily trips), distinguishing between motorboats and sailboats. The boat trips longer than one day were not mapped, as it was not possible to identify a general individual behavior, being the destinations and distances extremely variable.

Regarding the number and spatial distribution of leisure boats in the study area, the data collected differ between Italy and Croatia. Concerning the Italian coast, data on the number of leisure boats were retrieved from the Coast

Guard registry ('Capitaneria di Porto'), divided by maritime compartments, relative to the year 2018. The dataset distinguishes between motorboats and sailboats and specifies the distribution by classes of length. To obtain greater detail in the spatial distribution of the boats, the location of ports and marinas located in each maritime compartment was retrieved from the website Tuttobarche.it (https://www.tuttobarche.it/ricerca-rade-e-porti), along with the number of boat places in each marina. For the small subset of the marinas for which the number of boat places was not available, this parameter was estimated based on the surface of the marina, following a regression between the surface of the marina and the number of boat places resulting from the data from the other marinas. Finally, the boats registered in each maritime compartment were distributed among the marinas located therein, according to the number of boat places available. For each marina, it was assumed that the proportion of motorboats and sailboats, as well as the length distribution, correspond to the proportions reported for the respective compartment. In this way, an estimate of the spatial distribution of motor and sailboats in the Italian portion of the study area was obtained. It should be noted that registration to the Coast Guard's registry for boats with lengths <10 m is optional. Therefore, the number of boats registered is likely to be an underestimate of the total number of boats, especially regarding motorboats. Concerning the Croatian coast, the number of leisure boats permanently moored in the nautical ports of the Croatian coast was retrieved from the Croatian Bureau of Statistics, relative to the year 2018. In this case, the dataset provides the total number of boats in each county of Croatia, but unfortunately, no further information on the type of boats and their precise location could be retrieved.

Based on these data, the use of marine space by boaters was estimated by applying the average individual behavior obtained from the questionnaires to the number of boats present in the study area. In particular, the analysis is focused on mapping the marine space used for daily trips, whose destination can be assumed to be the area facing the homeport. Concerning the Italian coast, this was done by mapping a semi-circular marine area, centered on each marina, whose radius equals the average distance traveled by boaters, distinguishing between motorboats and sailboats. Since the boats registered include big vessels that are not likely to be used for daily trips, the number of boats used for daily trips in each marina was estimated based on the ratios derived from the questionnaires, distinguishing between the type of boat and classes of length. For each marina (or groups of marinas in case of very close ones), the number of boat trips/km2/year was obtained by multiplying the number of boats by the frequency of trips and then dividing by the surface of the semi-circular area. Concerning the Croatian coast, as the information retrieved is less detailed, we used a buffer from the coast with a distance equal to the average distance traveled by respondents using motorboats and sailboats. No fractioning relative to the portion of boats making daily trips was applied here, due to the lack of information about the type of boats. For each county, the number of boat trips/km2/year was obtained by multiplying the number of boats by the frequency of trips and then dividing by the surface of the coastal buffer (Fig. 28).

Figure 28: Flow of ES boating reported as daily trips by leisure boaters of Italy and Croatia, expressed as the number of boat passages per year per km 2.

Coastal erosion prevention capacity and flow

To assess the coastal erosion capacity and flow for the study area a methodology already used at the European scale (Liquete et al., 2013) was used. In there, the authors defined as capacity the natural potential that coastal ecosystems possess to protect the coast against erosion and inundation. This is based on geological and ecological characteristics. The authors defined instead exposure as the predicted need of coastal protection based on the climatic and oceanographic conditions of the area. They have been calculated as follow:

Coastal erosion prevention capacity= 0.33*coastal geomorphology + 0.25*coastal slope+0.21*marine submerged habitat+0.21*terrestrial coastal habitat

Coastal erosion prevention exposure= 0.34* maximum significant wave height+0.32*storm surge + 0.34*relative sea level rise (modified from the original *)

Coastal protection exposure together with coastal protection capacity gives and an indication of the service flow from a natural perspective, which is so calculated as:

Coastal erosion prevention flow = capacity – exposure

All the independent variables in the equations above have been mined from different data sources (Table 10). Given the differences in phenomena represented by the indicators to make them comparable a normalization 0 to 1 was performed. The methodology of Liquete et al., 2013 was developed for an assessment at a European scale. To adapt it for application at the Northern-Central Adriatic scale, some changes were made. The division of the study area into coastal segments has been performed to create segments of around 10km and not of 30km as performed in Liquete et al., 2013. For each of these segments, coastal erosion prevention capacity and flow were assessed. (*) Tides included in the original equation for exposure here are excluded from the equation because the tidal range in the present study area is not relevant to calculating the exposure. To better support these concepts data for the Italian part reported in the national mareographic system (https://www.mareografico.it/) was checked to represent a tidal range for the area of 1 meter on average. Liquete et al., 2013 considering the whole European continent included tides stated that the coastal system subjected to macrotidal ranges (e.g., the UK) will better accept an increase in exposure because it is better adapted to fluctuation. That is not the case for areas, such as the one in this manuscript, that are not used to these high tidal ranges.

Factor of coastal erosion prevention	Mapping method and adaptation, data source and unit of measurment
Coastal geomorphology	Coastal geomorphology for the study area was retrivied from Wolff et al., 2018 (reported as coastal material). In there, the artificial portion of the class "unerodible" was counted as 0 since the natural capacity was unexistent. The other part of the class unerodible together with the three other classes: rock with pocket beaches, sand and mud were rescaled to adapt those data to those required by the Liquete et al., 2013 methodology (from 1 to 4, adimentional).
Coastal slope	Coastal slope was calculated comparing DEM[1] and GEBCO[2] data and the avarage was extracted in the 10 km coastal segments (degree)
Marine submerged habitats	Suitability model derived distribution of *Posidonia oceanica* was retrived from EMODnet [3] referencing the work of Scardi et al. 2013 (categorical P/A, adimentional)
Terrestrial coastal habitat	Proportion of natural habitats within segments of coast approx. 10 km long and 10 km wide. Natural habitats have been extracted from Corine Land Cover 2018 (CLC [4]), corresponding the following land-cover types: all forest types (CLC code 31*); all scrub and/or herbaceous vegetation associations (CLC code 32*); among open spaces with little or no vegetation (CLC code 33*), the classes beaches dunes and sand, and sparsely vegetated areas (CLC codes 331 and 332 respectively); all wetlands, both inland and marine (CLC code 4**) (categorical P/A, adimentional)

Maximum significant wave height	Spectral significant wave height (VHM_0) avaraged yearly for the period 2008-2018 from the coupled hydrodynamic-wave model: MEDSEA_ANALYSIS_FORECAST_PHY_006_013 from CMEMS[5]. It was extracted and avaraged in the 10 km coastal units.
Storm surge	Storm surge data were retrivied from Wolff et al., 2018 and there reported as DCESL01. (meters)
Relative sea level rise (RSLR)	RSLR data were retrived at the regional scale for Italy from the national mareographic system [6]. For Croatia, data of RSLR were retrived from Baric et al., 2008 (millimeters/year)

---

[1] Copernicus Land monitoring service: https://land.copernicus.eu/imagery-in-situ/eu-dem/eu-dem-v1.1

[2] GEBCO (General Bathymetric Chart of the Ocean): https://www.gebco.net/data_and_products/gridded_bathymetry_data/

[3] EMODnet (European Marine Observation and Data Network): https://www.emodnet.eu/en/map-week-%E2%80%93-posidonia-oceanica-distribution-seagrass-species

[4] Copernicus Land Monitoring Service: https://land.copernicus.eu/pan-european/corine-land-cover

[5] CMEMS (Copernicus marine services): https://resources.marine.copernicus.eu/?option=com_csw&view=details&product_id=MEDSEA_ANALYSIS_FORECAST_PHY_006_013

[6] Mareografico: https://www.mareografico.it/

Table 10. Biophysical features are considered in the calculation of the coastal erosion potential capacity and flow.

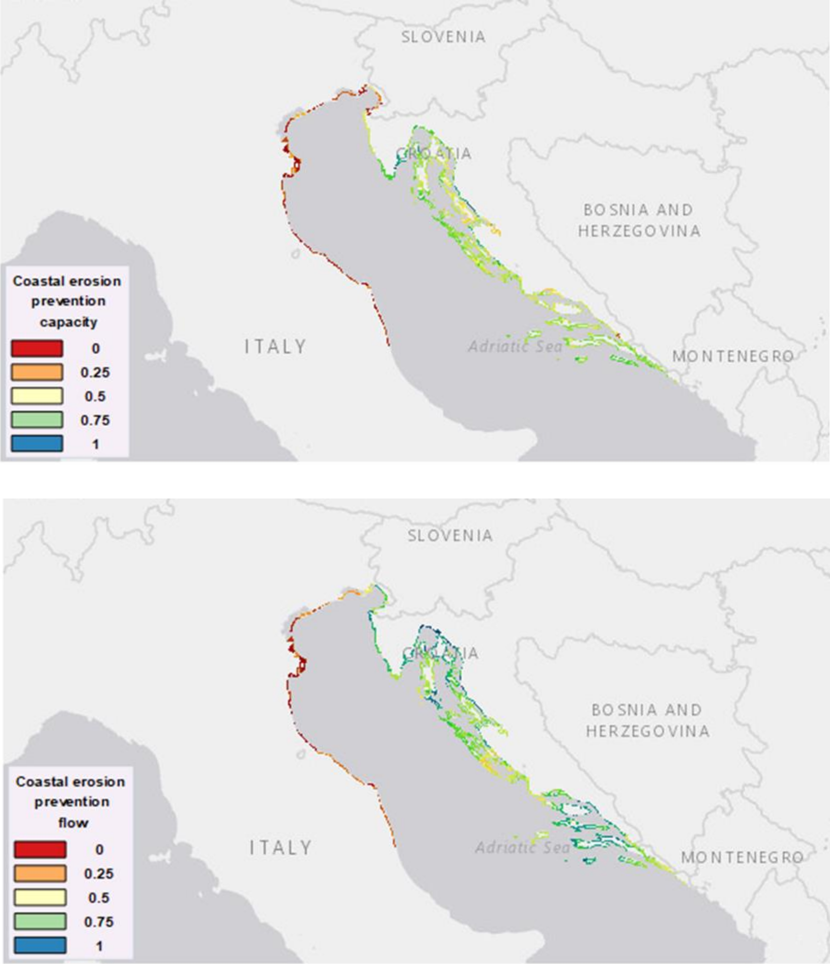


Figure 29: Normalized coastal erosion prevention capacities (a) and coastal erosion prevention flow (b) per the whole set of 10km coastal segments.

Carbon sequestration capacity and flow:

The ES carbon sequestration capacity and flow have been obtained as a single map because the capacity to sequester carbon coincides with the carbon needed to be sequestered so to benefit humans, therefore capacity and flow coincide. The indicator used in this case has been kilograms of carbon dioxide (CO2) per km2 per year. The habitats considered as important for this role in the coastal-marine habitat of the study area have been, broad-leaved forests, mixed forests, coniferous forests (as main constituents of the Mediterranean maquis), and, saltmarshes, sclerophyllous, vegetated dunes, *Posidonia oceanica* meadows, and burial rate of the sea bottom.

For each of these habitats, average rates of CO2 sequestration have been retrieved from literature, considering studies conducted in locations close or comparable to the Northern-Central Adriatic Sea (Table 11). The presence/absence of each habitat has been mapped based on the data sources listed in Table 11, and a map of carbon sequestration from each habitat has been obtained by multiplying the map of each habitat by their respective sequestration rate. These maps have then been summed up to produce the overall map of carbon sequestration of the study area (Fig. 30).

Factor of carbon sequestration	CO2 sequestration (kg CO2/km2/year)	Reference of sequestration ratio	Sources of spatial habitat data
Saltmarshes	51.9	Beaumont et al. 2014	CLC (2018) [1]
Broad-leaves forests	67.4	Barbati et al. 2014	CLC (2018)
Coniferous forests	89.7	Barbati et al. 2014	CLC (2018)
Sclerophyllus	8	Gratani et al. 2013	CLC (2018)
Posidonia oceanica	39	Campagne et al. 2015	EMODnet [2], Scardi et al. 2016
Vegetated dunes	17.4	Drius et al. 2019	CLC (2018) corected in the Italian side by Drius et al. 2019
Mixed_forest	78	Barbati et al. 2014 (avarage between coniferous and broad leaves forests rates)	CLC (2018)
Burial rate	0.0000007045	Giani et al. 2001	Shapefile of the Adriatic basin, converted as Raster

[1] Copernicus Land Monitoring Service: https://land.copernicus.eu/pan-european/corine-land-cover

[2] EMODnet (European Marine Observation and Data Network): https://www.emodnet.eu/en/map-week-%E2%80%93-posidonia-oceanica-distribution-seagrass-species

Table 11: Environmental features, CO2 sequestration potential, references and sources considered in the calculation for the ES carbon sequestration capacity and flow.

Considering the total area covered by each considered habitat in the study area, a total of 1.243.577 tCO2 is annually sequestered by the marine-coastal Adriatic Sea environment.

Figure 30: Normalized carbon sequestration potential as tons of carbon dioxide (CO2) sequestered by the selected habitat per km 2 per year (1x1Km resolution).

Mussel and whitefish aquaculture capacity:

The explanation for the methodologies used for mussel and whitefish aquaculture are here reported together because the approaches used are similar. However, it is important to notice that the capacities of mussel and whitefish aquaculture have been assessed just for the Italian part because of data limitations. While this approximation could be acceptable for mussel aquaculture for which production is far higher on the Italian side than on the Croatian one. The possible limitation arises from the capacity of whitefish aquaculture given that this type of farming is higher on the Croatian side than on the Italian one.

The ES mussel aquaculture capacity has been assessed through the eco-physiological model calibrated for *Mytilus galloprovincialis* (R package RAC, Baldan et al. 2018), which was forced using time-series of sea surface temperature and chlorophyll A data, and on data concerning sea currents. Mean monthly sea surface temperature (SST) over 30 years has been calculated for the area from MEDSEA_REANALYSIS_PHYS_006_004: SV03-MED-INGV-TEM-REAN-M (Sources: CMEMS). The Chlorophyll A was averaged for 20 years from OCEANCOLOUR_MED_CHL_L4_REP_OBEERVATIONS_009_078: DATASET-OC-MED-CHL-MULTI_CCI-L4-CHL_1KM_MONTHLY-REP-V02 to obtain a monthly mean surface ChlA representative of one mean year (Sources: CMEMS). Sea surface currents have been averaged over a 30-years time series from MEDSEA_REANALYSIS_PHYS_006_004: SV03-MED-INGV-CUR-REAN-M. These data, once averaged for the time dimensions, have been averaged for the space dimension in operational units or boxes. The eco-physiological model (or bioenergetic model) is a box model therefore the definition of areas in which the model should run is necessary. The territorial water (12nm) of the four regions from Friuli-Venezia Giulia to Marche were divided, parallel to the coast, based on GEBCO following depths ranging between 0-10 km, 10-20 km, 20-30 km. In a direction perpendicular to the coast, the subdivision is based on the abiotic characteristics of the area. In this sense, especially chlorophyll A was considered a factor shaping differences along the Italian coastal area considered. For example, the marine area in front of the Po river Delta has been treated as an intermediated region between Veneto and Emilia-Romagna because of the unique abiotic characteristics of the area (i.e., highly eutrophicated by Po river run-off). The operational units division is reported in Fig. 31.

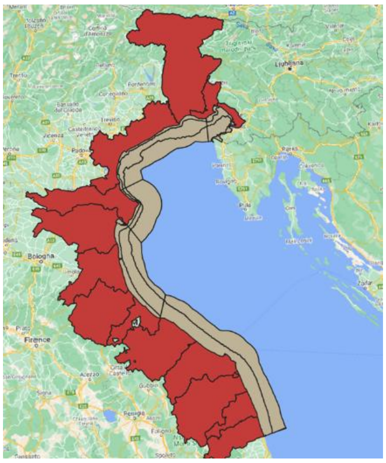


Figure 31: Division of territorial waters of the four regions: Friuli-Venezia Giulia, Veneto, Emilia-Romagna and Marche. They have been used as boxes to run the eco-physiological model for the evaluation of the ES mussel aquaculture capacity.

To each box monthly data of mean SST, monthly data of mean ChlA, area(km2), length, width, and depth were attached. The RAC model has been forced by the abiotic data and with data already included and standardize in the model (such as POC, POM, TSM). Indeed, the RAC model has fixed values that represent average values of POC, POM, and TSM for a general aquaculture marine system (for more details see Baldan et al., 2018). The clearance rate (L/h) as the mean filtration of an organism (*Mytilus galloprovincialis*) has been obtained. The carrying capacity (K), the ability of the ecosystem to sustain mussel aquaculture, has been calculated as:

$$\boldsymbol{K} = \frac{\dfrac{\boldsymbol{P}}{\boldsymbol{Q}} - (\boldsymbol{F} * \boldsymbol{V})}{\boldsymbol{C}}$$

Where:

- Carrying capacity (**K**): maximum number of mussels which could be produced in the area (adimensional)
- Primary production rate (**P**): 1000 (from Poddu et al. 1998 observations)*box area (mgC days-1)
- Critical Plankton Concentration (**Q**): constant (mg C/m3) (from Smaal&Vonck,1997)
- Flushing rate (**F)**: calculated by the time of residency in the box (box length (meters)/ current mean (both u and v velocity components) hence how much nutrients remain in the box before being flushed away from sea currents (days^-1).
- Volume of the single box (**V**): calculated as length* width*depth (meters^3)
- Clearance rate (**C**): obtained by the bioenergetic model forced by temperature and ChlA data (Litres/hours)

The carrying capacity (K) per box is divided by 10^6 because the mean mussel weight is 1 gram and multiplied by 15 as clams have an average weight of 15g. It is worth noticing that these results do not consider other economic activities of the area or other types of spatial constrain representing the maximum amount of mussels that can be theoretically produced. Noteworthy, the capacity is the ability of the ecosystem to provide a particular ecosystem service thus the ES mussel aquaculture capacity can be the irrealistic productive quantity of mussels (Fig. 32). Although Croatian data are not available the capacity of mussel production is higher on the Italian side than the Croatian. Hence, this result can be a nice approximation of the reality but given the uncertainty of outcomes of this generalization, we preferred not to include this ES in the multiple ES capacity and flow analysis.

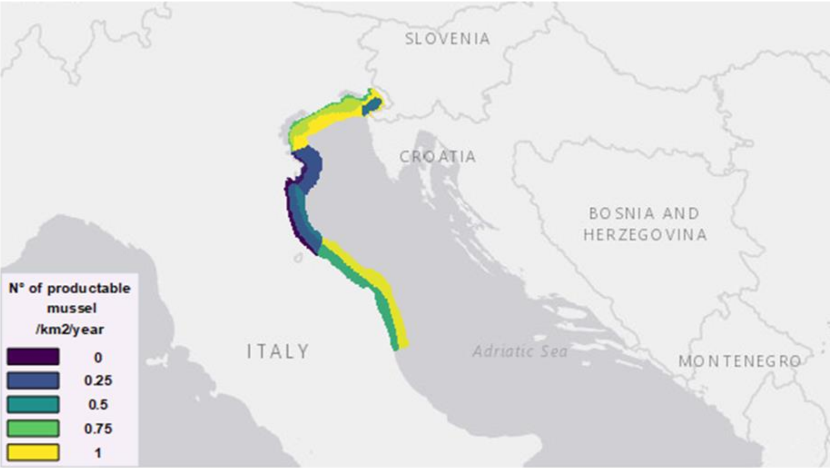


Figure 32: Normalized mussel aquaculture capacity as a result of the RAC bio-energetic box model based on sea surface temperature and chlorophyll A data.

The capacities of the ES whitefish aquaculture have been retrieved from Porporato et al., 2020 where just the Italian side it was available. The study assessed the suitability of the whole Italian territorial water (from coastline to 12nm) for whitefish aquaculture: European seabass (*Dicentrarchus labrax*) and gilthead seabream (*Sparus aurata*) in three different scenarios to the whole Italian territorial water. For the present study, just the Northern-Central Adriatic territorial sea was considered and the economic scenario was picked up as the one with the highest exploitation possible of the site (Fig. 33). Croatian data are not available, that is the reason for not including this ES in the multiple ES capacity and flow analysis.

Figure 33: Normalized whitefish aquaculture capacity as the suitability of whitefish farming based on sea surface temperature, Significant Wave Height (SWH), distance to the harbor, current sea uses, and cumulative impacts data (Porporato et al., 2020).

Mussel aquaculture and whitefish aquaculture flow:

Here, as above the ESs mussel and whitefish aquaculture are treated together since they are similar in the assessment methodology. Data for spatially assessing the ES mussel aquaculture flow have been retrieved from different sources. Existent mussel farms shapefile has been found in Tool4MSP portal, both for the Italian and the Croatian sides. The productivity of the Italian side has been defined by Prioli's reports (18th Censimento Nazionale sulla molluschicoltura del consorzio Unimar, Giuseppe Prioli) as time series of mussel aquaculture produced by region in a period 2006-2015 (Table 12). The average of these years per region has been divided by the areas of the farms present in the territorial water of the region to obtain the tons of mussel produced per km^2 per year.

Italian regions and Croatian counties	Mean annual productivity of the mussel farm (from 2006-2015) (tons)	Mean annual productivity of the whitefish farm (from 2006-2015) (tons)
Friuli-Venezia Giulia	2810	800
Veneto	4540	NA
Emilia-Romagna	18530	NA
Marche	3058	NA
Istra	110	287
Sibensk-kniska	33	108
Lika-senj	0	NA
Zadarska	230	5943
Slitsko-dalmatinska	66	284
Dubrovacko-neretvanska	294	518

Table 12: Mean annual productivity per area per site (sources: Prioli's reports UNIMAR, FishStat Plus (FAO), Tool4MSP).

The same methodology was applied to the Croatian side but annual data of the production of mussels from aquaculture have been retrieved from FAO (FishStat), as national-level data. Being that our assessment covers the whole country of Croatia. The surface under concession of mussel farming in Croatia per county has been retrieved from the Croatian website: https://ribarstvo.mps.hr/default.aspx?id=14. The mean annual productivity (2006-2015) for Croatia has been divided based on the proportion of confessed surface obtaining the annual productivity of mussels per each Croatian county (Fig. 34).

Figure 34: Normalized mussel production in tons per km2 per year.

The assessment of the whitefish aquaculture flow on the Croatian side used the same data sources and methodology steps for mussel aquaculture flow (FishStat for national data and surface of concession for division) but obtained an average annual value from 2011 to 2018 data (since data before were too coarse). The annual production average for the whole country is 7753 tons for European seabass (*Dicentrarchus labrax*) and gilthead seabream (*Sparus aurata*) together. This data of production is mainly subjectable at Zadarska county, the main producer of farmed whitefishes. The marine area of the four regions constituting the study area for the Italian side, instead, is not devoted to whitefish aquaculture. Infact, the only region considered as producer of farmed whitefishes was the Friuli-Venezia Giulia of which production was retrieved by a local analysis (Distretto di Pesca Nord Adriatico - Analisi della filiera dell’acquacoltura – 2015 Osservatorio Socio Economico della Pesca e dell'Acquacoltura). Noteworthy, the production record used for the present assessment for the Friuli-Venezia Giulia region considers other fishes (not specified in the original data) besides European seabass and gilthead seabream. Even Veneto and Emilia-romagna regions were reported as producers of whitefishes in the same report of Friuli-Venezia Giulia with 600 and 20 tons respectively. However, none of the information was given about the percentage of aquaculture occurring in the sea or lagoons (i.e., Venice lagoons for Veneto or Sacca di Goro for Emilia-Romagna). Through expert judgments (F. Pranovi, D. Brigolin, S. Rova) many of the white fishes production occurs in the lagoons of these regions, therefore it was decided to give 0 values to the Veneto and Emilia-Romagna regions. The same expert judgment and data were applied to the Marche region (Fig. 35).

Figure 35: Normalized whitefish production in tons per km2 per year.

Industrial fishery flow

The last ES considered in the assessment is the trawl fishing activity (considering OTB, rapido trawl and PTM fishing gears) (Russo et al., 2020). Here this ES has been measured as the spatial explicit fishing effort, estimated by using the Automatic Identification System (AIS). Specifically, data reported in Russo et al., 2020 was used to create a high-resolution map (1x1Km) of the fishing effort in the Northern-Central Adriatic sea (Fig. 36). Capacity of the industrial fishery has no spatial explicit data therefore it was not considered in the present assessment.

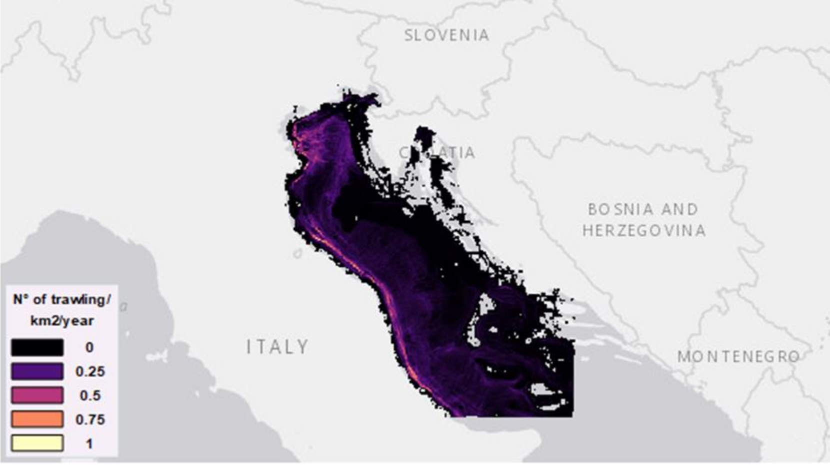
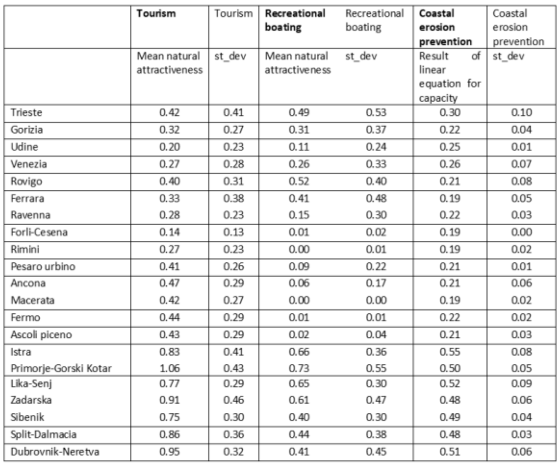


Figure 36: Normalized spatially explicit annual average of industrial fishing effort by trawling (2018).

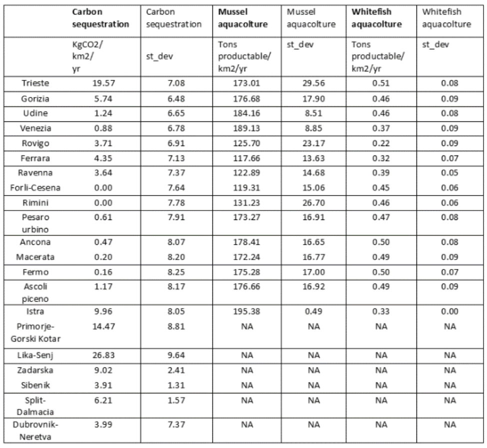
Multiple ESs spatial analysis

In the following tables, there are mean values (with the unit of measurement) and standard deviation of each ES capacity (Table 13) and flow (Table 14) before normalization. Data refers to each ES averaged in the 21 subunits, fourteen provinces comprised in the Marche, Emilia-Romagna, Veneto, and Friuli-Venezia Giulia Italian regions and the seven coastal counties of Croatia, of which the coastal portion (the first 10 km inland parallel to the coastline) and, the marine area within the territorial seawater (12 nm from the coastline).

To conclude, Fig. 37 reports all the seven normalized (0-1) ES capacities and flows considered in the present study.

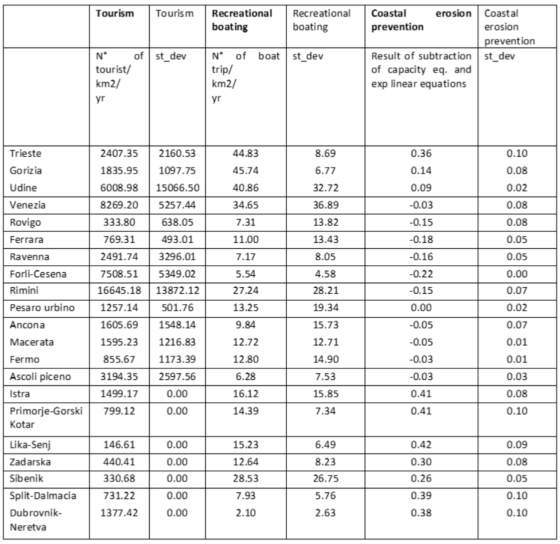


	Tourism	Tourism	**Recreational boating**	Recreational boating	**Coastal erosion prevention**	Coastal erosion prevention
	Mean natural attractiveness	st_dev	Mean natural attractiveness	st_dev	Result of linear equation for capacity	st_dev
Trieste	0.42	0.41	0.49	0.53	0.30	0.10
Gorizia	0.32	0.27	0.31	0.37	0.22	0.04
Udine	0.20	0.23	0.11	0.24	0.25	0.01
Venezia	0.27	0.28	0.26	0.33	0.26	0.07
Rovigo	0.40	0.31	0.52	0.40	0.21	0.08
Ferrara	0.33	0.38	0.41	0.48	0.19	0.05
Ravenna	0.28	0.23	0.15	0.30	0.22	0.03
Forli-Cesena	0.14	0.13	0.01	0.02	0.19	0.00
Rimini	0.27	0.23	0.00	0.01	0.19	0.02
Pesaro urbino	0.41	0.26	0.09	0.22	0.21	0.01
Ancona	0.47	0.29	0.06	0.17	0.21	0.06
Macerata	0.42	0.27	0.00	0.00	0.19	0.02
Fermo	0.44	0.29	0.01	0.01	0.22	0.02
Ascoli piceno	0.43	0.29	0.02	0.04	0.21	0.03
Istra	0.83	0.41	0.66	0.36	0.55	0.08
Primorje-Gorski Kotar	1.06	0.43	0.73	0.55	0.50	0.05
Lika-Senj	0.77	0.29	0.65	0.30	0.52	0.09
Zadarska	0.91	0.46	0.61	0.47	0.48	0.06
Sibenik	0.75	0.30	0.40	0.30	0.49	0.04
Split-Dalmacia	0.86	0.36	0.44	0.38	0.48	0.03
Dubrovnik-Neretva	0.95	0.32	0.41	0.45	0.51	0.06

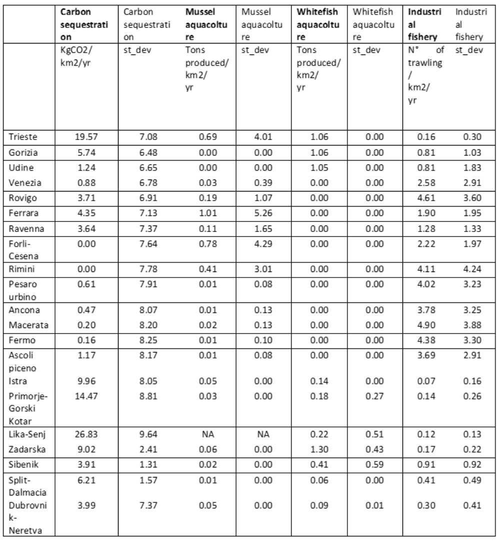


	Carbon sequestration	Carbon sequestration	**Mussel aquacolture**	Mussel aquacolture	**Whitefish aquacolture**	Whitefish aquacolture
	KgCO2/ km2/ yr	st_dev	Tons productable/ km2/yr	st_dev	Tons productable/ km2/yr	st_dev
Trieste	19.57	7.08	173.01	29.56	0.51	0.08
Gorizia	5.74	6.48	176.68	17.90	0.46	0.09
Udine	1.24	6.65	184.16	8.51	0.46	0.08
Venezia	0.88	6.78	189.13	8.85	0.37	0.09
Rovigo	3.71	6.91	125.70	23.17	0.22	0.09
Ferrara	4.35	7.13	117.66	13.63	0.32	0.07
Ravenna	3.64	7.37	122.89	14.68	0.39	0.05
Forli-Cesena	0.00	7.64	119.31	15.06	0.45	0.06
Rimini	0.00	7.78	131.23	26.70	0.46	0.06
Pesaro urbino	0.61	7.91	173.27	16.91	0.47	0.08
Ancona	0.47	8.07	178.41	16.65	0.50	0.08
Macerata	0.20	8.20	172.24	16.77	0.49	0.09
Fermo	0.16	8.25	175.28	17.00	0.50	0.07
Ascoli piceno	1.17	8.17	176.66	16.92	0.49	0.09
Istra	9.96	8.05	195.38	0.49	0.33	0.00
Primorje-Gorski Kotar	14.47	8.81	NA	NA	NA	NA
Lika-Senj	26.83	9.64	NA	NA	NA	NA
Zadarska	9.02	2.41	NA	NA	NA	NA
Sibenik	3.91	1.31	NA	NA	NA	NA
Split-Dalmacia	6.21	1.57	NA	NA	NA	NA
Dubrovnik-Neretva	3.99	7.37	NA	NA	NA	NA

Table 13. Extracted value per municipality (IT) and county (HR) per each ES capacity with the included unit of measurement. The mussel and whitefish aquaculture capacities have been assessed just for the Italian side.



	Tourism	Tourism	**Recreational boating**	Recreational boating	**Coastal erosion prevention**	Coastal erosion prevention
	N° of tourist/ km2/ yr	st_dev	N° of boat trip/ km2/ yr	st_dev	Result of subtraction of capacity eq. and exp linear equations	st_dev
Trieste	2407.35	2160.53	44.83	8.69	0.36	0.10
Gorizia	1835.95	1097.75	45.74	6.77	0.14	0.08
Udine	6008.98	15066.50	40.86	32.72	0.09	0.02
Venezia	8269.20	5257.44	34.65	36.89	-0.03	0.08
Rovigo	333.80	638.05	7.31	13.82	-0.15	0.08
Ferrara	769.31	493.01	11.00	13.43	-0.18	0.05
Ravenna	2491.74	3296.01	7.17	8.05	-0.16	0.05
Forli-Cesena	7508.51	5349.02	5.54	4.58	-0.22	0.00
Rimini	16645.18	13872.12	27.24	28.21	-0.15	0.07
Pesaro urbino	1257.14	501.76	13.25	19.34	0.00	0.02
Ancona	1605.69	1548.14	9.84	15.73	-0.05	0.07
Macerata	1595.23	1216.83	12.72	12.71	-0.05	0.01
Fermo	855.67	1173.39	12.80	14.90	-0.03	0.01
Ascoli piceno	3194.35	2597.56	6.28	7.53	-0.03	0.03
Istra	1499.17	0.00	16.12	15.85	0.41	0.08
Primorje-Gorski Kotar	799.12	0.00	14.39	7.34	0.41	0.10
Lika-Senj	146.61	0.00	15.23	6.49	0.42	0.09
Zadarska	440.41	0.00	12.64	8.23	0.30	0.08
Sibenik	330.68	0.00	28.53	26.75	0.26	0.05
Split-Dalmacia	731.22	0.00	7.93	5.76	0.39	0.10
Dubrovnik-Neretva	1377.42	0.00	2.10	2.63	0.38	0.10



	Carbon sequestration	Carbon sequestration	**Mussel aquacolture**	Mussel aquacolture	**Whitefish aquacolture**	Whitefish aquacolture	**Industrial fishery**	Industrial fishery
	KgCO2/ km2/yr	st_dev	Tons produced/ km2/ yr	st_dev	Tons produced/ km2/ yr	st_dev	N° of trawling / km2/ yr	st_dev
Trieste	19.57	7.08	0.69	4.01	1.06	0.00	0.16	0.30
Gorizia	5.74	6.48	0.00	0.00	1.06	0.00	0.81	1.03
Udine	1.24	6.65	0.00	0.00	1.05	0.00	0.81	1.83
Venezia	0.88	6.78	0.03	0.39	0.00	0.00	2.58	2.91
Rovigo	3.71	6.91	0.19	1.07	0.00	0.00	4.61	3.60
Ferrara	4.35	7.13	1.01	5.26	0.00	0.00	1.90	1.95
Ravenna	3.64	7.37	0.11	1.65	0.00	0.00	1.28	1.33
Forli-Cesena	0.00	7.64	0.78	4.29	0.00	0.00	2.22	1.97
Rimini	0.00	7.78	0.41	3.01	0.00	0.00	4.11	4.24
Pesaro urbino	0.61	7.91	0.01	0.08	0.00	0.00	4.02	3.23
Ancona	0.47	8.07	0.01	0.13	0.00	0.00	3.78	3.25
Macerata	0.20	8.20	0.02	0.13	0.00	0.00	4.90	3.88
Fermo	0.16	8.25	0.01	0.10	0.00	0.00	4.38	3.30
Ascoli piceno	1.17	8.17	0.01	0.08	0.00	0.00	3.69	2.91
Istra	9.96	8.05	0.05	0.00	0.14	0.00	0.07	0.16
Primorje-Gorski Kotar	14.47	8.81	0.03	0.00	0.18	0.27	0.14	0.26
Lika-Senj	26.83	9.64	NA	NA	0.22	0.51	0.12	0.13
Zadarska	9.02	2.41	0.06	0.00	1.30	0.43	0.17	0.22
Sibenik	3.91	1.31	0.02	0.00	0.41	0.59	0.91	0.92
Split-Dalmacia	6.21	1.57	0.01	0.00	0.06	0.00	0.41	0.49
Dubrovnik-Neretva	3.99	7.37	0.05	0.00	0.09	0.01	0.30	0.41

Table 14. Extracted value per municipality (IT) and county (HR) per each ES flow with the included unit of measurement.

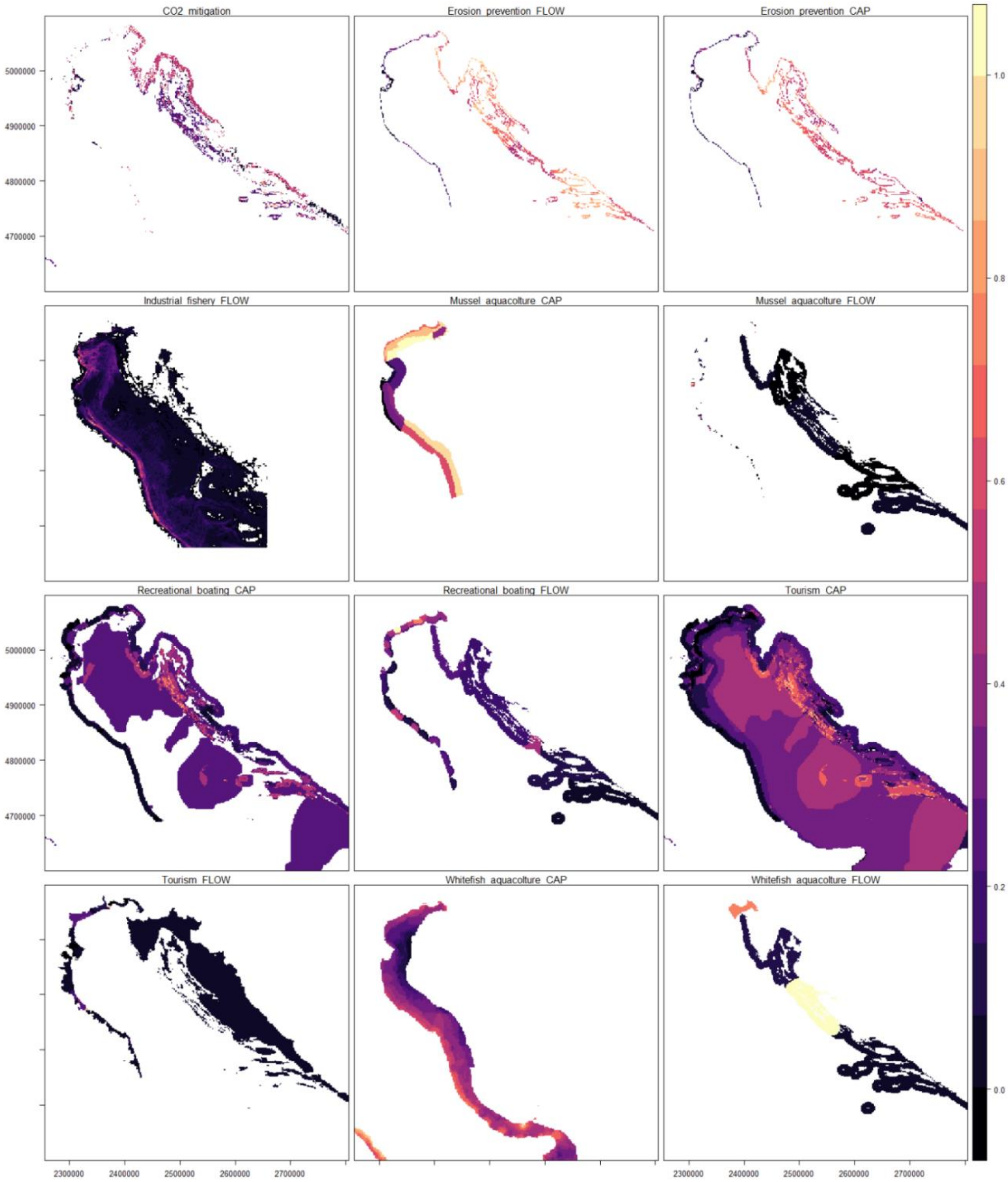


Figure 37: Normalized (0-1) ES capacity and flow for each of the seven ESs considered in the Northern-Central Adriatic multiple assessment.

Reference

Abdulla, A., Gomei, M., Hyrenbach, D., Notarbartolo-di-Sciara, G., & Agardy, T. (2009). Challenges facing a network of representative marine protected areas in the Mediterranean: prioritizing the protection of underrepresented habitats. ICES Journal of marine science, 66(1), 22-28.

Abunge, C., Coulthard, S., & Daw, T. M. (2013). Connecting marine ecosystem services to human well-being: Insights from participatory well-being assessment in Kenya. Ambio, 42(8), 1010–1021. https://doi.org/10.1007/s13280-013-0456-9

Andersen, K. H., Beyer, J. E., & Lundberg, P. (2009). Trophic and individual efficiencies of size-structured communities. Proceedings of the Royal Society B: Biological Sciences, 276(1654), 109–114. https://doi.org/10.1098/RSPB.2008.0951

Andersson, E., Barthel, S., & Ahrné, K. (2007). Measuring social–ecological dynamics behind the generation of ecosystem services. Ecological applications, 17(5), 1267-1278.

Aquaculture Croatian website: https://ribarstvo.mps.hr/default.aspx?id=14

Araújo, M. B., & New, M. (2007). Ensemble forecasting of species distributions. Trends in Ecology & Evolution, 22(1), 42–47. https://doi.org/10.1016/J.TREE.2006.09.010

Aree Protette- https://ambiente.regione.emiliaromagna.it/it/notizie/attualita/2019/febbraio/accordo-tra-le-tre-regioni-del-distretto-alto-adriatico-per-unarea-marina-a-tutela-di-delfini-e-tartarughe

Azzurro, E., Sbragaglia, V., Cerri, J., Bariche, M., Bolognini, L., Ben Souissi, J., ... & Moschella, P. (2019). Climate change, biological invasions, and the shifting distribution of Mediterranean fishes: A large-scale survey based on local ecological knowledge. Global change biology, 25(8), 2779-2792.

Baldan, D., Porporato, E. M. D., Pastres, R., & Brigolin, D. (2018). An R package for simulating growth and organic wastage in aquaculture farms in response to environmental conditions and husbandry practices. PloS one, 13(5), e0195732.

Balvanera, P., Daily, G. C., Ehrlich, P. R., Ricketts, T. H., Bailey, S. A., Kark, S., ... & Pereira, H. (2001). Conserving biodiversity. Science, 291, 2047.

Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J. S., Nakashizuka, T., Raffaelli, D., & Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecology letters, 9(10), 1146-1156.

Balvanera, P., Brauman, K. A., Cord, A. F., Drakou, E. G., Geijzendorffer, I. R., Karp, D. S., ... & Schröter, M. (2022). Essential ecosystem service variables for monitoring progress towards sustainability. Current Opinion in Environmental Sustainability, 54, 101152.

Baric, A., Grbec, B., & Bogner, D. (2008). Potential implications of sea-level rise for Croatia. Journal of Coastal Research, 24(2), 299-305. Copernicus Land Monitoring Service - https://land.copernicus.eu/pan-european/corine-land-cover

Barker, J. S. (2009). Defining fitness in natural and domesticated populations. Adaptation and fitness in animal populations: evolutionary and breeding perspectives on genetic resource management, 3-14.

Barnosky, A. D., Matzke, N., Tomiya, S., Wogan, G. O., Swartz, B., Quental, T. B., ... & Ferrer, E. A. (2011). Has the Earth's sixth mass extinction already arrived? Nature, 471(7336), 51-57.

Basconi, L., Rova, S., Stocco, A., & Pranovi, F. (2023). Ecosystem services for supporting coastal and marine resources management, an example from the Adriatic sea (Central Mediterranean sea). Ocean & Coastal Management, 235, 106486.

Basconi L. (2023). First Evidence of Spatial Relationships between Multiple Marine Ecosystem Services and Functioning Indicators. J Ecol & Nat Resour, 7(1): 000321.

Bastian, O., Haase, D., & Grunewald, K. (2012). Ecosystem properties, potentials and services–The EPPS conceptual framework and an urban application example. Ecological indicators, 21, 7-16.

Beaumont, N. J., Jones, L., Garbutt, A., Hansom, J. D., & Toberman, M. (2014). The value of carbon sequestration and storage in coastal habitats. Estuarine, Coastal and Shelf Science, 137, 32-40.

Bellard, C., Bertelsmeier, C., Leadley, P., Thuiller, W., & Courchamp, F. (2012). Impacts of climate change on the future of biodiversity. Ecology letters, 15(4), 365-377.

Bellwood, D. R., & Meyer, C. P. (2009). Searching for heat in a marine biodiversity hotspot. Journal of Biogeography, 36(4), 569–576. https://doi.org/10.1111/j.1365-2699.2008.02029.x

ben Rais Lasram, F., & Mouillot, D. (2009). Increasing southern invasion enhances congruence between endemic and exotic Mediterranean fish fauna. Biological invasions, 11(3), 697-711.

ben Rais Lasram, F., Guilhaumon, F., Albouy, C., Somot, S., Thuiller, W., & Mouillot, D. (2010). The Mediterranean Sea as a “cul-de-sac” for endemic fishes facing climate change. Global Change Biology, 16(12), 3233–3245. https://doi.org/10.1111/j.1365-2486.2010.02224.x

ben Rais Lasram, F., Tomasini, J. A., Guilhaumon, F., Romdhane, M. S., do Chi, T., & Mouillot, D. (2008). Ecological correlates of dispersal success of Lessepsian fishes. Marine Ecology Progress Series, 363, 273–286. https://doi.org/10.3354/meps07474

Bennett, E.M., & Chaplin-Kramer, R. (2016). Science for the sustainable use of ecosystem services. F1000Research, 5.

Bennett, E.M., Peterson, G. D., & Gordon, L. J. (2009). Understanding relationships among multiple ecosystem services. Ecology letters, 12(12), 1394-1404.

Berkes, F., Folke, C., & Colding, J. (Eds.). (2000). Linking social and ecological systems: management practices and social mechanisms for building resilience. Cambridge University Press.

Bertness, M. D., Bruno, J. F., Silliman, B. R., & Stachowicz, J. J. (2014). Marine community ecology and conservation.

Bevilacqua, S., Airoldi, L., Ballesteros, E., Benedetti- Cecchi, L., Boero, F., Bulleri, F., Cebrian, E., Cerrano, C., Claudet, J., Colloca, F., Coppari, M., Di Franco, A., Fraschetti, S., Garrabou, J., Guarnieri, G., Guerranti, C., Guidetti, P., Halpern, B. S., Katsanevakis, S., … Terlizzi, A. (2021). Mediterranean rocky reefs in the Anthropocene: Present status and future concerns. Advances in Marine Biology, 89, 1– 51.

Bevilacqua, S., & Terlizzi, A. (2020). Nestedness and turnover unveil inverse spatial patterns of compositional and functional β-diversity at varying depth in marine benthos. Diversity and Distributions, 26(6), 743–757. https://doi.org/10.1111/ddi.13025

Bianchi, C. N. (2007). Biodiversity issues for the forthcoming tropical Mediterranean Sea. Hydrobiologia, 580(1), 7-21.

Bianchi, C. N., & Morri, C. (2003). Global sea warming and “tropicalization” of the Mediterranean Sea: biogeographic and ecological aspects. Biogeographia – The Journal of Integrative Biogeography, 24. https://doi.org/10.21426/b6110129

Birkhofer, K., Andersson, G. K., Bengtsson, J., Bommarco, R., Dänhardt, J., Ekbom, B., ... & Smith, H. G. (2018). Relationships between multiple biodiversity components and ecosystem services along a landscape complexity gradient. Biological Conservation, 218, 247-253.

Boero F. (1999) Community ecology: a life cycle perspective.In: Farina A. (Ed.),Perspectives in Ecology. Backyus Publish-ers, Leiden: 335–342

Boero, F., & Bonsdorff, E. (2007). A conceptual framework for marine biodiversity and ecosystem functioning. Marine Ecology, 28(SUPPL. 1), 134–145. https://doi.org/10.1111/J.1439-0485.2007.00171.X

Böhnke-Henrichs, A., Baulcomb, C., Koss, R., Hussain, S. S., & de Groot, R. S. (2013). Typology and indicators of ecosystem services for marine spatial planning and management. Journal of environmental management, 130, 135-145.

Bonaldo, D., Antonioli, F., Archetti, R. (2019) Integrating multidisciplinary instruments for assessing coastal vulnerability to erosion and sea level rise: lessons and challenges from the Adriatic Sea, Italy. J Coast Conserv 23, 19–37.

Bonsdorff E., Norkko A., Bostro¨m C. (1995) Recruitment andpopulation maintenance of the bivalveMacoma balthica(L.)– factors affecting settling success and early survival on shal-low sandy bottoms. In: Eleftheriou A., Ansell A.D., SmithC.J. (Eds),Biology and Ecology of Shallow Coastal Waters.Proceedings of the 28th EMBS, 1993. Olsen & Olsen,Fredensborg: 253–260.

Bremner, J., Rogers, S. I., & Frid, C. L. J. (2006). Methods for describing ecological functioning of marine benthic assemblages using biological traits analysis (BTA). Ecological Indicators, 6(3), 609-622.

Brito-Morales, I., Schoeman, D. S., Everett, J. D., Klein, C. J., Dunn, D. C., García Molinos, J., ... & Richardson, A. J. (2022). Towards climate-smart, three-dimensional protected areas for biodiversity conservation in the high seas. Nature Climate Change, 12(4), 402-407.

Burkhard, B., de Groot, R., Costanza, R., Seppelt, R., Jorgensen, S. E., & Potschin, M. (2012). Solutions for sustaining natural capital and ecosystem services.

Burkhard, B., Kandziora, M., Hou, Y., & Müller, F. (2014). Ecosystem service potentials, flows and demands-concepts for spatial localisation, indication and quantification. Landscape online, 34, 1-32.

Cadotte, M. W., Cardinale, B. J., & Oakley, T. H. (2008). Evolutionary history and the effect of biodiversity on plant productivity. Proceedings of the National Academy of Sciences of the United States of America, 105(44), 17012–17017. https://doi.org/10.1073/PNAS.0805962105/SUPPL_FILE/0805962105SI.PDF

Calvo, E., Simó, R., Coma, R., Ribes, M., Pascual, J., Sabatés, A., ... & Pelejero, C. (2011). Effects of climate change on Mediterranean marine ecosystems: the case of the Catalan Sea. Climate Research, 50(1), 1-29.

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., ... & Naeem, S. (2012). Biodiversity loss and its impact on humanity. Nature, 486(7401), 59-67.

Cardinale, B. J., Matulich, K. L., Hooper, D. U., Byrnes, J. E., Duffy, E., Gamfeldt, L., Balvanera, P., O'Connor, M. I., & Gonzalez, A. (2011). The functional role of producer diversity in ecosystems. American Journal of Botany, 98(3), 572–592. https://doi.org/10.3732/ajb.1000364

Cardinale, B. J., Srivastava, D. S., Duffy, J. E., Wright, J. P., Downing, A. L., Sankaran, M., & Jouseau, C. (2006). Effects of biodiversity on the functioning of trophic groups and ecosystems. Nature 2006 443:7114, 443(7114), 989–992. https://doi.org/10.1038/nature05202

Cardinale, B. J., Wright, J. P., Cadotte, M. W., Carroll, I. T., Hector, A., Srivastava, D. S., Loreau, M., & Weis, J. J. (2007). Impacts of plant diversity on biomass production increase through time because of species complementarity. Proceedings of the National Academy of Sciences of the United States of America, 104(46), 18123–18128. https://doi.org/10.1073/PNAS.0709069104/SUPPL_FILE/09069DATASET2.XLS

Caspersen, J. P., & Pacala, S. W. (2001). Successional diversity and forest ecosystem function. Ecological Research, 16(5), 895-903.

Cattaneo, F., Barbanti, L., Gioacchini, P., Ciavatta, C., & Marzadori, C. (2014). 13C abundance shows effective soil carbon sequestration in Miscanthus and giant reed compared to arable crops under Mediterranean climate. Biology and fertility of soils, 50(7), 1121-1128.

Chan, K.M., Shaw, M.R., Cameron, D.R., Underwood, E.C., & Daily, G.C. (2006). Conservation planning for ecosystem services. PLoS Biol, 4(11), e379.

Chatzimentor, A., Almpanidou, V., Doxa, A., Dimitriadis, C., & Mazaris, A. D. (2021). Projected redistribution of sea turtle foraging areas reveals important sites for conservation. Climate Change Ecology, 2, 100038.

Chee, Y. E. (2004). An ecological perspective on the valuation of ecosystem services. Biological conservation, 120(4), 549-565.

Chen, C., Wang, Y., Jia, J., Mao, L., & Meurk, C. D. (2019). Ecosystem services mapping in practice: a Pasteur's quadrant perspective. Ecosystem Services, 40, 101042.

Cheung, W. W. L., Lam, V. W. Y., Sarmiento, J. L., Kearney, K., Watson, R., & Pauly, D. (2009). Projecting global marine biodiversity impacts under climate change scenarios. Fish and Fisheries, 10(3), 235–251. https://doi.org/10.1111/J.1467-2979.2008.00315.X

Christensen V, Pauly D. (1993). Trophic models of aquatic ecosystems. International Center for Living Aquatic Resources Management, International Council for the Exploration of the Sea, editors. 390 p.(ICLARM conference proceedings).

Christensen, N. L., Bartuska, A. M., Brown, J. H., Carpenter, S., D'Antonio, C., Francis, R., ... & Woodmansee, R. G. (1996). The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological applications, 6(3), 665-691.

CMEMS (Copernicus marine services) - https://resources.marine.copernicus.eu/

Coast Guard registry ('Capitaneria di Porto') - https://www.guardiacostiera.gov.it/

Cohen, E. (1978). The impact of tourism on the physical environment. Annals of Tourism research, 5(2), 215-237.)

Cohen-Kiener, A. (2009). Claiming Earth as Common Ground: The Ecological Crisis Through the Lens of Faith. SkyLight Paths Publishing.

Coll, M., Piroddi, C., Steenbeek, J., Kaschner, K., Lasram, F. B. R., Aguzzi, J., Ballesteros, E., Bianchi, C. N., Corbera, J., Dailianis, T., Danovaro, R., Estrada, M., Froglia, C., Galil, B. S., Gasol, J. M., Gertwage, R., Gil, J., Guilhaumon, F., Kesner-Reyes, K., … Voultsiadou, E. (2010). The Biodiversity of the Mediterranean Sea: Estimates, Patterns, and Threats. PLOS ONE, 5(8), e11842. https://doi.org/10.1371/JOURNAL.PONE.0011842

Coll, M., Santojanni, A., Palomera, I., Tudela, S., & Arneri, E. (2007). An ecological model of the Northern and Central Adriatic Sea: analysis of ecosystem structure and fishing impacts. Journal of Marine Systems, 67(1-2), 119-154.

Colléter, M., Gascuel, D., Albouy, C., Francour, P., Tito de Morais, L., Valls, A., & le Loc'h, F. (2014). Fishing inside or outside? A case studies analysis of potential spillover effect from marine protected areas, using food web models. Journal of Marine Systems, 139, 383–395. https://doi.org/10.1016/j.jmarsys.2014.07.023

Colloca, F., Scarcella, G., & Libralato, S. (2017). Recent trends and impacts of fisheries exploitation on Mediterranean stocks and ecosystems. Frontiers in Marine Science, 4, 244.

Connell, J. H., & Slatyer, R. O. (1977). Mechanisms of succession in natural communities and their role in community stability and organization. The american naturalist, 111(982), 1119-1144.

Costanza, R. (2008). Ecosystem services: multiple classification systems are needed. Biol Conservat, 141, 350-352.

Costanza, R., D'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. v., Paruelo, J., Raskin, R. G., Sutton, P., & van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. Nature 1997 387:6630, 387(6630), 253–260. https://doi.org/10.1038/387253a0

Costanza, R., D'arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'neill, R. v, Paruelo, J., Raskin, R. G., Sutton, P., & van den Belt, M. (1998). The value of ecosystem services: putting the issues in perspective. Ecological Economics, 25.

Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., & Grasso, M. (2017). Twenty years of ecosystem services: How far have we come and how far do we still need to go? Ecosystem Services, 28, 1–16. https://doi.org/10.1016/j.ecoser.2017.09.008

Cowles, H. C. (1899). The ecological relations of the vegetation on the sand dunes of Lake Michigan. I. Geographical relations of the dune floras. Bot. Gaz. 27: 95-117, 167-202, 281-308, 361-391.

Croatian Bureau of Statistics (2018) https://www.dzs.hr/default_e.htm

D’Amen, M., & Azzurro, E. (2020). Lessepsian fish invasion in Mediterranean marine protected areas: a risk assessment under climate change scenarios. ICES Journal of Marine Science, 77(1), 388-397.

Daily, G. (1997). Chapter 1: What are ecosystem services. Scientist, January, 7–8.

Daily, G. C., & Matson, P. A. (2008). Ecosystem services: From theory to implementation. Proceedings of the national academy of sciences, 105(28), 9455-9456.

Darwin, C. (1909). The origin of species. New York: PF Collier & son.

de Groot, R. S., Alkemade, R., Braat, L., Hein, L., & Willemen, L. (2010). Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity, 7(3), 260–272. https://doi.org/10.1016/J.ECOCOM.2009.10.006

de Groot, R. S., Wilson, M. A., & Boumans, R. M. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological economics, 41(3), 393-408.

de Juan, S., & Lleonart, J. (2010). A conceptual framework for the protection of vulnerable habitats impacted by fishing activities in the Mediterranean high seas. Ocean and Coastal Management, 53(11), 717–723. https://doi.org/10.1016/j.ocecoaman.2010.10.005

DEM (Digital Elevation Model)- Copernicus Land monitoring service: https://land.copernicus.eu/imagery-in-situ/eu-dem/eu-dem-v1.1

Deng, X., Li, Z., & Gibson, J. (2016). A review on trade-off analysis of ecosystem services for sustainable land-use management. Journal of Geographical Sciences, 26(7), 953-968.

Depellegrin, D., Menegon, S., Farella, G., Ghezzo, M., Gissi, E., Sarretta, A., Venier, C., & Barbanti, A. (2017). Multi-objective spatial tools to inform maritime spatial planning in the Adriatic Sea. Science of the Total Environment, 609, 1627–1639. https://doi.org/10.1016/j.scitotenv.2017.07.264

Deser, C., Phillips, A., Bourdette, V., & Teng, H. (2012). Uncertainty in climate change projections: the role of internal variability. Climate dynamics, 38(3), 527-546.

Devictor, V., Mouillot, D., Meynard, C., Jiguet, F., Thuiller, W., & Mouquet, N. (2010). Spatial mismatch and congruence between taxonomic, phylogenetic and functional diversity: The need for integrative conservation strategies in a changing world. Ecology Letters, 13(8), 1030–1040. https://doi.org/10.1111/j.1461-0248.2010.01493.x

Díaz, S., & Cabido, M. (2001). Vive la différence: plant functional diversity matters to ecosystem processes. Trends in ecology & evolution, 16(11), 646-655.

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., Hill, R., Chan, K. M. A., Baste, I. A., Brauman, K. A., Polasky, S., Church, A., Lonsdale, M., Larigauderie, A., Leadley, P. W., van Oudenhoven, A. P. E., van der Plaat, F., Schröter, M., Lavorel, S., … Shirayama, Y. (2018). Assessing nature's contributions to people: Recognizing culture, and diverse sources of knowledge, can improve assessments. Science, 359(6373), 270–272. https://doi.org/10.1126/SCIENCE.AAP8826/SUPPL_FILE/AAP8826-DIAZ-SM.PDF

Diaz-Amela, E. L. E. N. A., Marba, N., & Duarte, C. M. (2007). Consequences of Mediterranean warming events in seagrass (Posidonia oceanica) flowering records. Global change biology, 13(1), 224-235.

Dick, J., Turkelboom, F., Woods, H., Iniesta-Arandia, I., Primmer, E., Saarela, S. R., ... & Zulian, G. (2018). Stakeholders' perspectives on the operationalisation of the ecosystem service concept: Results from 27 case studies. Ecosystem services, 29, 552-565.

Diffenbaugh, N. S., Pal, J. S., Giorgi, F., & Gao, X. (2007). Heat stress intensification in the Mediterranean climate change hotspot. Geophysical Research Letters, 34(11).

Distretto di Pesca Nord Adriatico - Analisi della filiera dell'acquacoltura – 2015 Osservatorio Socio Economico della Pesca e dell'Acquacoltura - https://www.venetoagricoltura.org/2015/06/temi/il-distretto-di-pesca-nord-adriatico-2015/

Dolbeth, M., Vendel, A. L., Pessanha, A., & Patrício, J. (2016). Functional diversity of fish communities in two tropical estuaries subjected to anthropogenic disturbance. Marine Pollution Bulletin, 112(1-2), 244-254.

Doxa, A., Almpanidou, V., Katsanevakis, S., Queirós, A. M., Kaschner, K., Garilao, C., Kesner-Reyes, K., & Mazaris, A. D. (2022). 4D marine conservation networks: Combining 3D prioritization of present and future biodiversity with climatic refugia. Global Change Biology, 28(15), 4577–4588. https://doi.org/10.1111/gcb.16268

Drius, M., Jones, L., Marzialetti, F., de Francesco, M. C., Stanisci, A., & Carranza, M. L. (2019). Not just a sandy beach. The multi-service value of Mediterranean coastal dunes. Science of the total environment, 668, 1139-1155.Giani et al., 2001

du Pontavice, H., Gascuel, D., Reygondeau, G., Maureaud, A., & Cheung, W. W. L. (2020). Climate change undermines the global functioning of marine food webs. Global Change Biology, 26(3), 1306–1318. https://doi.org/10.1111/gcb.14944

du Pontavice, H., Gascuel, D., Reygondeau, G., Stock, C., & Cheung, W. W. L. (2021). Climate-induced decrease in biomass flow in marine food webs may severely affect predators and ecosystem production. Global Change Biology, 27(11), 2608–2622. https://doi.org/10.1111/GCB.15576

Duarte, C. M., & Cebrián, J. (1996). The fate of marine autotrophic production. Limnology and oceanography, 41(8), 1758-1766.

Duffy, G. A., & Chown, S. L. (2017). Explicitly integrating a third dimension in marine species distribution modelling. Marine Ecology Progress Series, 564, 1–8. https://doi.org/10.3354/meps12011

Duffy, S., Shackelton, L. A., & Holmes, E. C. (2008). Rates of evolutionary change in viruses: patterns and determinants. Nature Reviews Genetics, 9(4), 267-276.

Eastwood, A., Brooker, R., Irvine, R. J., Artz, R. R. E., Norton, L. R., Bullock, J. M., Ross, L., Fielding, D., Ramsay, S., Roberts, J., Anderson, W., Dugan, D., Cooksley, S., & Pakeman, R. J. (2016). Does nature conservation enhance ecosystem services delivery? Ecosystem Services, 17(2016), 152–162. https://doi.org/10.1016/j.ecoser.2015.12.001

Eddy, T. D., Bernhardt, J. R., Blanchard, J. L., Cheung, W. W. L., Colléter, M., du Pontavice, H., Fulton, E. A., Gascuel, D., Kearney, K. A., Petrik, C. M., Roy, T., Rykaczewski, R. R., Selden, R., Stock, C. A., Wabnitz, C. C. C., & Watson, R. A. (2021). Energy Flow Through Marine Ecosystems: Confronting Transfer Efficiency. Trends in Ecology & Evolution, 36(1), 76–86. https://doi.org/10.1016/J.TREE.2020.09.006

Edwards, M., & Richardson, A. J. (2004). Impact of climate change on marine pelagic phenology and trophic mismatch. Nature, 430(7002), 881-884.

Egoh, B., Reyers, B., Rouget, M., Bode, M., & Richardson, D. M. (2009). Spatial congruence between biodiversity and ecosystem services in South Africa. Biological conservation, 142(3), 553-562.

El Hourany, R., Mejia, C., Faour, G., Crépon, M., & Thiria, S. (2021). Evidencing the impact of climate change on the phytoplankton community of the Mediterranean Sea through a bioregionalization approach. Journal of Geophysical Research: Oceans, 126(4), e2020JC016808.

Elise, S., Urbina-Barreto, I., Pinel, R., Mahamadaly, V., Bureau, S., Penin, L., Adjeroud, M., Kulbicki, M., & Bruggemann, J. H. (2019). Assessing key ecosystem functions through soundscapes: A new perspective from coral reefs. Ecological Indicators, 107(April). https://doi.org/10.1016/j.ecolind.2019.105623

FAO – Food and Agricultural Organisation of United Nations - https://www.fao.org/about/en/

FAO-GFCM (2018). Data Collection Reference Framework (DCRF).Version: 21.2

Farella, G., Menegon, S., Fadini, A., Depellegrin, D., Manea, E., Perini, L., & Barbanti, A. (2020). Incorporating ecosystem services conservation into a scenario-based MSP framework: An Adriatic case study. Ocean & Coastal Management, 193, 105230.

Fennell, D. A., & Cooper, C. (2020). Sustainable tourism. Channel View Publications.

Fiori, E., Zavatarelli, M., Pinardi, N., Mazziotti, C., & Ferrari, C. R. (2016). Observed and simulated trophic index (TRIX) values for the Adriatic Sea basin. Natural Hazards and Earth System Sciences, 16(9), 2043–2054. https://doi.org/10.5194/nhess-16-2043-2016

Fischer, A., & Eastwood, A. (2016). Coproduction of ecosystem services as human–nature interactions—An analytical framework. Land use policy, 52, 41-50.

Fisher, B., Turner, K., Zylstra, M., Brouwer, R., De Groot, R., Farber, S., ... & Jefferiss, P. (2008). Ecosystem services and economic theory: integration for policy-relevant research. Ecological applications, 18(8), 2050-2067.

Fisher, B., Turner, R. K., & Morling, P. (2009). A systems approach to definitions and principles for ecosystem services. Ecological Economics, 18, 2050-2067.

FishStat Plus (FAO) - https://www.fao.org/fishery/en/statistics/software/fishstatj/en

Flynn, D. F. B., Mirotchnick, N., Jain, M., Palmer, M. I., & Naeem, S. (2011). Functional and phylogenetic diversity as predictors of biodiversity-Ecosystem-function relationships. Ecology, 92(8), 1573–1581. https://doi.org/10.1890/10-1245.1

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., ... & Snyder, P. K. (2005). Global consequences of land use. science, 309(5734), 570-574.

Fortuna, C. M., Cañadas, A., Holcer, D., Brecciaroli, B., Donovan, G. P., Lazar, B., ... & Mackelworth, P. C. (2018). The coherence of the European Union marine Natura 2000 network for wide-ranging charismatic species: a Mediterranean case study. Frontiers in Marine Science, 5, 356.Prioli's reports (18th Censimento Nazionale sulla molluschicoltura del consorzio Unimar, Giuseppe Prioli

Friedrich, L. A., Glegg, G., Fletcher, S., Dodds, W., Philippe, M., & Bailly, D. (2020). Using ecosystem service assessments to support participatory marine spatial planning. Ocean and Coastal Management, 188(October 2019), 105121. https://doi.org/10.1016/j.ocecoaman.2020.105121

Froese R., Pauly D. (Eds.), 2017. FishBase. World Wide Web electronic publication. Retrieved from www.fishbase.org (accessed on 17 April 2017).

Gambaiani, D. D., Mayol, P., Isaac, S. J., & Simmonds, M. P. (2009). Potential impacts of climate change and greenhouse gas emissions on Mediterranean marine ecosystems and cetaceans. Journal of the Marine biological Association of the United Kingdom, 89(1), 179-201.

Garrabou, J., Gómez-Gras, D., Medrano, A., Cerrano, C., Ponti, M., Schlegel, R., Bensoussan, N., Turicchia, E., Sini, M., Gerovasileiou, V., Teixido, N., Mirasole, A., Tamburello, L., Cebrian, E., Rilov, G., Ledoux, J. B., Souissi, J. ben, Khamassi, F., Ghanem, R., … Harmelin, J. G. (2022). Marine heatwaves drive recurrent mass mortalities in the Mediterranean Sea. Global Change Biology, 28(19), 5708–5725. https://doi.org/10.1111/gcb.16301

Gascuel, D., Bozec, Y. M., Chassot, E., Colomb, A., & Laurans, M. (2005). The trophic spectrum: theory and application as an ecosystem indicator. ICES Journal of Marine Science, 62(3), 443–452. https://doi.org/10.1016/J.ICESJMS.2004.12.013

Gascuel, D., Guénette, S., & Pauly, D. (2011). The trophic-level-based ecosystem modelling approach: theoretical overview and practical uses. ICES Journal of Marine Science, 68(7), 1403–1416. https://doi.org/10.1093/ICESJMS/FSR062

GEBCO (General Bathymetric Chart of the Ocean): https://www.gebco.net/data_and_products/gridded_bathymetry_data/ EMODnet (European Marine Observation and Data Network): https://www.emodnet.eu/en/map-week-%E2%80%93-posidonia-oceanica-distribution-seagrass-species

Gilman, S. E., Urban, M. C., Tewksbury, J., Gilchrist, G. W., & Holt, R. D. (2010). A framework for community interactions under climate change. Trends in ecology & evolution, 25(6), 325-331.

Gómez-Baggethun, E., De Groot, R., Lomas, P. L., & Montes, C. (2010). The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecological economics, 69(6), 1209-1218.

Gratani, L., Varone, L., Ricotta, C., & Catoni, R. (2013). Mediterranean shrublands carbon sequestration: environmental and economic benefits. Mitigation and Adaptation Strategies for Global Change, 18(8), 1167-1182.Campagne et al., 2015

Grêt-Regamey, A., Weibel, B., Bagstad, K. J., Ferrari, M., Geneletti, D., Klug, H., ... & Tappeiner, U. (2014). On the effects of scale for ecosystem services mapping. PloS one, 9(12), e112601.

Grinnell, J. (1917). The niche-relationships of the California Thrasher. The Auk, 34(4), 427-433.

Guisande, C., Patti, B., Vaamonde, A., Manjarrés-Hernández, A., Pelayo-Villamil, P., García-Roselló, E., & Granado-Lorencio, C. (2013). Factors affecting species richness of marine elasmobranchs. Biodiversity and conservation, 22(8), 1703-1714.

Gupta, A. S., Jourdain, N. C., Brown, J. N., & Monselesan, D. (2013). Climate drift in the CMIP5 models. Journal of Climate, 26(21), 8597-8615.

Habel, S., Fletcher, C. H., Anderson, T. R., & Thompson, P. R. (2020). Sea-level rise induced multi-mechanism flooding and contribution to urban infrastructure failure. Scientific reports, 10(1), 1-12.

Haines-Young, R., & Potschin, M. (2010). The links between biodiversity, ecosystem services and human well-being. Ecosystem Ecology: a new synthesis, 1, 110-139.

Haines-Young, R., & Potschin, M. (2011). Common international classification of ecosystem services (CICES): 2011 Update. Nottingham: Report to the European Environmental Agency.

Hattam, C., Broszeit, S., Langmead, O., Praptiwi, R. A., Lim, V. C., Creencia, L. A., Tran, D. H., Maharja, C., Mitra Setia, T., Wulandari, P., Sugardjito, J., Javier, J., Jose, E., Gajardo, L. J., Then, A. Y. H., Affendi, Y. A., Johari, S., Justine, E. V., Syed Hussein, M. A., … Austen, M. (2021). A matrix approach to tropical marine ecosystem service assessments in South east Asia. Ecosystem Services, 51(June), 101346. https://doi.org/10.1016/j.ecoser.2021.101346

Hauck, J., Görg, C., Varjopuro, R., Ratamäki, O., Maes, J., Wittmer, H., & Jax, K. (2013). “Maps have an air of authority”: potential benefits and challenges of ecosystem service maps at different levels of decision making. Ecosystem Services, 4, 25-32.

Hawkins, E., & Sutton, R. (2009). The potential to narrow uncertainty in regional climate predictions. Bulletin of the American Meteorological Society, 90(8), 1095-1108.

Hazen, E. L., Suryan, R. M., Santora, J. A., Bograd, S. J., Watanuki, Y., & Wilson, R. P. (2013). Scales and mechanisms of marine hotspot formation. Marine Ecology Progress Series, 487, 177–183. https://doi.org/10.3354/meps10477

Hein, L., Van Koppen, K., De Groot, R. S., & Van Ierland, E. C. (2006). Spatial scales, stakeholders and the valuation of ecosystem services. Ecological economics, 57(2), 209-228.

Heink, U., Hauck, J., Jax, K., & Sukopp, U. (2016). Requirements for the selection of ecosystem service indicators–The case of MAES indicators. Ecological Indicators, 61, 18-26.

Hilborn, R. (2011). Future directions in ecosystem based fisheries management: A personal perspective. Fisheries Research, 108(2–3), 235–239. https://doi.org/10.1016/j.fishres.2010.12.030

Hobday, A. J., Alexander, L. v., Perkins, S. E., Smale, D. A., Straub, S. C., Oliver, E. C. J., Benthuysen, J. A., Burrows, M. T., Donat, M. G., Feng, M., Holbrook, N. J., Moore, P. J., Scannell, H. A., sen Gupta, A., & Wernberg, T. (2016). A hierarchical approach to defining marine heatwaves. Progress in Oceanography, 141, 227–238. https://doi.org/10.1016/j.pocean.2015.12.014

Holmlund, C. M., & Hammer, M. (1999). Ecosystem services generated by fish populations. Ecological economics, 29(2), 253-268.

Hölting, L., Beckmann, M., Volk, M., & Cord, A. F. (2019). Multifunctionality assessments–More than assessing multiple ecosystem functions and services? A quantitative literature review. Ecological Indicators, 103, 226-235.

Hutchinson, G. E. (1959). Homage to Santa Rosalia or why are there so many kinds of animals?. The American Naturalist, 93(870), 145-159.

ICES (International Council for the Exploration of the Sea) (2014) - Report of the workshop to develop recom mendations for potentially useful Food Web Indicators (WKFooWI). ICES CM 2014\ACOM: 48

ICZM-2002/413/EC - Integrated coastal zone management: http://ec.europa.eu/environment/iczm/situation.htm, accessed on 29 May 2020)

Jax, K. (2005). Function and “functioning” in ecology: What does it mean? Oikos, 111(3), 641–648. https://doi.org/10.1111/j.1600-0706.2005.13851.x

Jax, K., Furman, E., Saarikoski, H., Barton, D. N., Delbaere, B., Dick, J., Duke, G., Görg, C., Gómez-Baggethun, E., Harrison, P. A., Maes, J., Pérez-Soba, M., Saarela, S. R., Turkelboom, F., van Dijk, J., & Watt, A. D. (2018). Handling a messy world: Lessons learned when trying to make the ecosystem services concept operational. Ecosystem Services, 29, 415–427. https://doi.org/10.1016/j.ecoser.2017.08.001

Jennings, S., Warr, K. J., & Mackinson, S. (2002). Use of size-based production and stable isotope analyses to predict trophic transfer efficiencies and predator-prey body mass ratios in food webs. Marine Ecology Progress Series, 240(July 2014), 11–20. https://doi.org/10.3354/meps240011

Jones, J. B. (1992). Environmental impact of trawling on the seabed: a review. New Zealand Journal of Marine and Freshwater Research, 26(1), 59-67.

Jones, K. E., & Paramor, O. A. L. (2010). Inter-disciplinarity in ecosystems research: developing social robustness in environmental science. Ecosystem Ecology: A New Synthesis. In: BES Ecological Review Series, Cambridge University Press, Cambridge, UK.

Jones, L., Norton, L., Austin, Z., Browne, A. L., Donovan, D., Emmett, B. A., ... & Willis, G. F. (2016). Stocks and flows of natural and human-derived capital in ecosystem services. Land use policy, 52, 151-162.

Jukic-Peladic, S., Vrgoc, N., Krstulovic-Sifner, S., Piccinetti, C., Piccinetti-Manfrin, G., Marano, G., & Ungaro, N. (2001). Long-term changes in demersal resources of the Adriatic Sea: comparison between trawl surveys carried out in 1948 and 1998. Fisheries research, 53(1), 95-104.

Karachle, P. K., & Stergiou, K. I. (2017). An update on the feeding habits of fish in the Mediterranean Sea (2002-2015). Mediterranean Marine Science, 18(1), 43-52.

Katsanevakis, S., Coll, M., Fraschetti, S., Giakoumi, S., Goldsborough, D., Mačić, V., Mackelworth, P., Rilov, G., Stelzenmüller, V., Albano, P. G., Bates, A. E., Bevilacqua, S., Gissi, E., Hermoso, V., Mazaris, A. D., Pita, C., Rossi, V., Teff-Seker, Y., & Yates, K. (2020). Twelve Recommendations for Advancing Marine Conservation in European and Contiguous Seas. Frontiers in Marine Science, 7(October), 1–18. https://doi.org/10.3389/fmars.2020.565968

Katsanevakis, S., Rilov, G., & Edelist, D. (2018). Impacts of marine invasive alien species on European fisheries and aquaculture - plague or boon? Engaging Marine Scientists and Fishers to Share Knowledge and Perceptions - Early Lessons. CIESM Monograph N° 50, 50, 125–132.

Kelley, B. P., Sharan, R., Karp, R. M., Sittler, T., Root, D. E., Stockwell, B. R., & Ideker, T. (2003). Conserved pathways within bacteria and yeast as revealed by global protein network alignment. Proceedings of the National Academy of Sciences, 100(20), 11394-11399.

Kiørboe, T., Saiz, E., Tiselius, P., & Andersen, K. H. (2018). Adaptive feeding behavior and functional responses in zooplankton. Limnology and Oceanography, 63(1), 308-321.

Klain, S. C., Satterfield, T. A., & Chan, K. M. A. (2014). What matters and why? Ecosystem services and their bundled qualities. Ecological Economics, 107, 310–320. https://doi.org/10.1016/j.ecolecon.2014.09.003

Kortsch, S., Primicerio, R., Fossheim, M., Dolgov, A. V., & Aschan, M. (2015). Climate change alters the structure of arctic marine food webs due to poleward shifts of boreal generalists. Proceedings of the Royal Society B: Biological Sciences, 282(1814), 20151546.

Kozic, I., & Mikulic, J. (2011), "Implementing an Indicator-System for Evaluating and Monitoring the Sustainability of Tourism in the Republic of Croatia", Privredna Kretanja i Ekonomska Politika, Manuscript accepted for publication.

Kremen, C. (2005) Managing ecosystem services: What do we need to know about their ecology? Ecol. Lett. 8, 468–479.

Kroll, F., Müller, F., Haase, D., & Fohrer, N. (2012). Rural–urban gradient analysis of ecosystem services supply and demand dynamics. Land use policy, 29(3), 521-535.

Kyprioti, A., Almpanidou, V., Chatzimentor, A., Katsanevakis, S., Mazaris, A. D. (2021). Is the current Mediterranean network of marine protected areas resilient to climate change? Science of the Total Environment, 79, 148397.

La Notte, A., D'Amato, D., Mäkinen, H., Paracchini, M. L., Liquete, C., Egoh, B., ... & Crossman, N. D. (2017). Ecosystem services classification: A systems ecology perspective of the cascade framework. Ecological indicators, 74, 392-402.

Lamarque, P., Quétier, F., & Lavorel, S. (2011). The diversity of the ecosystem services concept and its implications for their assessment and management. Comptes rendus biologies, 334(5-6), 441-449.

Lascaratos, A., Roether, W., Nittis, K., & Klein, B. (1999). Recent changes in deep water formation and spreading in the Eastern Mediterranean Sea: A review. Progress in Oceanography, 44(1–3), 5–36. https://doi.org/10.1016/S0079-6611(99)00019-1

Lewontin, R. (1969) Meaning of Stability. Brookhaven Sym Biol, 13

Liang, J., Crowther, T. W., Picard, N., Wiser, S., Zhou, M., Alberti, G., Schulze, E. D., McGuire, A. D., Bozzato, F., Pretzsch, H., De-Miguel, S., Paquette, A., Hérault, B., Scherer-Lorenzen, M., Barrett, C. B., Glick,

H. B., Hengeveld, G. M., Nabuurs, G. J., Pfautsch, S., … Reich, P. B. (2016). Positive biodiversity-productivity relationship predominant in global forests. Science (New York, N.Y.), 354(6309). https://doi.org/10.1126/SCIENCE.AAF8957

Libralato, S., Caccin, A., & Pranovi, F. (2015). Modeling species invasions using thermal and trophic niche dynamics under climate change. Frontiers in Marine Science, 2, 29.

Libralato, S., Pranovi, F., Stergiou, K. I., & Link, J. S. (2014). Trophodynamics in marine ecology: 70 years after Lindeman. Marine Ecology Progress Series, 512, 1–7. https://doi.org/10.3354/meps11033

Lindegren, M., Holt, B. G., MacKenzie, B. R., & Rahbek, C. (2018). A global mismatch in the protection of multiple marine biodiversity components and ecosystem services. Scientific Reports, 8(1), 1–8. https://doi.org/10.1038/s41598-018-22419-1

Lindeman, R. L. (1942). The Trophic-Dynamic Aspect of Ecology. Ecology, 23(4), 399–417. https://doi.org/10.2307/1930126

Link, J. S. (2005). Translating ecosystem indicators into decision criteria. ICES Journal of Marine Science, 62(3), 569-576.

Liquete, C., Zulian, G., Delgado, I., Stips, A., & Maes, J. (2013). Assessment of coastal protection as an ecosystem service in Europe. Ecological Indicators, 30, 205-217.data from ISTAT

Loreau, M. (1998). Biodiversity and ecosystem functioning: a mechanistic model. Proceedings of the National Academy of Sciences, 95(10), 5632-5636.

Lovelock, J. E. 1972. Gaia as seen through the atmosphere. Atmospheric Environment 6, 579- 580.

Luck, G. W., Daily, G. C., & Ehrlich, P. R. (2003). Population diversity and ecosystem services. Trends in Ecology & Evolution, 18(7), 331-336.

M. Ecosystem Assessment (2005). Ecosystems and human well-being: wetlands and water. World Resources Institute.

Mace, G. M., Norris, K., & Fitter, A. H. (2012). Biodiversity and ecosystem services: a multilayered relationship. Trends in Ecology & Evolution, 27(1), 19–26. https://doi.org/10.1016/J.TREE.2011.08.006

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J. P., ... & Bidoglio, G. (2012). Mapping ecosystem services for policy support and decision making in the European Union. Ecosystem services, 1(1), 31-39.

Manea, E., Bianchelli, S., Fanelli, E., Danovaro, R., & Gissi, E. (2020). Towards an ecosystem- based marine spatial planning in the deep Mediterranean Sea. Science of the Total Environment, 715, 136884.

Manea, E., di Carlo, D., Depellegrin, D., Agardy, T., & Gissi, E. (2019). Multidimensional assessment of supporting ecosystem services for marine spatial planning of the Adriatic Sea. Ecological Indicators, 101, 821–837. https://doi.org/10.1016/j.ecolind.2018.12.017

Mareografico - https://www.mareografico.it/

MArine Protected Areas in the MEDiterranean - https://www.mapamed.org/

Martín-Vélez, V., & Abellán, P. (2022). Effects of climate change on the distribution of threatened invertebrates in a Mediterranean hotspot. Insect Conservation and Diversity.

Maureaud, A., Gascuel, D., Colléter, M., Palomares, M. L. D., du Pontavice, H., Pauly, D., & Cheung, W. W. L. (2017). Global change in the trophic functioning of marine food webs. PLoS ONE, 12(8), 1–21. https://doi.org/10.1371/journal.pone.0182826

Mayer, A., Kaufmann, L., Kalt, G., Matej, S., Theurl, M. C., Morais, T. G., Leip, A., & Erb, K. H. (2021). Applying the Human Appropriation of Net Primary Production framework to map provisioning ecosystem services and their relation to ecosystem functioning across the European Union. Ecosystem Services, 51, 101344. https://doi.org/10.1016/j.ecoser.2021.101344

McLeod, A., & Reeve, M. (2005). The health status of quota refugees screened by New Zealand's Auckland Public Health Service between 1995 and 2000. The New Zealand Medical Journal (Online), 118(1224).

Meacham, M., Queiroz, C., Norström, A. V., & Peterson, G. D. (2016). Social-ecological drivers of multiple ecosystem services: what variables explain patterns of ecosystem services across the Norrström drainage basin?. Ecology and Society, 21(1).

Molinero, J. C., Ibanez, F., Nival, P., Buecher, E., & Souissi, S. (2005). North Atlantic climate and

Möllmann, C., Cormon, X., Funk, S., Otto, S. A., Schmidt, J. O., Schwermer, H., ... & Quaas, M. (2021). Tipping point realized in cod fishery. Scientific Reports, 11(1), 1-12.

Monserrat, M., Comeau, S., Verdura, J., Alliouane, S., Spennato, G., Priouzeau, F., ... & Mangialajo, L. (2022). Climate change and species facilitation affect the recruitment of macroalgal marine forests. Scientific reports, 12(1), 1-12.

Montoya, J. M., & Raffaelli, D. G. (Eds.). (2010). The effects of climate change on biotic interactions and ecosystem services. Royal Society.

Morgera, E. (2020). Corporate accountability in international environmental law (No. 2nd). Oxford University Press.

Moschella, P., & Commission Internationale pour l'Exploration Scientifique de la Mer Mediterranee- CIESM, Monaco. (2008). The new CIESM Tropicalization Programme- effects of climate warming on Mediterranean key taxa. In CIESM Workshop Monographs (Vol. 35, pp. 47-50). CIESM, Monaco.

Mouillot, D., Albouy, C., Guilhaumon, F., Lasram, F. B. R., Coll, M., Devictor, V., Meynard, C. N., Pauly, D., Tomasini, J. A., Troussellier, M., Velez, L., Watson, R., Douzery, E. J. P., & Mouquet, N. (2011). Protected and threatened components of fish biodiversity in the Mediterranean Sea. Current Biology, 21, 1044– 1050.

Moullec, F., Barrier, N., Drira, S., Guilhaumon, F., Hattab, T., Peck, M. A., & Shin, Y. J. (2022). Using species distribution models only may underestimate climate change impacts on future marine biodiversity. Ecological Modelling, 464, 109826. https://doi.org/10.1016/J.ECOLMODEL.2021.109826

Moullec, F., Velez, L., Verley, P., Barrier, N., Ulses, C., Carbonara, P., Esteban, A., Follesa, C., Gristina, M., Jadaud, A., Ligas, A., Díaz, E. L., Maiorano, P., Peristeraki, P., Spedicato, M. T., Thasitis, I., Valls, M., Guilhaumon, F., & Shin, Y. J. (2019). Capturing the big picture of Mediterranean marine biodiversity with an end-to-end model of climate and fishing impacts. Progress in Oceanography, 178(December 2018). https://doi.org/10.1016/j.pocean.2019.102179

Mumby, P. J., Broad, K., Brumbaugh, D. R., Dahlgren, C. P., Harborne, A. R., Hastings, A., Holmes, K. E., Kappel, C. v., Micheli, F., & Sanchirico, J. N. (2008). Coral Reef Habitats as Surrogates of Species, Ecological Functions, and Ecosystem Services. Conservation Biology, 22(4), 941–951. https://doi.org/10.1111/J.1523-1739.2008.00933.X

Naeem, S., & Wright, J. P. (2003). Disentangling biodiversity effects on ecosystem functioning: deriving solutions to a seemingly insurmountable problem. Ecology letters, 6(6), 567-579.

Nahuelhual, L., Vergara, X., Kusch, A., Campos, G., & Droguett, D. (2017). Mapping ecosystem services for marine spatial planning: Recreation opportunities in Sub-Antarctic Chile. Marine Policy, 81(December 2016), 211–218. https://doi.org/10.1016/j.marpol.2017.03.038

Naidoo, R., & Ricketts, T. H. (2006). Mapping the economic costs and benefits of conservation. PLoS Biol, 4(11), e360.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R. E., Lehner, B., Malcolm, T. R., & Ricketts, T. H. (2008). Global mapping of ecosystem services and conservation priorities. Proceedings of the National Academy of Sciences of the United States of America, 105(28), 9495–9500. https://doi.org/10.1073/pnas.0707823105

Newbold, T., Tittensor, D. P., Harfoot, M. B., Scharlemann, J. P., & Purves, D. W. (2020). Non-linear changes in modelled terrestrial ecosystems subjected to perturbations. Scientific reports, 10(1), 1-10.

Niner, H. J., Barut, N. C., Baum, T., Diz, D., del Pozo, D. L., Laing, S., ... & Rees, S. (2022). Issues of context, capacity and scale: Essential conditions and missing links for a sustainable blue economy. Environmental Science & Policy, 130, 25-35.

Odum, E. P. (1969). No Title. 164, 262–270. https://doi.org/10.1126/SCIENCE.164.3877.262/ASSET/ABCC476F-CE12-4FB2-920F-E1BE32FB5B4F/ASSETS/SCIENCE.164.3877.262.FP.PNG

Ogie, R. I., Holderness, T., Dunn, S., & Turpin, E. (2018). Assessing the vulnerability of hydrological infrastructure to flood damage in coastal cities of developing nations. Computers, Environment and Urban Systems, 68, 97-109.

Oguz, T., Malanotte-Rizzoli, P., & Ducklow, H. W. (2001). Simulations of phytoplankton seasonal cycle with multi-level and multi-layer physical-ecosystem models: the Black Sea example. Ecological Modelling, 144(2-3), 295-314.

Ostrom, E. (2009) A general framework for analyzing sustainability of social-ecological systems. Science, 325, 419–22.

Palomares M.L.D., Pauly D. (1998). Predicting food consumption of fish populations as functions of mortality,food type, morphometrics, temperature and salinity. Mar Freshw Res; 49(5):447.

Parmesan, C. (2006). Ecological and evolutionary responses to recent climate change. Annual review of ecology, evolution, and systematics, 637-669.

Parmesan, C., & Yohe, G. (2003). A globally coherent fingerprint of climate change impacts across natural systems. nature, 421(6918), 37-42.

Pecuchet, L., Lindegren, M., Hidalgo, M., Delgado, M., Esteban, A., Fock, H. O., ... & Payne, M. R. (2017). From traits to life-history strategies: Deconstructing fish community composition across European seas. Global Ecology and Biogeography, 26(7), 812-822.

Pedrono, M., Locatelli, B., Ezzine-de-Blas, D., Pesche, D., Morand, S., & Binot, A. (2016). Impact of climate change on ecosystem services. In Climate change and agriculture worldwide (pp. 251-261). Springer, Dordrecht.

Peng, J., Tian, L., Liu, Y., Zhao, M., & Wu, J. (2017). Ecosystem services response to urbanization in metropolitan areas: Thresholds identification. Science of the Total Environment, 607, 706-714.

Perini, L., Calabrese, L., Luciani, P., Olivieri, M., Galassi, G., & Spada, G. (2017). Sea-level rise along the Emilia-Romagna coast (Northern Italy) in 2100: scenarios and impacts. Natural Hazards and Earth System Sciences, 17(12), 2271-2287.

Peterson, M. J., Hall, D. M., Feldpausch-Parker, A. M., & Peterson, T. R. (2010). Obscuring Ecosystem Function with Application of the Ecosystem Services Concept. Conservation Biology, 24(1), 113–119. https://doi.org/10.1111/J.1523-1739.2009.01305.X

Petz, K., & van Oudenhoven, A. P. (2012). Modelling land management effect on ecosystem functions and services: a study in the Netherlands. International Journal of Biodiversity Science, Ecosystem Services & Management, 8(1-2), 135-155.

Pezdevšek Malovrh, Š., Paletto, A., Posavec, S., Dobšinská, Z., Đorđević, I., Marić, B., ... & Laktić, T. (2019). Evaluation of the operational environment factors of Nature conservation policy implementation: Cases of selected EU and non-EU countries. Forests, 10(12), 1099.

Pinardi, N., Arneri, E., Crise, A., Ravaioli, M., & Zavatarelli, M. (2006). The physical, sedimentary and ecological structure and variability of shelf areas in the Mediterranean sea (27). The sea, 14, 1243-330.

Poloczanska, E. S., Brown, C. J., Sydeman, W. J., Kiessling, W., Schoeman, D. S., Moore, P. J., Brander, K., Bruno, J. F., Buckley, L. B., Burrows, M. T., Duarte, C. M., Halpern, B. S., Holding, J., Kappel, C. v., O’Connor, M. I., Pandolfi, J. M., Parmesan, C., Schwing, F., Thompson, S. A., & Richardson, A. J. (2013). Global imprint of climate change on marine life. Nature Climate Change, 3(10), 919–925. https://doi.org/10.1038/nclimate1958

Poloczanska, E. S., Brown, C. J., Sydeman, W. J., Kiessling, W., Schoeman, D. S., Moore, P. J., Brander, K., Bruno, J. F., Buckley, L. B., Burrows, M. T., Duarte, C. M., Halpern, B. S., Holding, J., Kappel, C. v., O'Connor, M. I., Pandolfi, J. M., Parmesan, C., Schwing, F., Thompson, S. A., & Richardson, A. J. (2013). Global imprint of climate change on marine life. Nature Climate Change, 3(10), 919–925. https://doi.org/10.1038/nclimate1958

Pörtner, H.-O., Roberts, D. C., Masson-Delmotte, V., Zhai, P., Tignor, M., Poloczanska, E., & Mintenbeck, K. (2019). Technical summary: the ocean and cryosphere in a changing climate. September.

Pörtner, H.O., Scholes, R.J., Agard, J., Archer, E., Arneth, A., Bai, X., Barnes, D., Burrows, M., Chan, L., Cheung, W.L., Diamond, S., Donatti, C., Duarte, C., Eisenhauer, N., Foden, W., Gasalla, M. A., Handa, C., Hickler, T., Hoegh-Guldberg, O., Ichii, K., Jacob, U., Insarov, G., Kiessling, W., Leadley, P., Leemans, R., Levin, L., Lim, M., Maharaj, S., Managi, S., Marquet, P. A., McElwee, P., Midgley, G., Oberdorff, T., Obura, D., Osman, E., Pandit, R., Pascual, U., Pires, A. P. F., Popp, A., Reyes- García, V., Sankaran, M., Settele, J., Shin, Y. J., Sintayehu, D. W., Smith, P., Steiner, N., Strassburg, B., Sukumar, R., Trisos, C., Val, A.L., Wu, J., Aldrian, E., Parmesan, C., Pichs-Madruga, R., Roberts, D.C., Rogers, A.D., Díaz, S., Fischer, M., Hashimoto, S., Lavorel, S., Wu, N., Ngo, H.T. (2021). IPBES-IPCC co-sponsored workshop report on biodiversity and climate change; IPBES and IPCC. DOI:10.5281/zenodo.4782538.

Potschin, M. B., & Haines-Young, R. H. (2011). Ecosystem services: Exploring a geographical perspective. Http://Dx.Doi.Org/10.1177/0309133311423172, 35(5), 575–594. https://doi.org/10.1177/0309133311423172

Potschin-Young, M., Haines-Young, R., Görg, C., Heink, U., Jax, K., & Schleyer, C. (2018). Understanding the role of conceptual frameworks: Reading the ecosystem service cascade. Ecosystem Services, 29, 428–440. https://doi.org/10.1016/j.ecoser.2017.05.015

Poulain, P. M., Barbanti, R., Font, J., Cruzado, A., Millot, C., Gertman, I., Griffa, A., Molcard, A., Rupolo, V., le Bras, S., & de La Villeon, L.

P. (2007). MedArgo: A drifting profiler program in the Mediterranean Sea. Ocean Science, 3(3), 379–395. https://doi.org/10.5194/os-3-379-2007

Pranovi, F., Libralato, S., Zucchetta, M., & Link, J. (2014). Biomass accumulation across trophic levels: Analysis of landings for the Mediterranean Sea. Marine Ecology Progress Series, 512(October), 201–216. https://doi.org/10.3354/meps10881

Pranovi, F., Raicevich, S., Franceschini, G., Farrace, M. G., & Giovanardi, O. (2000). Rapido trawling in the northern Adriatic Sea: effects on benthic communities in an experimental area. ICES Journal of Marine Science, 57(3), 517-524.

Pranovi, F., Raicevich, S., Franceschini, G., Torricelli, P., & Giovanardi, O. (2001). Discard analysis and damage to non-target species in the" rapido" trawl fishery. Marine Biology, 139(5), 863-875.

QGIS Development Team (2014). QGIS Geographic Information System. Open Source Geospatial Foundation Project. https://qgis.org

Queiroz, C., Meacham, M., Richter, K., Norström, A. V., Andersson, E., Norberg, J., & Peterson, G. (2015). Mapping bundles of ecosystem services reveals distinct types of multifunctionality within a Swedish landscape. Ambio, 44(1), 89-101.

Quijas, S., Schmid, B., & Balvanera, P. (2010). Plant diversity enhances provision of ecosystem services: A new synthesis. Basic and Applied Ecology, 11(7), 582–593. https://doi.org/10.1016/J.BAAE.2010.06.009

R Development Core Team R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN: 3-900051-07-0. Available: http://www.R-project.org (2012).

Reiss, J., Bridle, J. R., Montoya, J. M., & Woodward, G. (2009). Emerging horizons in biodiversity and ecosystem functioning research. Trends in Ecology and Evolution, 24(9), 505–514. https://doi.org/10.1016/j.tree.2009.03.018

Resplandy, L., Keeling, R. F., Eddebbar, Y., Brooks, M. K., Wang, R., Bopp, L., ... & Oschlies, A. (2018). Quantification of ocean heat uptake from changes in atmospheric O 2 and CO 2 composition. Nature, 563(7729), 105-108.

Resplandy, L., Keeling, R. F., Eddebbar, Y., Brooks, M., Wang, R., Bopp, L., Long, M. C., Dunne, J. P., Koeve, W., & Oschlies, A. (2019). Quantification of ocean heat uptake from changes in atmospheric O2 and CO2 composition. Scientific Reports, 9(1), 1–10. https://doi.org/10.1038/s41598-019-56490-z

Rex, M. A., McClain, C. R., Johnson, N. A., Etter, R. J., Allen, J. A., Bouchet, P., & Warén, A. (2005). A source- sink hypothesis for abyssal biodiversity. The American Naturalist, 165, 163– 178.

Rilov, G., Mazaris, A. D., Stelzenmüller, V., Helmuth, B., Wahl, M., Guy-Haim, T., Mieszkowska, N., Ledoux, J. B., & Katsanevakis, S. (2019). Adaptive marine conservation planning in the face of climate change: What can we learn from physiological, ecological and genetic studies? Global Ecology and Conservation, 17, e00566. https://doi.org/10.1016/j.gecco.2019.e00566

Rinaldi, A., & Giovanardi, F. (2011). Contribution of Richard A. Vollenweider toward understanding eutrophication of the coastal Adriatic Sea. Aquatic ecosystem health & management, 14(2), 200-203.

Rodríguez Jr, J. P., TDB, B., EM, C., GS, C., & SJ, A. J.,… Peterson, GD (2006). Trade-offs across space, time, and ecosystem services. Ecology and Society, 11(1).

Rodríguez-Loinaz, G., Alday, J. G., & Onaindia, M. (2015). Multiple ecosystem services landscape index: A tool for multifunctional landscapes conservation. Journal of environmental management, 147, 152-163.

Rogers, H. M., Glew, L., Honzák, M., & Hudson, M. D. (2010). Prioritizing key biodiversity areas in Madagascar by including data on human pressure and ecosystem services. Landscape and Urban Planning, 96(1), 48-56.

Rogers, S. I., Casini, M., Cur, P., Heat, M., Irigoe, X., Kuos, H., Scheida, M., (2010). Marine strategy framework directive task group 4 report food webs, Publications Office of the European Union, Luxembourg. 63pp.

Romano, B., & Zullo, F. (2014). The urban transformation of Italy's Adriatic coastal strip: Fifty years of unsustainability. Land use policy, 38, 26-36.

Rombouts, I., Beaugrand, G., Fizzala, X., Gaill, F., Greenstreet, S. P. R., Lamare, S., le Loc'H, F., McQuatters-Gollop, A., Mialet, B., Niquil, N., Percelay, J., Renaud, F., Rossberg, A. G., & Féral, J. P. (2013). Food web indicators under the Marine Strategy Framework Directive: From complexity to simplicity? Ecological Indicators, 29, 246–254. https://doi.org/10.1016/j.ecolind.2012.12.021

Rosenberg, A. A., & McLeod, K. L. (2005). Implementing ecosystem-based approaches to management for the conservation of ecosystem services. Marine Ecology Progress Series, 300, 270-274.

Rossi, T., Connell, S. D., & Nagelkerken, I. (2017). The sounds of silence: regime shifts impoverish marine soundscapes. Landscape Ecology, 32(2), 239–248. https://doi.org/10.1007/s10980-016-0439-x

Rova, S., & Pranovi, F. (2017). Analysis and management of multiple ecosystem services within a social-ecological context. Ecological Indicators, 72, 436-443.

Rova, S., Müller, F., Meire, P., & Pranovi, F. (2019). Sustainability perspectives and spatial patterns of multiple ecosystem services in the Venice lagoon: Possible roles in the implementation of the EU Water Framework Directive. Ecological Indicators, 98, 556-567.

Ruckelshaus, M., McKenzie, E., Tallis, H., Guerry, A., Daily, G., Kareiva, P., ... & Bernhardt, J. (2015). Notes from the field: lessons learned from using ecosystem service approaches to inform real-world decisions. Ecological Economics, 115, 11-21.

Rugani, B., de Souza, D. M., Weidema, B. P., Bare, J., Bakshi, B., Grann, B., ... & Verones, F. (2019). Towards integrating the ecosystem services cascade framework within the Life Cycle Assessment (LCA) cause-effect methodology. Science of the Total Environment, 690, 1284-1298.

Russo, E., Monti, M. A., Mangano, M. C., Raffaeta, A., Sara, G., Silvestri, C., & Pranovi, F. (2020). Temporal and spatial patterns of trawl fishing activities in the Adriatic Sea (Central Mediterranean Sea, GSA17). Ocean & coastal management, 192, 105231.Porporato et al., 2020

Sabatés, A. N. A., Martìn, P., Lloret, J., & Raya, V. (2006). Sea warming and fish distribution: the case of the small pelagic fish, Sardinella aurita, in the western Mediterranean. Global change biology, 12(11), 2209-2219.

Sala, E., Ballesteros, E., Dendrinos, P., Di Franco, A., Ferretti, F., Foley, D., ... & Zabala, M. (2012). The structure of Mediterranean rocky reef ecosystems across environmental and human gradients, and conservation implications. PloS one, 7(2), e32742.

Sampaio, E., & Rosa, R. (2020). Climate change, multiple stressors, and responses of marine biota. Climate Action, 264-275.

Sampaio, E., Santos, C., Rosa, I. C., Ferreira, V., Pörtner, H. O., Duarte, C. M., ... & Rosa, R. (2021). Impacts of hypoxic events surpass those of future ocean warming and acidification. Nature Ecology & Evolution, 5(3), 311-321.

Scardi, M., Martin, C.S., Valavanis, V., Fraschetti, S., Belluscio, A., Gristina, M., Salomidi, M., Punzo, E., Panayotidis, P., Giannoulaki, M., 2013. Modeling of protected habitats using predictor variables. Mediterranean Sensitive Habitats (MEDISEH) Final Report. DG MARE Specific Contract SI2.600741.

Schickele, A., Guidetti, P., Giakoumi, S., Zenetos, A., Francour, P., & Raybaud, V. (2021). Improving predictions of invasive fish ranges combining functional and ecological traits with environmental suitability under climate change scenarios. Global Change Biology, 27(23), 6086-6102.

Schneiders, A., van Daele, T., van Landuyt, W., & van Reeth, W. (2012). Biodiversity and ecosystem services: Complementary approaches for ecosystem management? Ecological Indicators, 21, 123–133. https://doi.org/10.1016/J.ECOLIND.2011.06.021

Schramski, J. R., Dell, A. I., Grady, J. M., Sibly, R. M., & Brown, J. H. (2015). Metabolic theory predicts whole-ecosystem properties. Proceedings of the National Academy of Sciences of the United States of America, 112(8), 2617–2622. https://doi.org/10.1073/PNAS.1423502112

Schröter, M., Barton, D. N., Remme, R. P., & Hein, L. (2014). Accounting for capacity and flow of ecosystem services: A conceptual model and a case study for Telemark, Norway. Ecological indicators, 36, 539-551.

Schröter, M., Remme, R. P., & Hein, L. G. (2012). How and where to map supply and demand of ecosystem services for policy-relevant outcomes? Letter to the Editor. Ecological Indicators, 23, 220-221.

Schröter, M., Van der Zanden, E. H., van Oudenhoven, A. P., Remme, R. P., Serna-Chavez, H. M., De Groot, R. S., & Opdam, P. (2014). Ecosystem services as a contested concept: a synthesis of critique and counter-arguments. Conservation Letters, 7(6), 514-523.

Schulp, C. J. E., Alkemade, R., Klein Goldewijk, K., & Petz, K. (2012). Mapping ecosystem functions and services in Eastern Europe using global-scale data sets. International Journal of Biodiversity Science, Ecosystem Services and Management, 8(1–2), 156–168. https://doi.org/10.1080/21513732.2011.645880

Schwartz, M. W., Brigham, C. A., Hoeksema, J. D., Lyons, K. G., Mills, M. H., & Van Mantgem, P. J. (2000). Linking biodiversity to ecosystem function: implications for conservation ecology. Oecologia, 122(3), 297-305.

Seppelt, R., Dormann, C. F., Eppink, F. V., Lautenbach, S., & Schmidt, S. (2011). A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. Journal of applied Ecology, 48(3), 630-636.

Shephard, S., Brophy, D., & Reid, D. G. (2010). Can bottom trawling indirectly diminish carrying capacity in a marine ecosystem?. Marine Biology, 157(11), 2375-2381.

Shimamura, T., & Mizunoya, T. (2020). Sustainability prediction model for capital city relocation in Indonesia based on inclusive wealth and system dynamics. Sustainability, 12(10), 4336.

Smaal, A. C., & Vonck, A. P. M. A. (1997). Seasonal variation in C, N and P budgets and tissue composition of the mussel Mytilus edulis. Marine Ecology Progress Series, 153, 167-179. Tool4MSP - http://data.tools4msp.eu/tools4msp/

Snelgrove, P. V., Thrush, S. F., Wall, D. H., & Norkko, A. (2014). Real world biodiversity–ecosystem functioning: a seafloor perspective. Trends in ecology & evolution, 29(7), 398-405.

Solan, M., R. J. Aspden, and D. M. Paterson (2012). Marine Biodiversity and Ecosystem Functioning: Frameworks, Methodologies, and Integration. OUP Oxford. Google-Books-ID: xsDxllOQQ5MC.

Srivastava, D. S., Cardinale, B. J., Downing, A. L., Duffy, J. E., Jouseau, C., Sankaran, M., & Wright, J. P. (2009). Diversity has stronger top-down than bottom-up effects on decomposition. Ecology, 90(4), 1073-1083.

SROCC, IPCC, 2019: IPCC Special Report on the Ocean and Cryosphere in a Changing Climate [H.-O. Pörtner, D.C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, A. Alegría, M. Nicolai, A. Okem, J. Petzold, B. Rama, N.M. Weyer (eds.)]. Cambridge University Press, Cambridge, UK and New York, NY, USA, 755 pp. https://doi.org/10.1017/9781009157964.

Stachowicz, J. J., Bruno, J. F., & Duffy, J. E. (2007). Understanding the effects of marine biodiversity on communities and ecosystems. Annual Review of Ecology, Evolution, and Systematics, 739-766.

STECF (Scientific, Technical and Economic Committee for Fisheries) database - https://jeodpp.jrc.ec.europa.eu/ftp/jrc-opendata/FAD/fdi/ (2014-2018)

Stelzenmüller, V., Coll, M., Mazaris, A. D., Giakoumi, S., Katsanevakis, S., Portman, M., Renate, D., Mackelworth, P., Gimpel, G., Albano, P. G., Almpanidou, V., Claudet, J., Essl, F., Evagelopoulos, T., Heylmans, J. J., Genov, T., Kark, S., Micheli, F., Pennino, M. G., … Ojaveer, H. (2018). A risk- based approach to cumulative effect assessments for marine management. Science of the Total Environment, 612, 1132– 1140.

Steneck, R. S., & Dethier, M. N. (1994). A functional group approach to the structure of algal-dominated communities. Oikos, 476-498.

Stobart, B., Warwick, R., González, C., Mallol, S., Díaz, D., Reñones, O., & Goñi, R. (2009). Long-term and spillover effects of a marine protected area on an exploited fish community. Marine Ecology Progress Series, 384, 47–60. https://doi.org/10.3354/meps08007

Strong, J. A., Andonegi, E., Bizsel, K. C., Danovaro, R., Elliott, M., Franco, A., ... & Solaun, O. (2015). Marine biodiversity and ecosystem function relationships: the potential for practical monitoring applications. Estuarine, Coastal and Shelf Science, 161, 46-64.

Swift, M. J., Izac, A. M., & Van Noordwijk, M. (2004). Biodiversity and ecosystem services in agricultural landscapes—are we asking the right questions?. Agriculture, ecosystems & environment, 104(1), 113-134.

Tallis, H., & Polasky, S. (2011). Assessing multiple ecosystem services: an integrated tool for the real world. Natural capital: theory and practice of mapping ecosystem services, 34-50.

Tallis, H., Mooney, H., Andelman, S., Balvanera, P., Cramer, W., Karp, D., ... & Walz, A. (2012). A global system for monitoring ecosystem service change. Bioscience, 62(11), 977-986.

Tam, J. C., Link, J. S., Large, S. I., Andrews, K., Friedland, K. D., Gove, J., Hazen, E., Holsman, K., Karnauskas, M., Samhouri, J. F., Shuford, R., Tomilieri, N., & Zador, S. (2017). Comparing apples to oranges: Common trends and thresholds in anthropogenic and environmental pressures across multiple marine ecosystems. Frontiers in Marine Science, 4(SEP). https://doi.org/10.3389/fmars.2017.00282

Tam, J. C., Link, J. S., Rossberg, A. G., Rogers, S. I., Levin, P. S., Rochet, M. J., Bundy, A., Belgrano, A., Libralato, S., Tomczak, M., van de Wolfshaar, K., Pranovi, F., Gorokhova, E., Large, S. I., Niquil, N., Greenstreet, S. P. R., Druon, J. N., Lesutiene, J., Johansen, M., … Rindorf, A. (2017). Towards ecosystem-based management: Identifying operational food-web indicators for marine ecosystems. ICES Journal of Marine Science, 74(7), 2040–2052. https://doi.org/10.1093/icesjms/fsw230

TEEB, R. O. (2010). Mainstreaming the Economics of Nature.

Thuiller, W., Lafourcade, B., Engler, R., & Araújo, M. B. (2009). BIOMOD – a platform for ensemble forecasting of species distributions. Ecography, 32(3), 369–373. https://doi.org/10.1111/J.1600-0587.2008.05742.X

Tittensor, D. P., Novaglio, C., Harrison, C. S., Heneghan, R. F., Barrier, N., Bianchi, D., Bopp, L., Bryndum-Buchholz, A., Britten, G. L., Büchner, M., Cheung, W. W. L., Christensen, V., Coll, M., Dunne, J. P., Eddy, T. D., Everett, J. D., Fernandes-Salvador, J. A., Fulton, E. A., Galbraith, E. D., … Blanchard, J. L. (2021). Next-generation ensemble projections reveal higher climate risks for marine ecosystems. Nature Climate Change 2021 11:11, 11(11), 973–981. https://doi.org/10.1038/s41558-021-01173-9

Troy, A., & Wilson, M. A. (2006). Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. Ecological economics, 60(2), 435-449.

Turkelboom, F., Leone, M., Jacobs, S., Kelemen, E., García-Llorente, M., Baró, F., ... & Rusch, V. (2018). When we cannot have it all: Ecosystem services trade-offs in the context of spatial planning. Ecosystem services, 29, 566-578.

Turkelboom, F., Raquez, P., Dufrêne, M., Raes, L., Simoens, I., Jacobs, S., ... & Keune, H. (2013). CICES going local: Ecosystem services classification adapted for a highly populated country. In Ecosystem Services (pp. 223-247). Elsevier.

Tuttobarche.it website - https://www.tuttobarche.it/ricerca-rade-e-porti

Tzedakis, P. C., Lawson, I. T., Frogley, M. R., Hewitt, G. M., & Preece, R. C. (2002). Buffered tree population changes in a Quaternary refugium: Evolutionary implications. Science, 297(5589), 2044–2047. https://doi.org/10.1126/science.1073083

UNEP-WCMC - Brown, C., Reyers, B., Ingwall-King, L., Mapendembe, A., Nel, J., O'Farrell, P., Dixon, M. & Bowles-Newark, N. J. (2014). Measuring ecosystem services: Guidance on developing ecosystem service indicators, Cambridge, UK.

Van de Waal, D. B., & Litchman, E. (2020). Multiple global change stressor effects on phytoplankton nutrient acquisition in a future ocean. Philosophical Transactions of the Royal Society B, 375(1798), 20190706.

Van Oudenhoven, A. P., Petz, K., Alkemade, R., Hein, L., & de Groot, R. S. (2012). Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecological indicators, 21, 110-122.

Van Oudenhoven, A. P., Schröter, M., Drakou, E. G., Geijzendorffer, I. R., Jacobs, S., van Bodegom, P. M., ... & Albert, C. (2018). Key criteria for developing ecosystem service indicators to inform decision making. Ecological Indicators, 95, 417-426.

Van Vallen L., A new evolutionary law. Evol. Theory 1, 1–30 (1973).

Vargas-Yáñez, M., García, M. J., Salat, J., García-Martínez, M. C., Pascual, J., & Moya, F. (2008). Warming trends and decadal variability in the Western Mediterranean shelf. Global and Planetary Change, 63(2-3), 177-184.

Vassallo, P., Paoli, C., Aliani, S., Cocito, S., Morri, C., & Bianchi, C. N. (2020). Benthic diversity patterns and predictors: A study case with inferences for conservation. Marine Pollution Bulletin, 150, 110748.

Veidemane, K., Ruskule, A., Strake, S., Purina, I., Aigars, J., Sprukta, S., Ustups, D., Putnis, I., & Klepers, A. (2017). Application of the marine ecosystem services approach in the development of the maritime spatial plan of Latvia. International Journal of Biodiversity Science, Ecosystem Services and Management, 13(1), 398–411. https://doi.org/10.1080/21513732.2017.1398185

Vergara, X., Carmona, A., & Nahuelhual, L. (2021). Spatial coupling and decoupling between ecosystem services provisioning and benefiting areas: Implications for marine spatial planning. Ocean & Coastal Management, 203, 105455.

Vergés, A., McCosker, E., Mayer-Pinto, M., Coleman, M. A., Wernberg, T., Ainsworth, T., & Steinberg, P. D. (2019). Tropicalisation of temperate reefs: Implications for ecosystem functions and management actions. Functional Ecology, 33(6), 1000–1013. https://doi.org/10.1111/1365-2435.13310

Viitasalo, M., & Bonsdorff, E. (2022). Global climate change and the Baltic Sea ecosystem: direct and indirect effects on species, communities and ecosystem functioning. Earth System Dynamics, 13(2), 711-747.

Villamagna, A. M., Angermeier, P. L., & Bennett, E. M. (2013). Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. Ecological Complexity, 15, 114-121.

Villéger, S., Brosse, S., Mouchet, M., Mouillot, D., & Vanni, M. J. (2017). Functional ecology of fish: current approaches and future challenges. Aquatic Sciences, 79(4), 783-801.

Violle, C., Navas, M. L., Vile, D., Kazakou, E., Fortunel, C., Hummel, I., & Garnier, E. (2007). Let the concept of trait be functional!. Oikos, 116(5), 882-892.

Vollenweider, R. A., Giovanardi, F., Montanari, G., & Rinaldi, A. (1998). Characterization of the trophic conditions of marine coastal waters with special reference to the NW Adriatic Sea: proposal for a trophic scale, turbidity and generalized water quality index. Environmetrics: The official journal of the International Environmetrics Society, 9(3), 329-357.Fiori et al., (2016)

Wallace, K. J. (2007). Classification of ecosystem services: Problems and solutions. Biological Conservation, 139(3–4), 235–246. https://doi.org/10.1016/j.biocon.2007.07.015

Walther, G. R. (2010). Community and ecosystem responses to recent climate change. Philosophical Transactions of the Royal Society B: Biological Sciences, 365(1549), 2019-2024.

Walther, G. R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T. J., ... & Bairlein, F. (2002). Ecological responses to recent climate change. Nature, 416(6879), 389-395.

Whittaker, R. H. I. (2010). A Consideration of Climax Theory : The Climax as a Population and Pattern Author (s): R . H . Whittaker Published by : Ecological Society of America Stable URL : http://www.jstor.org/stable/1943519 Your use of the JSTOR archive indicates your acceptance. Ecological Monographs, 23(1), 41–78.

Winter, S., Bauer, T., Strauss, P., Kratschmer, S., Paredes, D., Popescu, D., Landa, B., Guzmán, G., Gómez, J. A., Guernion, M., Zaller, J. G., & Batáry, P. (2018). Effects of vegetation management intensity on biodiversity and ecosystem services in vineyards: A meta-analysis. Journal of Applied Ecology, 55(5), 2484–2495. https://doi.org/10.1111/1365-2664.13124

Wolff, C., Vafeidis, A. T., Muis, S., Lincke, D., Satta, A., Lionello, P., ... & Hinkel, J. (2018). A Mediterranean coastal database for assessing the impacts of sea-level rise and associated hazards. Scientific data, 5(1), 1-11.

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., Jackson, J. B. C., Lotze, H. K., Micheli, F., Palumbi, S. R., Sala, E., Selkoe, K. A., Stachowicz, J. J., & Watson, R. (2006). Impacts of biodiversity loss on ocean ecosystem services. Science, 314(5800), 787–790.

WTTC (2012). Travel and tourism—economic impact 2012, Croatia (http://www.wttc.org/site_media/uploads/downloads/croatia2012.pdf, 5 February 2021)

Wu, H. (1995). Studies on Denaturation of Proteins XIII. A Theory of Denaturation. Advances in Protein Chemistry, 46(C), 6–26. https://doi.org/10.1016/S0065-3233(08)60330-7

Yachi, S., & Loreau, M. (1999). Biodiversity and ecosystem productivity in a fluctuating environment: The insurance hypothesis. Proceedings of the National Academy of Sciences of the United States of America, 96(4), 1463–1468. https://doi.org/10.1073/pnas.96.4.1463

Yildirim, S., & Kaplan, M. (2022). Seafood Security and Sustainability Through Sustainable Development: A Review of Turkish Seafood Market. Research Anthology on Strategies for Achieving Agricultural Sustainability, 951-970.

Zanetto, G., & Soriani, S. (1996). Tourism and environmental degradation: the Northern Adriatic Sea. Sustainable tourism? European experiences., 137-152.

Zhang, Z., Liu, Y., Wang, Y., Liu, Y., Zhang, Y., & Zhang, Y. (2020). What factors affect the synergy and tradeoff between ecosystem services, and how, from a geospatial perspective? Journal of Cleaner Production, 257, 120454.