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Towards an ecosystem-based approach in marine ecosystem accounting

Seagrass ecosystems in the Mediterranean Sea: from diversity to restoration

Addamo, AM; La Notte, A

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Contents

Abstract

The aim of this report is to highlight the relevance of implementing an ecosystem-based approach in the marine ecosystem accounting. The evidences are based on non-exhaustive literature review and data and analyses present in non-public external study reports commissioned by Joint Research Centre (section 1, 2, 3), and are used to present a conceptual framework (section 4) for marine accounts. The report will focus on seagrass ecosystems, with a case study in the Mediterranean Sea, explaining their importance as essential (vulnerable) ecosystems with several key roles, from biodiversity hotspots to climate change mitigation, and highlighting their characteristics, condition, threats, and potential values of ecosystem services fundamental for the society and economy. Finally, the report will summarize the main methodologies applied for seagrass restoration (section 3) and include a brief narrative on the marine ecosystem accounting (section 4) as pivotal implementation of conservation, protection and restoration actions in the framework of European legislations, such as Biodiversity Strategy for 2030 and the Proposal on Nature Restoration Law, Marine Strategy Framework Directive, Common Fisheries Policy Regulation , Ecosystem-based Approach for Maritime Spatial Planning Directive, Nature-based Solutions, Sustainable Blue Economy, Taxonomy Regulation for Sustainable Activities, and the Regulation (EU) No 691/2011 on European environmental economic accounts.

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1 Introduction

Marine ecosystems provide numerous services and benefits to the world population. The European and global policy context have highlighted the relevant link between economy and environment in several technical and legislative tools such as Biodiversity Strategy for 2030 (BDS2030) and the Proposal on Nature Restoration Law (NRL), Marine Strategy Framework Directive (MSFD), Common Fisheries Policy Regulation (CFP), Ecosystem-based Approach for Maritime Spatial Planning Directive (MSP), Nature-based Solutions (NBS), Sustainable Blue Economy (SBE), Taxonomy Regulation for Sustainable Activities, and the Regulation (EU) No 691/2011 on European environmental economic accounts. In particular, the United Nations Sustainable Development Goals (SDGs) especially SDG14 - Life below water, the Kunming-Montreal Global Biodiversity Framework (GBF) (CBD, 2022a) adopted under the United Nations Convention on Biological Diversity that includes genetic and phylogenetic diversity as indicators to prevent diversity loss and improve its contributions to species, ecosystems, and society among the four goals and 23 targets to be achieved by 2030. Recently, United Nations System of Environmental-Economic Accounting Ecosystem Accounting (SEEA EA) have been adopted as standard to guide and measure the contribution of the environment to the economy targeting specific accounts that reflects the role of ecosystems and their services in a consistent and comprehensive way (UN, 2021).

Promote conservation of marine ecosystems and sustainable use of their resources in an equitable way implies an in-depth scientific knowledge of the ecological functioning and environmental status of the marine ecosystems. Moreover, it involves an ecosystem-based approach that applies appropriate scientific methodologies focused on levels of biological organization that encompass the essential processes, functions and interactions among organisms and their environment (CBD, 2004). Unfortunately, after more than 50 years, is still very current the challenge defined by ecologist Odum in 1971: "*obtaining data on biomass representative of the large systems of nature such as forests and seas is a very difficult task that has occupied ecological research for 50 years with thousands of methods and varying results. When small spots are studied or small samples taken, the data are not representative because of the large statistical variation that is characteristic of most ecosystems. Efforts to sample large sections of systems are laborious and expensive*"). The importance of ecosystem-based studies is more than just large size, and does not simplify an ecosystem to derive cause and effect more easily. A ecosystem-based approach attempts to include many more pathways and feedbacks in the system than do simpler systems becoming an ecological study of such a spatial and temporal scale as to include most if not all processes of the ecosystem (Odum 1971, Mitsch & Day Jr. 2004).

The ultimate aim of the report is to provide a glimpse of the relevance of comprehensive ecosystem-based approach for an efficient marine preservation and benefit-sharing through sustainable use of marine resources.

2 Seagrass ecosystems

Seagrasses are marine flowering plants able to create extensive meadows that provide habitat for multiple life stages of many commercially- and recreationally-important fishes, shellfish and other marine organisms, and are source of multiple ecosystem services, including nursery habitat, improved water quality, coastal protection, carbon sequestration, stabilize sediment, and reduce coastal erosion (Valdez et al. 2020 and reference therein). More than 70 seagrass species around the world are found in 159 countries on six continents, potentially covering over 300,000 km^2, with more than 1 billion people living within 100 km of a seagrass meadow (Short et al. 2007). Unfortunately, seagrass habitats and associated ecosystem services provision are in decline globally: estimates range of 30–60% of the total area covered by seagrass species, and include total loss in some places (Evans et al. 2018). Causes of degradation and loss of seagrass involve anthropogenic influences (e.g. direct removal during coastal development, destructive fishing methods, run-off of nutrients and other pollutants from land-based sources, and climate change) and causal mechanisms that increase frequency and intensity of stressors (e.g. light reduction, extreme weather events, high nutrient concentrations, and poor sediment conditions) (Valdez et al. 2020 and reference therein).

2.1 Distribution & diversity of seagrass ecosystems

Seagrasses are widely distributed along temperate and tropical coastlines of the world and classified into six global bioregions based on species assemblages, species distributional ranges, and tropical and temperate influences (Short et al. 2007, Figure 1). The diversity of marine flowering plants of the Order Alismatales, currently taxonomically verified, includes 6 families, 15 genera and 77 species (WoRMS, 2023) (Table 1).

Figure 1. Global map of seagrass distribution, species richness and bioregions. *Source*: adapted from GRID-Arendal, https://www.grida.no/resources/13590. Creator credit: Hisham Ashkar; Short et al. 2007; UNEP WCMC, Short 2018. Map produced by Levi Westerveld/GRID-Arendal (2019). Projection: Goode Homolosine.

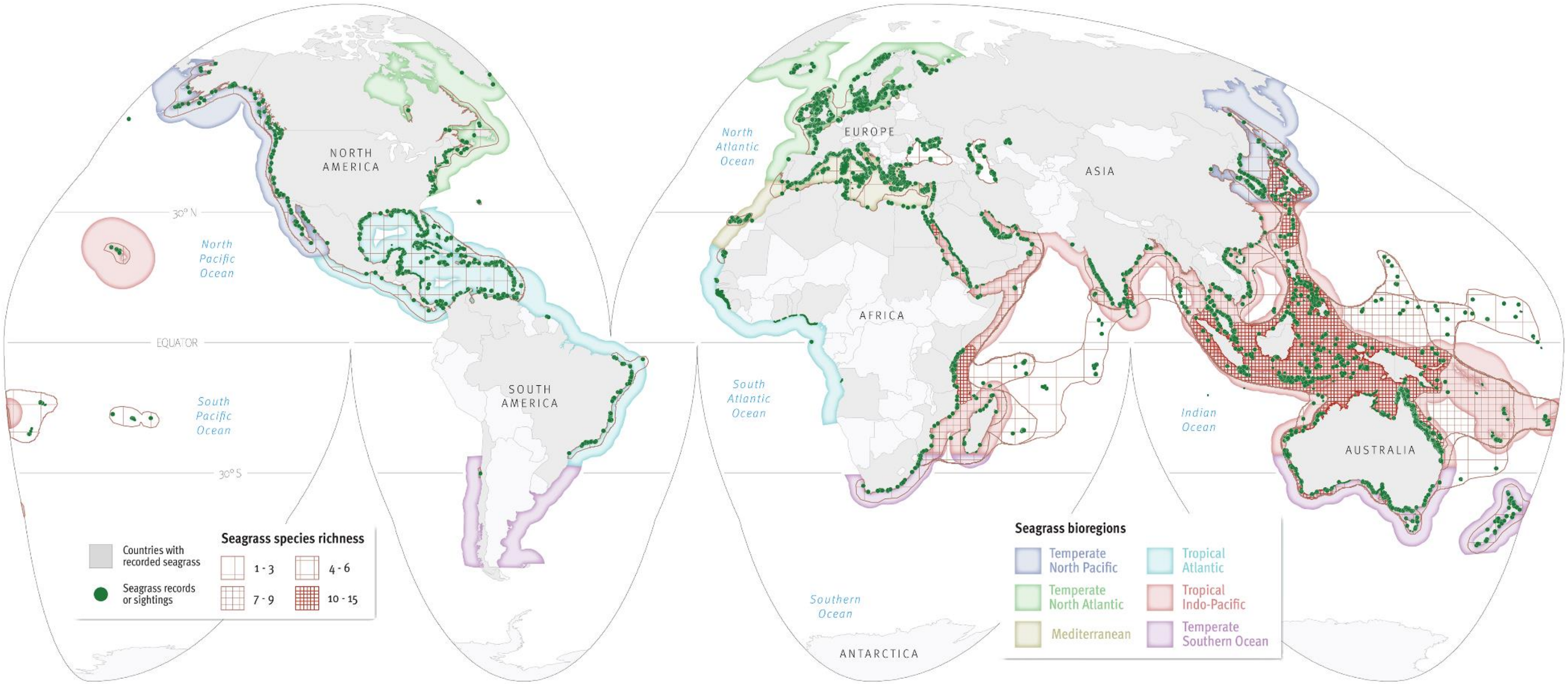


Table 1. Classification of seagrass. *Source:* WoRMs (2023)

Family	Genera[1]	Species[1,2]
Cymodoceaceae	*Amphibolis* C.Agardh *Cymodocea* K.D.Koenig *Halodule* Endl. *Oceana* Byng & Christenh. *Syringodium* Kütz. *Thalassodendron* Hartog	*Amphibolis antarctica* (Labillardière) Sonder & Ascherson ex Ascherson, 1868 *Amphibolis griffithii* (J.M.Black) Hartog, 1970 *Cymodocea angustata* Ostenf. ***Cymodocea nodosa*** (Ucria) Asch. *Cymodocea rotundata* Asch. & Schweinf. *Halodule bermudensis* Hartog *Halodule ciliata* (Hartog) Hartog *Halodule emarginata* Hartog *Halodule pinifolia* (Miki) Hartog *Halodule uninervis* (Forssk.) Asch. *Halodule wrightii* Asch. *Oceana serrulata* (R.Brown) Byng & Christenh. *Syringodium filiforme* Kütz. *Syringodium isoetifolium* (Asch.) Dandy *Thalassodendron ciliatum* (Forssk.) Hartog *Thalassodendron leptocaule* Maria C.Duarte, Bandeira & Romeiras, 2012 *Thalassodendron pachyrizum* Hartog
Hydrocharitaceae	*Enhalus* L.C.Richard, 1811 *Halophila* Du Petit-Thouars, 1806 *Thalassia* Banks ex König 1805	*Enhalus acoroides* (Linnaeus f.) Royle, 1839 *Halophila australis* Doty & B.C.Stone, 1966 *Halophila baillonis* Ascherson, 1874 *Halophila balfourii* Solereder, 1913 *Halophila beccarii* Ascherson, 1871 *Halophila decipiens* Ostenfeld, 1902 *Halophila engelmannii* Ascherson, 1875 *Halophila gaudichaudii* J.Kuo, 2006 *Halophila hawaiiana* Doty & B.C.Stone, 1966 *Halophila johnsonii* Eiseman, 1980 *Halophila major* (Zollinger) Miquel, 1856 *Halophila mikii* J.Kuo, 2006 *Halophila minor* (Zollinger) Hartog, 1957 *Halophila nipponica* J.Kuo, 2006 *Halophila okinawensis* J.Kuo, 2006 *Halophila ovalis* (R.Brown) Hooker f., 1858 *Halophila spinulosa* (R.Brown) Ascherson, 1875 ***Halophila stipulacea*** (Forsskål) Ascherson, 1867 *Halophila sulawesii* J.Kuo, 2007 *Halophila tricostata* M.Greenway, 1979 *Thalassia hemprichii* (Ehrenberg) Ascherson, 1871 *Thalassia testudinum* K.D.Koenig, 1805
Posidoniaceae	*Posidonia* König, 1805	*Posidonia angustifolia* Cambridge & Kuo, 1979 *Posidonia australis* J.D.Hooker, 1858 *Posidonia denhartogii* Kuo & Cambridge, 1984 *Posidonia kirkmanii* J.Kuo & Cambridge, 1984 ***Posidonia oceanica*** (Linnaeus) Delile, 1813 *Posidonia ostenfeldii* Hartog, 1970 *Posidonia sinuosa* Cambridge & Kuo, 1979
Potamogetonaceae	*Althenia* Petit, 1829 *Lepilaena* J.Drummond ex Harvey, 1855	*Althenia australis* (J.Drummond ex Harvey) Ascherson, 1887 *Althenia cylindrocarpa* (Koernicke) Ascherson, 1899 *Althenia filiformis* Petit, 1829 *Althenia marina* (E.L.Robertson) Yu Ito, 2016 *Althenia preissii* (Lehmann) Ascherson & Graebner, 1907 *Lepilaena cylindrocarpa* (Koernicke) Bentham, 1878

Family	Genera[1]	Species[1,2]
		Lepilaena marina E.L.Robertson, 1984 *Lepilaena preissii* (Lehmann) F.Mueller, 1874
Ruppiaceae	*Ruppia* L.	***Ruppia cirrhosa*** (Petagna) Grande, 1918 ***Ruppia maritima*** Linnaeus, 1753 *Ruppia megacarpa* Mason, 1967 *Ruppia tuberosa* Davis & Tomlinson, 1974
Zosteraceae	*Phyllospadix* Hook *Zostera* L	*Phyllospadix iwatensis* Makino, 1931 *Phyllospadix japonicus* Makino *Phyllospadix juzepczukii* Tzvelev *Phyllospadix scouleri* Hook. *Phyllospadix serrulatus* Rupr.t ex Asch. *Phyllospadix torreyi* S.Watson *Zostera asiatica* Miki, 1932 *Zostera caespitosa* Miki, 1932 *Zostera caulescens* Miki, 1932 *Zostera chilensis* (J.Kuo) S.W.L.Jacobs & D.H.Les, 2009 *Zostera japonica* Ascherson & Graebner, 1907 *Zostera muelleri* Irmisch ex Ascherson, 1867 *Zostera nigricaulis* (J.Kuo) S.W.L.Jacobs & D.H.Les, 2009 ***Zostera noltei*** Hornemann, 1832 *Zostera polychlamys* (J.Kuo) S.W.L.Jacobs & D.H.Les, 2009 ***Zostera Zostera marina*** Linnaeus, 1753 *Zostera Zosterella capensis* Setchell *Zostera tasmanica* G.Martens ex Ascherson, 1868

[1] Only checked names i.e. verified by a taxonomic editor; [2] in **bold** species present in the Mediterranean Sea.

The Kunming-Montreal Global Biodiversity Framework of the Convention on Biological Diversity (CBD 2022a) include goal to enhance biodiversity and ecosystem functions and services, ecological integrity and connectivity, and to strengthen the genetic and phylogenetic diversity, indicators, and targets for wild species in natural habitats by specifying "tolerable" losses of genetic (GD) and phylogenetic diversity (PD) (i.e. variation at the DNA level within species, which facilitates species adaptation and ecosystem function), (see CBD 2022a, Frankhman 2022, Hoban et al. 2020, Robuchon et al. 2023). In particular, the new adopted indicators include: 1) proportion of populations within species with effective population size above 500;2) proportion of populations maintained within species; 3) number of plant and animal genetic resources for food and agriculture secured in either medium- or long-term conservation facilities; 4) proportion of local breeds classified as being at risk of extinction; 5) comprehensiveness of conservation of socioeconomically as well as culturally valuable species; 6) genetic scorecard for wild species; 7) expected loss of phylogenetic diversity, 8) changing status of evolutionary distinct and globally endangered species (EDGE Index); and 8) ecological connectivity indicators (e.g., Convention on Migratory Species connectivity indicator; maintenance and restoration of connectivity of natural ecosystems; connectivity Indicator (in development)) (CBD 2022b, Robuchon et al. 2023). Genetic and phylogenetic diversity, and connectivity are the foundation of species and ecosystem diversity, though they have often been neglected. For this reason, their inclusion of indicators of CBD goals highlight the increasing recognition of the role of genetic and phylogenetic biodiversity, and connectivity for contributing to species' survival, ecosystem resilience and productivity, and nature's contribution to people (CBD 2022b, Hoban et al. 2020, Robuchon et al. 2023).

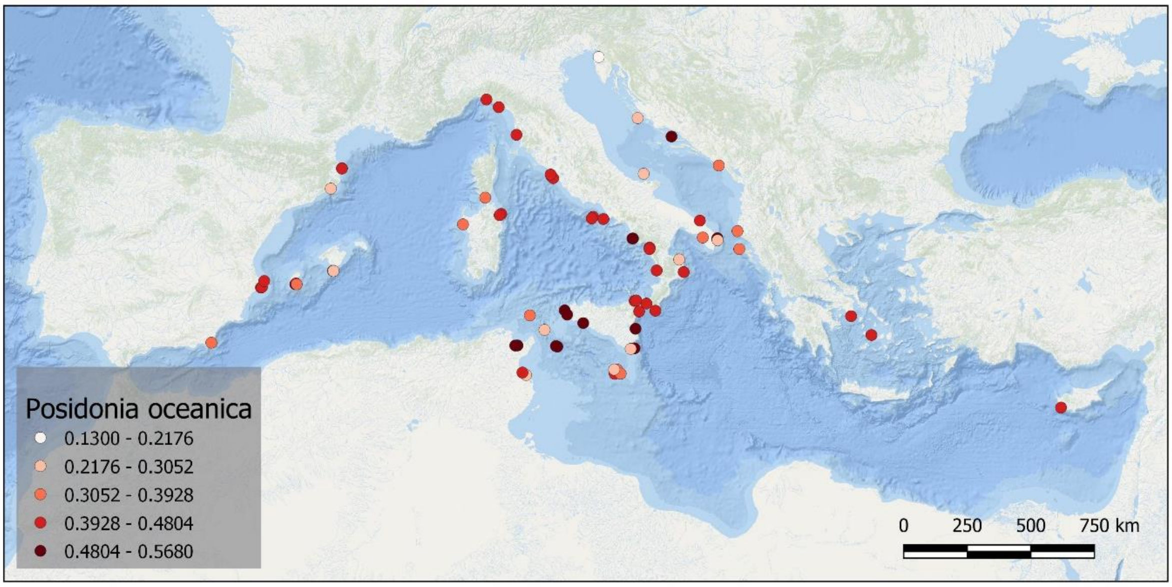
2.1.1 Mediterranean case study and knowledge gaps

The diversity of seagrass species that occur in the Mediterranean Sea includes 6 major seagrass families Cymodoceaceae, Hydrocharitaceae, Posidoniaceae, Ruppiaceae, and Zosteraceae, and 7 species *Cymodocea nodosa*, *Halophila stipulacea*, *Posidonia oceanica*, *Ruppia cirrhosa*, *Ruppia maritima*, *Zostera noltei*, and *Zostera Zostera marina* respectively (Table 1).

Although there are numerous studies on diversity, distribution and ecology of the seagrass species (see e.g., Jordà et al. 2012, Effrosynidis et al. 2019), most of them mainly focus on the endemic Mediterranean species

P. oceanica. Despite being outnumbered by the predecessors, a similar trend is also seen in molecular studies where *P. oceanica* takes the lead (Repullés et al. 2022) and other species are neglected. The genetic data currently available for the endemic seagrass species *Posidonia oceanica* in the Mediterranean Sea demonstrated a biogeographical variation of its diversity (Repullés et al. 2022, Figure 2). The maximum values were found in the Central Mediterranean, around Sicily, north-eastern coasts of Tunisia, and in the Adriatic Sea. Particular attention to the values obtained for south-eastern Sicily, where very close localities present highly disparate genetic diversity values (Repullés et al. 2022).

Figure 2. Genetic diversity (bottom-left) of *Posidonia oceanica* in the Mediterranean Sea. Darker colours represent higher genetic diversity values. Note: the maximum values were found in the Central Mediterranean, around Sicily, north-eastern coasts of Tunisia, and in the Adriatic Sea. Repullés et al. (2022) draw attention to the disagreement in certain localities where several studies were done (see, for instance, values obtained for south-eastern Sicily), where very close localities present highly disparate genetic diversity values. *Source:* Repullés et al. 2022.



Frankhman (2022) evaluated the proposed genetic goals and targets for CBD under six scenarios (3, 5 or 10% loss of genetic diversity [heterozygosity] over 8 or 32 years) by predicting their consequences on genetic diversity, inbreeding, fitness, and evolutionary potential when applied at the same rate for 100 years. Results demonstrated substantial genetic harm to species when continued for 100 years that will compromise species persistence, especially in the context of environmental change. Therefore, Frankhman (2022) proposed alternative indicators that would reflect improvements in the genetic status of populations and species, namely (1) the number of species and their populations being maintained at sizes sufficient to retain evolutionary potential in perpetuity, and (2) the number of species for which population genetic connectivity has been improved.

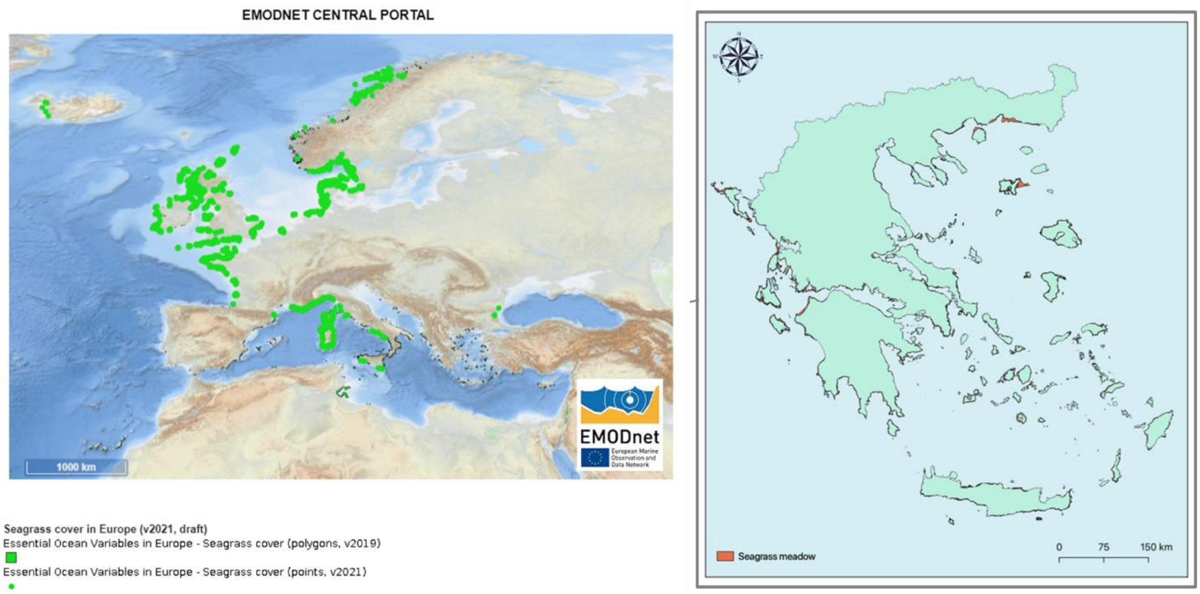
Gene flow governs the contemporary spatial structure and dynamic of populations as well as their long-term evolution, and investigating such aspects in marine ecosystems would provide crucial information about how they could react to future perturbations and open the way to novel approaches to spatial ecology (Legrand et al. 2022). Moreover, in the context of marine biodiversity loss and spatial conservation planning, it is pivotal to understand the eco-evolutionary dynamics and forces that continuously shape population structures and anticipate climate-driven redistributions, to improve protection strategies and management of marine resources (see Legrand et al. 2022).

Focusing on detailed seagrass distribution within each bioregion, the lack of information is still among the knowledge gaps that need to but can be filled by a combination of in-situ and earth observations (see e.g. Hossain & Hashim 2019, Mumby et al. 1997, Traganos et al. 2022). European examples of are (a) the latest version of the current known extent and distribution of seagrass meadows in European waters provided by EMODnet Seabed Habitats (Figure 3, left); and (b) the details of seagrass meadows in the Greek seas (Figure 3, right).

Although the EMODnet data product should be considered a work in progress and not an official product, the layer shows the best compilation of current evidence for the essential ocean variable (EOV) “Seagrass cover and composition” (sub-variable of “Areal extent of seagrass meadows”) as defined by the Global Ocean Observing System (GOOS, https://www.goosocean.org/). The map includes information of seagrass species belonging to four major families: Zosteraceae, Hydrocharitaceae, Posidoniaceae and Cymodoceaceae (Lillis et al. 2021).

A combination of optical, and acoustic remote sensing techniques (satellite imagery and sidescan sonar, respectively), as well as in-situ methodologies (visual census; SCUBA diving, Towed Underwater Cameras, and Remotely Operated Vehicles) was employed to map the spatial distribution of seagrass habitats in the coastal waters of the Hellenic territory. The map includes information of *Posidonia oceanica*, covering the vast majority of seabed at depths between the shoreline and 25 – 30 m (or deeper in insular areas), followed by the species *Cymodocea nodosa*, *Zostera noltei*, and *Halophila stipulacea* (Panayotidis et al. 2022*ab*).

Figure 3. Left: Seagrass cover (Essential Ocean Variable) in Europe. *Source*: EMODnet Seabed Habitats https://www.emodnet-seabedhabitats.eu). Right: Seagrass Meadows in The Greek Seas recorded at approximately 70% of the Hellenic coastline (Eastern Ionian, Aegean and Levantine Seas), and their surface area exceeded 2,673.1 km2. *Source*: Panayotidis et al. 2022*ab*



Finally, habitat suitability modelling is an additional useful tool for foreseen and forestall effective management and conservation of marine ecosystems and resources (see e.g. Bertelli et al. 2022, Boström et al. 2014, Rowden et al. 2017, Staehr et al. 2019). Habitat suitability models (HSMs) and species distribution models (SDMs) can be used to predict the likelihood of species occurrence based on an understanding of the environmental variables that determine species distribution (Hirzel & Le Lay 2008, Elith et al. 2011), so the understanding of species-habitat associations can be developed even when biological datasets are limited (Bertelli et al. 2022).

Habitat suitability in terms of the seabed spatial extent available for the growth of seagrass meadows and the seawater clarity conditions are critical factors for the formation of well-structured and extensive meadows and their results are of great importance and usefulness for the effective management and conservation of valuable marine ecosystems (Panayotidis et al. 2022*ab*). In this context, developing a fast and cost-efficient HSM would facilitate the implementation of environmental management and conservation (see Bakirman & Gumusay 2020, Boström et al. 2014, Staehr et al. 2019).

The spatial habitat GIS modelling approach applied in López-Márquez (2022) for predicating seagrass coverage in the Mediterranean Sea follows a similar model approach developed by Staehr et al. (2019). The index maps developed for each environmental variable (e.g. oxygen, temperature, salinity, sediment, light, physical exposure and bathymetry) were combined to produce a seagrass index that shows the best suitable area for seagrass habitats as a result of the information of the seven variables analysed for four species (*Cymodocea nodosa*, *Posidonia oceanica*, *Zostera Z. marina*, and *Z. noltei*) inhabiting the Mediterranean Sea

(Figure 4). The habitat suitability model was calculated as a simple multiplicative model that gives equal importance to each environmental data layer analysed:

$$Seagrass_{index} = Oxygen_{index} * Temperature_{index} * Salinity_{index} * Sediment_{index} * Light_{index} * Physical\ exposure_{index} * Bathymetry_{index}$$

where each factor is calculated using a geoprocessing tool that perform a conditional if/else evaluation on each of the input cells of an input raster, as follow:

- *Oxygen*$_{index}$ shows the negative effect of anoxic conditions on the plant metabolism in term of growth and survival: oxygen concentration values lower than 2mg/L are considered unfavorable for seagrass presence

  Oxygen$_{index}$ *= CON("oxygen" > 2, 1, exp(1 * ("oxygen" - 2)))*

- *Temperature*$_{index}$ shows the tolerance limits of seagrass growth and distribution included: temperature values between 16 and 18°C are considered optimal range for seagrass growth

  Temperature$_{index}$ *= CON("temperature">18, -0.0323*("temperature")2 + 1.1086*"temperature"-8.551, IF("temperature"<16, -0.0323*("temperature")2 + 1.1086*"temperature"-8.551, CON("temperature" >16,1)))*

- *Salinity*$_{index}$ shows the negative effect of osmotic pressures on seagrass that might cause mortality of plant: salinity values over 39 psu are considered cause of seagrass mortality

  Salinity$_{index}$ *= CON("salinity" < 39, 1, exp(1*(39-"salinity")))*

- *Sediment*$_{index}$ show the effect of sea bottom on the ability for seagrass settlement: different sediment types were assigned with values ranging between 0 (unsuitable) and 1 (suitable)

  Sediment$_{index}$ *=*

  Seabed: 0.001
  Fine mud or Sandy mud or Muddy sand: 0.1
  Sand: 1
  Coarse-mixed sediment: 0.5
  Rock or Other hard substrata: 0.5
  Sandy mud: 0.1
  Muddy sand: 0.1
  Posidonia oceanica *meadows: 1*
  Fine mud: 0.1
  Mixed sediment: 0.1
  Facies of the euryhaline and/or Eurythermal lagoon biocenosis: 0.001
  Dead mattes of Posidonia oceanica*: 0.001*
  Coralligenous platforms: 0.5

- *Bathymetry* $_{index}$ shows the negative effect of high depths as limiting factor for seagrass distribution and settlement: depth values below and over 40m are considered suitable (1) and unsuitable (1) for seagrass presence

  Bathymetry$_{index}$ *= CON("Bathymetry" >40, 0, 1)*

- *Light*$_{index}$ shows the important factors in the regulation of seagrass maximum depth distribution : values of 25 and 140 μmol phot m^{-2} s^{-1} are considered minimum and optimal value for seagrass growth

  Light$_{index}$ *= CON("light" >= 140, 1, CON(("light"<140) & ("light">25), -0.00002*("light")**2 + 0.0094* "light"-0.0667, CON("light" <= 25, 0)))*

- *Physical exposure*$_{index}$ shows the negative effect of waves exposure for seagrass coverage: values exposure levels below the cut-off value of 0.2 N/m^2 are considered to enable coverage of seagrass

  Exposure$_{index}$ *= CON((“Exposure”>0.2, (20*EXP(-15*“exposure”)), 1)*

Figure 4. Seagrass index for the Mediterranean Sea. Areas in grey represent null probability to be a suitable zone for seagrass habitats, while the rest of colours represent the different probabilities to be a suitable zone for seagrass habitats. *Source*: López-Márquez 2022

As mentioned in Staehr et al. (2019), the result has values ranging from 0 to 1, reflecting 0-100% of the expected presence of the seagrass. The seagrass index was done on the 100x100m resolution layers (Figure 4). The values of seagrass index have been reclassified with binary values 0 (absence) and 1 (presence), and where grid values different from 0 are considered as a potential area of suitable habitats for the seagrasses whereas the rest of the areas do not potentially have suitable conditions for the establishment of the seagrass habitats (Figure 5).

Figure 5. Seagrass index for the Mediterranean Sea. Areas in dark green represent null probability (value 0) to be a suitable zone for seagrass habitats, while light green represent the probability (value 1) to be a suitable zone for seagrass habitats. *Source*: López-Márquez 2022

Value ranges of each variable were restricted to the best optimal conditions for the growth of the four seagrass species included in the study, and in some cases variables could have a more relaxed limit range for the good condition of seagrass growth. Nevertheless, the overall agreement at this large scale between the model output and the currently available data on distribution of seagrass meadows in European waters, collated by EMODnet Seabed Habitats, indicates that the model contains the most important controlling factors (e.g. temperature, salinity, light, oxygen, physical exposure, bathymetry, and sediment) that determine the seagrass distribution (López-Márquez 2022, Figure 6).

Figure 6. Model validation combining the seagrass model outputs and the information of the distribution of the seagrass species in red points (top) and polygons (bottom). Light green areas represent the potential zones for the seagrass habitats. *Source*: López-Márquez 2022

The model appears to be a useful management tool to identify the environmental conditions that are currently restricting the distribution of seagrass. It could identify the most suitable habitats for seagrasses and could help to make decisions on conservational strategies and management of this key ecosystem in the Mediterranean Sea.

Moreover, López-Márquez (2022) demonstrates that the model developed by Staehr et al (2019) for eelgrass species in Danish coastal waters could be applied on a higher scale as the Mediterranean Sea, and could represent a guideline for other marine habitats and marine regions. Nevertheless, the model has some flaws that need to be carefully considered during its application, for example:

- the model is obtained by the aggregation of different sources of information, and resulted in a raster file with grid size 100 x 100 m, thus the data pre-processing is extremely important and time-consuming;
- at the time of the study, some databases were not complete in terms of information accuracy, and grids with missing information were necessarily filled with mean values of each variables to avoid

problems in the data processing. Therefore, the information of each layers is not perfectly an accurate interpretation of the reality, and a loss of accuracy in the model could result in an underestimated probability and/or percentage of suitable areas for seagrass habitats;

- the lack of information in the scientific literature about the tolerance limits of some variables for seagrass survival hinders the correct definition of optimal o good ranges for each variable.

2.2 Ecosystem services capacity, cascade phenomena & functional connectivity

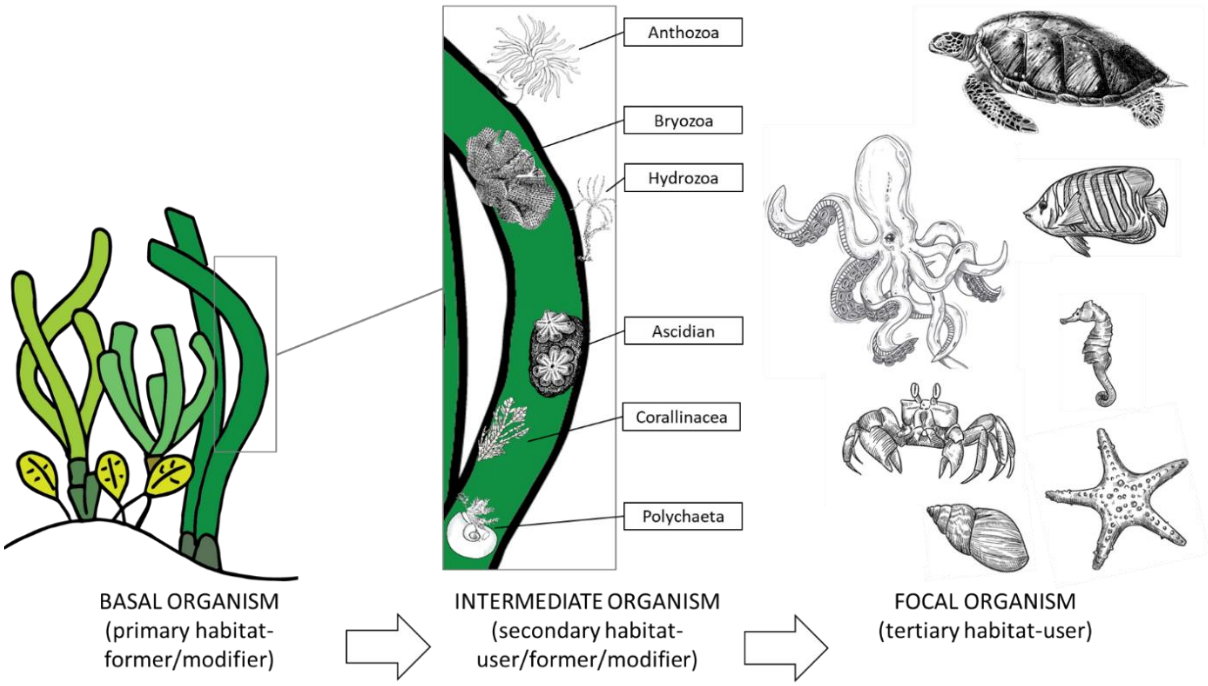
An in-depth knowledge on the biodiversity and its influence on the structure and function of faunal assemblages is a pivotal necessity in the more effective management and conservation of marine resources (see Morin et al. 2018).

Seagrasses ecosystems provide key ecological role as nursery, feeding grounds and habitat for diverse and abundant faunal assemblages (see e.g. Jones et al. 2021, Lin et al. 2018, Marin et al. 2022, Zarco-Perello & Enríquez 2019), and support numerous ecosystem functions and services including provisioning, regulating and maintenance, and cultural services (see IPBES 2019). Despite several studies provide information about seagrass services capacity such as nursery habitat, improved water quality, raw material, coastal protection, and carbon sequestration (e.g. Nordlund et al. 2017, UNEP 2020, Costa 2020, Addamo et al. under review, Figure 7), further ecosystem services (ES) research are still needed to map and assess the ecosystem services including benefits' distribution and trade-off analyses of the beneficiaries (Costa 2020), and to investigate variability of ES within and among seagrass meadows and within the seascape by comparing delivery of services (see Nordlund et al. 2018, Addamo et al., in submission, Harvey et al., in submission, La Notte et al., in submission).

While it remains unclear whether seagrass diversity (both taxonomic and functional) affects the role of seagrass habitats, the species diversity of marine flowering plants seems to have limited effect on seagrass fish assemblages (in terms of abundance and richness), which instead appear to be driven by specific seagrass traits and cover (Jones et al. 2021). Indeed, an increase of structurally complex meadows (e.g. higher canopy, longer and wider leaves, greater numbers of leaves per shoot, and lower overall shoot density) lead to a consequently increase in functional complexity of the habitat and food resource availability, and a reduction of predation pressure (Gullström et al. 2008, Hovel et al. 2002, Jones et al. 2021, Vonk et al. 2010).

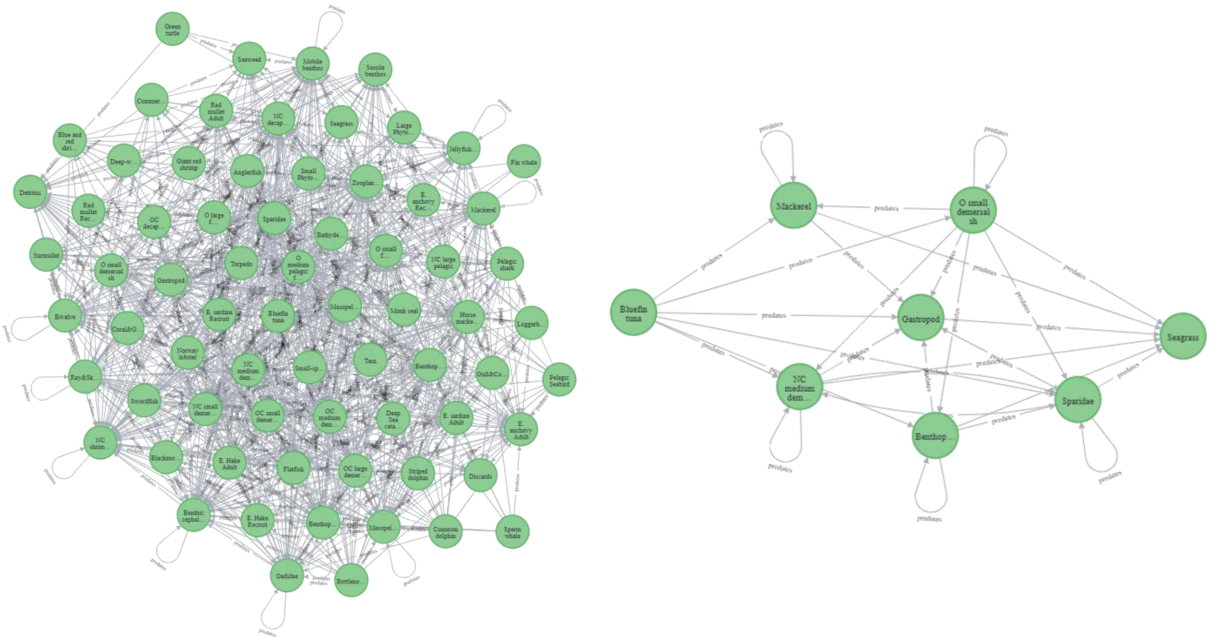
Finally, studies in shallow-water ecosystems such as kelp forests and sandy seagrass beds unravelled and demonstrated habitat cascades as a general phenomenon that enhances species abundance and diversity. Habitat cascades are characterized by a hierarchy of facilitative positive species interactions in which a basal habitat former (typically a large primary producer) as seagrass creates biogenic formation or habitat modification for an intermediate secondary habitat formers (e.g. an epiphyte) that in turn generate living space for the focal tertiary organisms (e.g. invertebrates, fish) (Thomsen et al. 2010, Figure 7).

Figure 7. Habitat cascades from seagrass meadows with three organism levels interacting in succession. *Source:* own elaboration



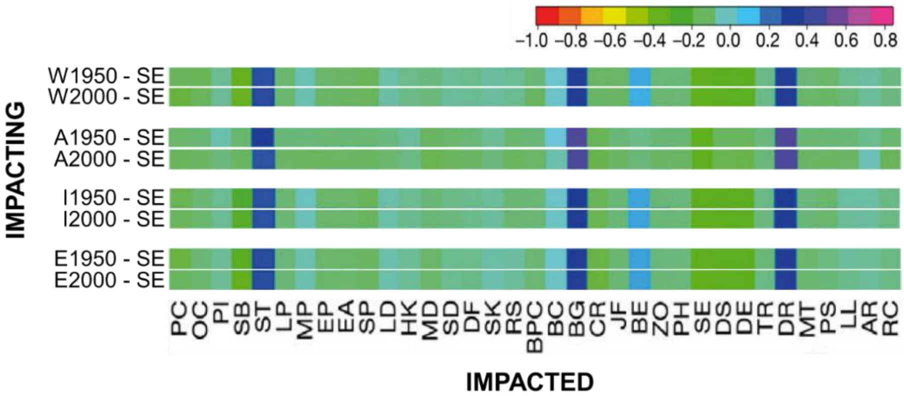
In a broader view, the habitat cascade phenomenon influences another type of indirect interaction in benthic marine ecosystems namely the **trophic cascades**, which are predatory interactions involving three or more trophic levels connected by predation (Pinnegar et al. 2000). A predator decline is a pre-date of habitat decline (Lotze et al. 2006), suggesting that alterations to predator populations may be a major driver of change for coastal systems (Jackson 2001, Jackson et al. 2012) and of maintaining coastal ecosystem services (Atwood & Hammill 2018). The prevalence, strength and direction of trophic cascades varied across ecosystem types: e.g. with predators having a large positive effect on plants in salt marshes, a moderate positive effect on plants in kelp and mangroves, and no effect on plants in seagrasses (Atwood & Hammill 2018). Nevertheless, the paucity of literature on trophic cascades for coastal plant systems, especially seagrasses and mangroves, highlights the need for further research before large-scale generalizations about the trophic cascade effects in coastal plant communities can be made (Atwood & Hammill 2018). Understanding the mechanisms behind diverse ecological networks (e.g. trophic interactions and flows) and the roles of human activities on marine structures and functions is critical when managing marine resources (Cury et al. 2003). The seagrass meadows are one of the most productive coastal ecosystems with important blue carbon sequestration and storage, and crucial habitats and feeding grounds for consumers at high trophic levels (see Lee et al. 2021, Piroddi et al. 2022, Figure 8).

Figure 8. Trophic chain in the Mediterranean Sea among 71 Functional Groups (FG) of four trophic level (TL) (left) and between a top predator (bluefin tuna, TL4) and primary producer (seagrass, TL1). Data on food web retrieved from Piroddi et al. 2022. Arrows represent predator-prey relationships. *Source:* own elaboration.



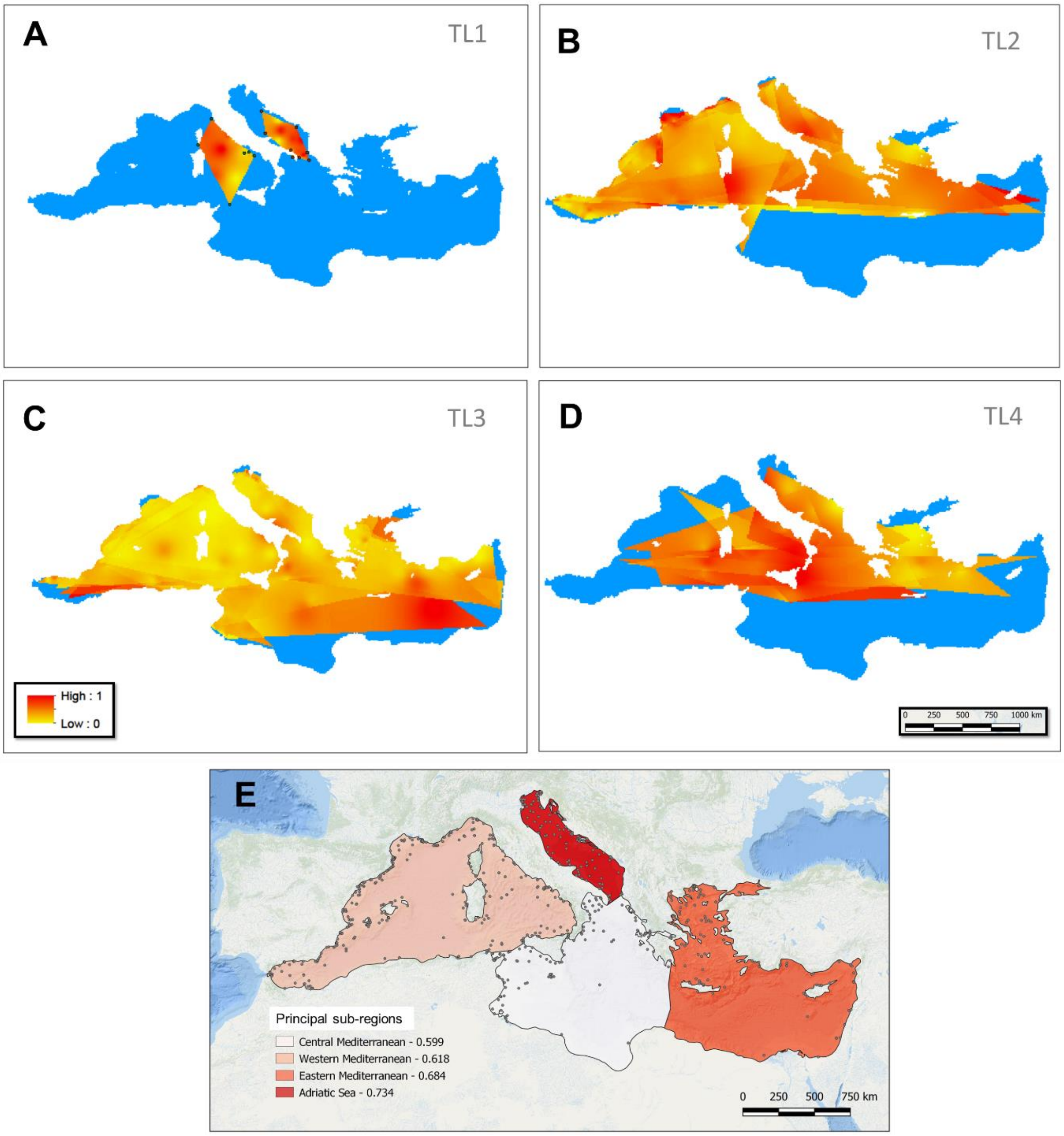
In the attempt to understand and assess the Mediterranean marine ecosystem structures and functions as a whole, Piroddi et al. (2015) used an ecosystem modelling approach, in particular a **mixed trophic impact** (MTI) analysis, to quantify the impact that a theoretical change of a unit in the biomass of a functional group as seagrass would have on other groups (including fishing activities) in the ecosystems (Christensen et al. 2008). In the case of seagrass as other lower trophic level (TL) organisms have been demonstrated to positively affect the rest of TLs characterizing the food web of the Mediterranean Sea (Piroddi et al. 2015, Figure 9).

Figure 9. Mix trophic impact relationships between seagrass (SE) and other functional groups in the four different Marine Strategy Framework Directive (MSFD) areas of the Mediterranean Sea (W= Western; A=Adriatic; I= Ionian/Central; E: Aegean/Levantine) for the years 1950 and 2000. Positive values (from light blue to purple) indicate positive impacts; negative values (from light green to red) indicate negative impacts. The colours should not be interpreted in an absolute sense: the impacts are relative, but comparable between groups. Group abbreviations: PC = piscivorous cetaceans; OC= other cetaceans; PI= pinnipeds; SB= seabirds; ST= sea turtles; LP= large pelagic fishes; MP= medium pelagic fishes; EP= European pilchard; EA= European anchovy; SP= other small pelagic fishes; LD= large demersal fishes; HK= European hake; MD= medium demersal fishes; SD= small demersal fishes; DF= deep-sea fishes; SK= sharks; RS= rays and skates; BPC= benthopelagic cephalopods; BC= benthic cephalopods; BG= bivalves and gastropods; CR= crustaceans; JR= Jellyfish; BE= benthos; ZO= zooplankton; PH= phytoplankton; SE= seagrass; DS= discards; DE= detritus; TR= trawlers; DR= dredges; MT= mid-water trawlers; PS= purse seiners; LL= long liners; AR= artisanal fisheries; RC= recreational fisheries. *Source*: adapted from Piroddi et al. 2015



The interconnectedness as key role in habitat and trophic cascade phenomena reveal the importance of the emerging field of **marine functional connectivity** (MFC). The MFC includes two intertwined components: (a) **structural connectivity**, purely related to the physical characteristics of the seascape, measuring its heterogeneity and structuring (e.g. ice cover, marine currents, chemical barriers); and (b) **functional connectivity**, represents all the movements of organisms that result in the exchange of genes, biomass or energy between heterogeneous habitat patches (Darnaude et al. 2022*a*, Figure 10). Therefore, MFC characterizes all the migratory and passive flows of marine organisms that determines biodiversity patterns and the demographic, ecological and evolutionary interdependency of populations and communities, as well as the flow of energy and organic matter, and ecosystem at sea and at the sea-continent interface (see Darnaude et al 2022*ab*).

Figure 10. Functional connectivity maps on genetic connectivity found at the four trophic levels (TL) throughout the Mediterranean Sea. For TL1 (**A**) only data for the endemic seagrass species *Posidonia oceanica* were found. For TL2 (**B**), TL3 (**C**) and TL4 (**D**), the resulting maps were obtained by merging the genetic data of 42 marine species (18 species for TL2; 18 species for TL3, and 6 species for TL4, see Annex 1). Red colours represent high FST values and thus, high genetic divergence or differentiation (low connectivity). Yellow colours represent low FST values and low divergence (high connectivity). The median values of genetic diversity (**E**) for the main sub-regions (Western, Central, Eastern Mediterranean and Adriatic Sea), was obtained analysing 52 species from 750 different populations and 331 localities distributed throughout the Mediterranean Sea (Annex 1). The general patterns of marine connectivity at trophic levels, showed that even in supposedly great dispersal species, the described barriers between the main sub-regions effectively create some impediments to the gene flow and genetic diversity (**E**). These barriers are permeable since gene flow is shown, although with low values, but differentiation increase with the distance (isolation by distance). The four sub-regions, isolated by barriers more or less permeable to certain organisms, integrate different areas with particular characteristics (currents, fronts, specific habitat factors) that might present contrasting patterns of genetic diversity and connectivity. Note: Repullés et al. (2022) have mapped patterns of genetic connectivity using the Genetic Landscape GIS Toolbox (Perry et al., 2010), developed for terrestrial species genetics, in ArcGIS v9.3 (Environmental Systems Research Institute, Redlands, CA, USA). *Source*: adapted from Repullés et al. 2022.



Humans activities and global change often result in changes in structural and functional connectivity (Darnaude et al. 2022*a*), as well as create, modify, and destroy habitat cascades via global habitat destruction, climatic change, over-harvesting, pollution, or transfer of invasive species (Thomsen et al. 2010). On account of this key role of interconnectedness, the assessment of ecosystem condition should strongly rooted in the concept of **ecosystem integrity,** which is defined as the ecosystem's capacity to maintain its characteristic composition, structure, functioning and self-organisation over time within a natural range of variability (Pimentel et al. 2000, Rendon et al. 2019, United Nations et al. 2021). In particular, **good ecosystem condition** should be considered when represents good physical, chemical, and biological condition, or good physical, chemical and biological quality with self-reproduction or selfrestoration capability, in which species composition, ecosystem structure and ecological functions are not impaired (cf. definition of the Taxonomy Regulation (EU) 2020/8528) (Vallecillo et al. 2022). In this context, the MFC is mentioned as potential variable that needs to be developed for the **seascape characteristics indicator** of habitat fragmentation and ecological corridors (Vallecillo et al. 2022), which maintain necessary demographic transitions in marine species and ecosystems under threat, and under future resilience of food webs as climate and ocean change continues (Peterson et al. 2020). Finally, constructing effective networks of restoration or conservation areas and promoting sustainable harvesting requires knowledge of functional connectivity (Darnaude et al. 2022*a*).

3 Restoration of seagrass ecosystems

Marine ecosystems provides human society with fundamental and relevant functions and services including physical, ecological, economic, social and cultural benefits (IPBES 2019). However, the pressure from anthropogenic activities has degraded the structure and function of marine ecosystems at global scale threatening the services that they support (Halpern et al. 2007, IBPES 2019). Currently, the main countermeasures to marine ecosystems degradation are (i) the management of natural resource through mitigation actions (e.g. by treating urban and industrial wastewater), and (ii) the regulation of human activities and behaviour mainly through legislative instrument (e.g. Environmental Impact Assessment, Strategic Environmental Assessment, etc.), and (iii) conservation measures (e.g. by establishing marine protected areas and reserve zones) (Costa 2021, De'ath et al. 2012, Lotze et al. 2011, Parravicini et al. 2012). However, these measures have not been proving to be sufficient to maintain marine ecosystems health or to reverse marine habitats decline and their resulting planetary health impacts (Luypaert et al. 2020, Talukder et al. 2022).

Seagrass meadows are among the most common biologically rich and highly productive coastal habitats on Earth, covering more than 300,000 km^2 in at least 159 countries, nurturing fish populations, weakening storm surges, nursery habitat to endangered and charismatic species (e.g. dugongs, seahorses, and sea turtles), and providing numerous other services to coastal communities (UNEP, 2020). Unfortunately, seagrass habitats declining rate is estimated 7% per year worldwide, and since the late 19th century, almost 30 % of known seagrass area across the globe has been lost. The main threats to seagrass meadows include urban, industrial, and agricultural run-off, coastal development, dredging, unregulated fishing and boating activities, and climate change (UNEP 2020).

Biodiversity loss in the oceans can be reversed through habitat restoration. Indeed, the cessation of the impact and the implementation of protected areas do not allow a reasonably quick natural recovery of the ecosystems (Perrow and Davy, 2002), while the process of assisting the recovery of a degraded ecosystem is increasingly recognised as an essential tool to pursue marine conservation together with human-behaviour regulation and management (Abelson et al. 2016; Waltham et al. 2020).

Although marine restoration ecology is still lagging behind terrestrial and (freshwater) aquatic ecosystem restoration, data gathered from the growing number of case studies on the recovery of marine ecosystems in the last decades are providing important guides to enhance future recoveries (Papadopoulou et al. 2017). With increased focus on ecological restoration as a conservation strategy, also boosted from the UN Decade on Ecosystem Restoration 2021–2030 which aims to recover 350 million hectares of degraded ecosystems globally by 2030 (Waltham et al. 2020), a variety of methods that enhance restoration success need to be explored (Costa 2021, Valdez et al. 2020, Box 1).

Box 1. Synthesis of best practices on seagrass restoration methodologies

- Transplantation and gardening of asexual materials such cuttings, rhizome fragments or cores (Davis & Short 1997), and adults/shoots (Meehan and West, 2002; Eriander et al., 2016: Calvo et al., 2021) collected from donor meadows with or without a nursery phase. Adults are typically used in exposed areas where seeds would not resist at the water movement (Papadopoulou et al., 2017; Pajusalu et al., 2019).
- Transplantation and gardening sexually derived propagules[1] (Statton et al., 2012; van Katwijk et al., 2016), seeds (Marion and Orth, 2008; Orth et al., 2012; Infantes et al., 2016), and seedlings (Infantes et al., 2011) with or without a nursery phase.
- Transplantation of entire sods e.g. including sediment (Uhrin et al., 2009; Matheson et al., 2016).
- Substrate enhancement methods by: adding artificial structures such as biodegradable mesh (Kidder et al., 2015), hessian bags (Tanner, 2015), shells (Lee and Park, 2008), rope (Pajusalu et al., 2019), metal, bamboo, wood (Papadopoulou et al., 2017) or bivalves (Bekkby et al., 2020); stabilizing the substrate, removing algae mats, providing aeration to improve oxygen conditions in the sediment (Papadopoulou et al., 2017); reducing the nutrient loading to the area (Peralta et al., 2003) or, on the contrary, adding sediment fertilization in areas that are nutrient-depleted (Balestri and Lardicci, 2014).

[1] The term propagule refers to structures that act to propagate an organism to the next stage in its life-cycle, and can be either sexually produced structures such as fruits, seeds and larvae, or clonally produced structures.

Source: Costa (2021)

Despite to be 10–400 times more expensive than the restoration costs documented for terrestrial ecosystems, with an estimated median cost of USD 106,782 (equiv. EUR 97,745) per hectare in a global context (Tan et al. 2020), seagrass restoration have shown a success rate of 38% worldwide (Bayraktarov et al. 2016, Costa 2021, Box 2). Moreover, decades of seagrass restoration have also demonstrated the beneficial effects of positive species relationships and feedbacks for seagrass ecosystem stability, expansion, and recovery from disturbance (Tan et al 2020, Valdez et al. 2020). Since oceans continue to change and stressors become more prevalent, harnessing positive interactions between species through innovative approaches will likely become key to successful seagrass restoration (Valdez et al. 2020).

Box 2. Synthesis of main findings from seagrass restoration studies

- Success rate of seagrass meadow restorations has been relatively low worldwide (38%) (Bayraktarov et al., 2016). Restoration using seagrass propagules has so far demonstrated low and variable outcomes, with more than 90% of propagules failing to survive (Tanner, 2015).

- Reduced water quality (mainly eutrophication), and construction activities lead to poorer restoration success than, dredging, local direct impact and natural causes (van Katwijk et al., 2016).

- Low survival rates appear to be related to limited knowledge about availability and collection of quality seed, skills in seed handling and delivery, and suitability of restoration sites (Statton et al., 2012; Orth et al., 2012; van Katwijk et al., 2016; Kendrick et al., 2017).

- Poor site selection has been identified as the major limitation on survival of transplanted seagrass (Bastyan and Cambridge, 2008; Pirrotta et al., 2015; van Katwijk et al., 2016; Calvo et al., 2021). Information about suitability of an area to be restored and the potential for success might be not completely reliable because they have been based on data from short-term monitoring (Pirrotta et al., 2015; Lanuru et al., 2018).

- The majority of the projects are short-terms making difficult to assess performance. Long-term monitoring are a few (Bastyan and Cambridge, 2008; Pirrotta et al., 2015).

- Survival and growth rates of transplanted seagrass are positively affected by the number of plants or seeds initially transplanted suggesting the requirement of a (usually large) critical mass for recovery (van Katwijk et al., 2016). There is a positive correlation between proximity to and recovery of donor beds (van Katwijk et al., 2016). Successful transplantation seems to occur into areas that were previously vegetated with seagrass (Bastyan and Cambridge, 2008).

- Transplanting may not replace lost meadows but could usefully enrich zones where seagrass had been destroyed, such as pipeline scars and shoreline developments (Bastyan and Cambridge, 2008).

- Anchoring the planting units is essential (Bastyan and Cambridge, 2008).

- Transplant success is influenced by the nature of the substratum. For example for *Posidonia* bottoms with dead matte habitat show higher survival and growth rates of transplant shoots than sand, pebble and rock (Terrados et al., 2013; Lanuru et al., 2018; Calvo et al., 2021).

- Most transplantation methods are labour intensive and time consuming, generally limiting the study to small areas e.g. tens to hundreds of square meters (Fishman et al., 2004).

- Mechanical tools have been proved successful to reduce human labour during restoration projects such as mechanical harvesters for species which have seeds contained within spathes (e.g. *Zostera*) (Orth et al., 2012; Pickerell et al., 2005), or shakers and collectors for species that release seeds from fruits that float (e.g. *Posidonia*, *Halophila*) (Statton et al., 2012; Statton et al., 2013).

- To improve the sustainability of plant material samples to be used for transplanting activities, rhizomes and cuttings accumulate during fall/winter can be collected easily in large numbers (Balestri et al., 2011).

Source: Costa (2021)

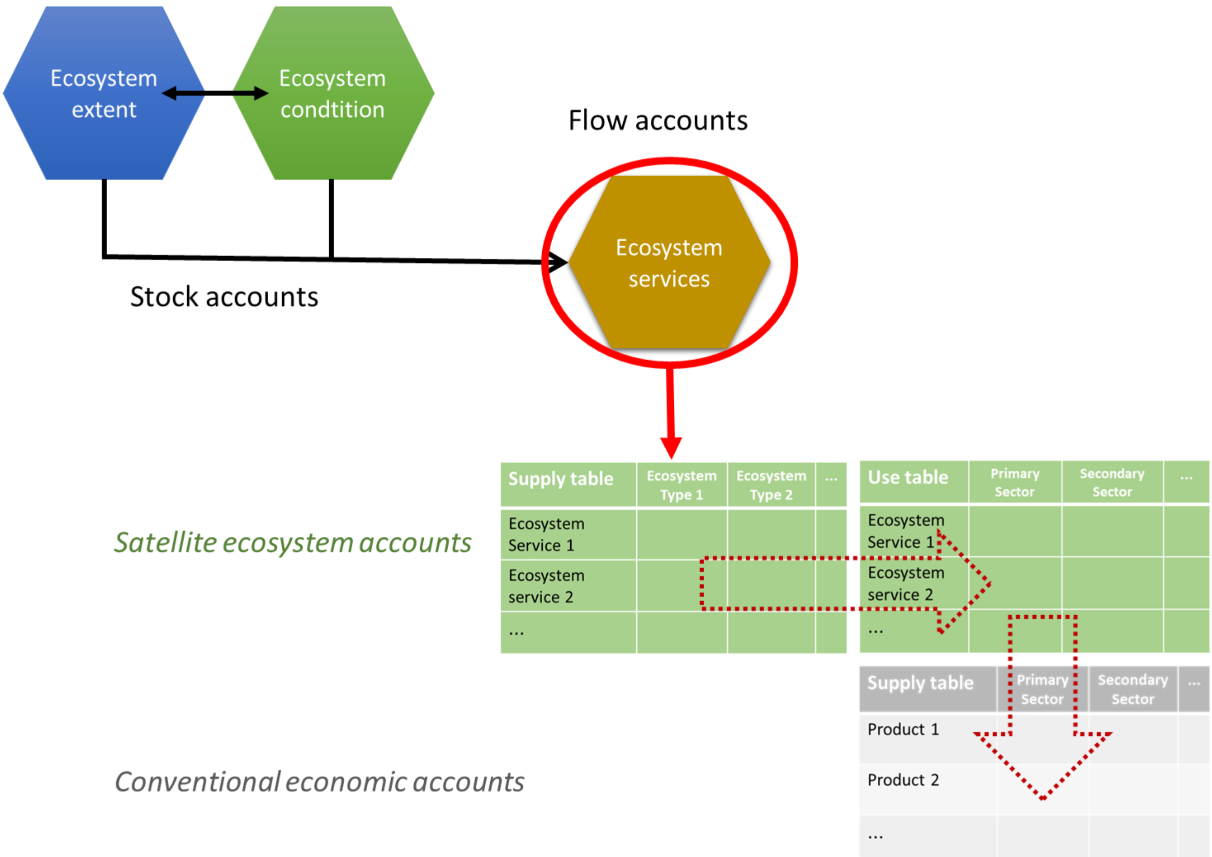
4 Context of marine ecosystem accounts

There is the need to mainstream ecological content into economic contexts to become increasingly policy effective. One framework that facilitate such integration is the System of integrated Environmental and Economic Accounts – Ecosystem Accounting (SEEA EA). SEEA EA has a long history of methodological and implementation work and debate (Edens et al. 2021) that has ultimately generated a standard officially adopted by the UN Statistical Commission (ref 52/108 in UNSC 2021).

Although the SEEA EA explicitly report Ocean Accounts (UN 2021), this critical thematic area still needs more focused work, and in this respect the Global Ocean Accounts Partnership proposes the Technical Guidance on Ocean Accounting v.1.0 (GOAP 2021), which focus specifically on marine ecosystem accounts. The Global Ocean Accounts Partnership (GOAP) work is compliant with SEEA EA and is driving several applications (Addamo et al. in submission, Giacutan et al. 2022, Grilli et al. 2021, La Notte et al. in submission, Mengo et al. 2022).

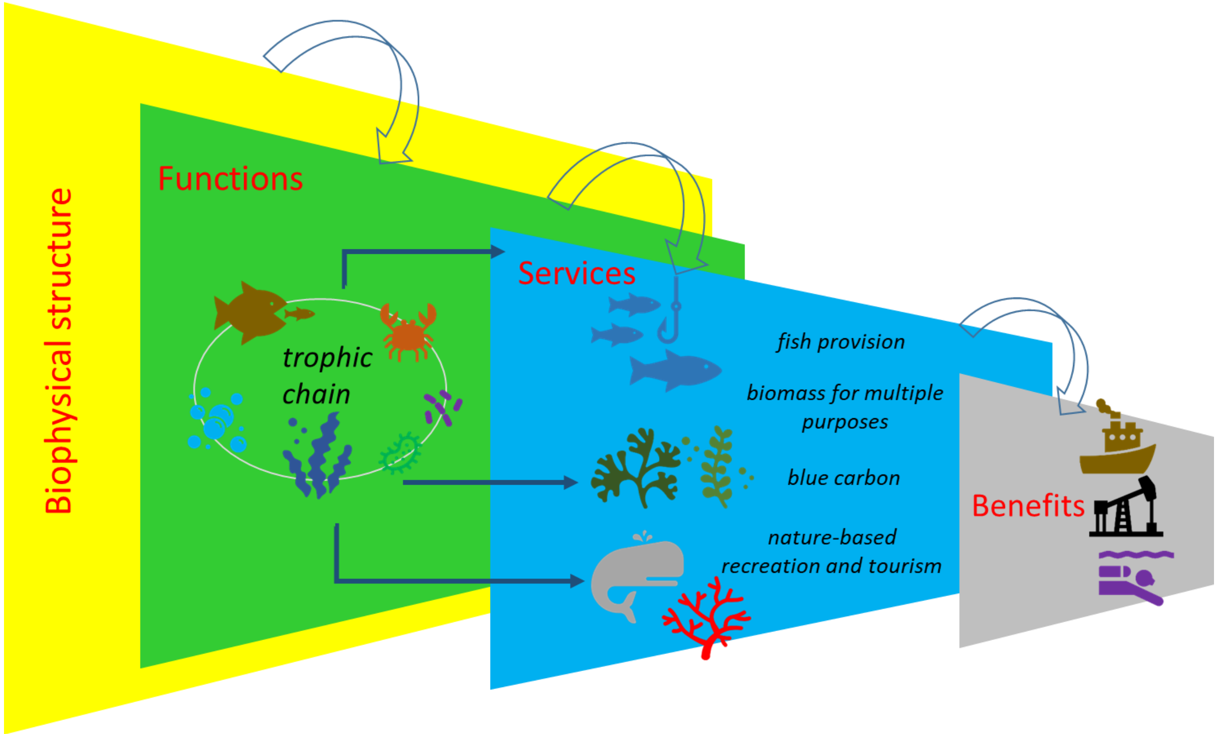
In the SEEA EA framework, status and changes in ecosystems are accounted through extent and condition accounts, and the services they provide are reported in Supply and Use tables that constitute an entry point into the economic system (Figure 11).

Figure 11. Ecosystem accounting satellite framework. *Source*: adapted from United Nations et al. 2021



There is a link between extent, condition and ecosystem services (La Notte et al. 2022) although not operationally defined in the SEEA EA general framework (UN 2021). Such link implies that good condition of, for example, seagrass habitats will affect the quantity and quality of ecosystem services provided. There is in fact a cascade model (Haines-Young & Potschin 2018) that connects biophysical structure to function, and in turn to service and benefits (Figure 12).

Figure 12. The "telescopic" cascade model applied to marine ecosystem services. *Source*: adapted from La Notte et al. 2017



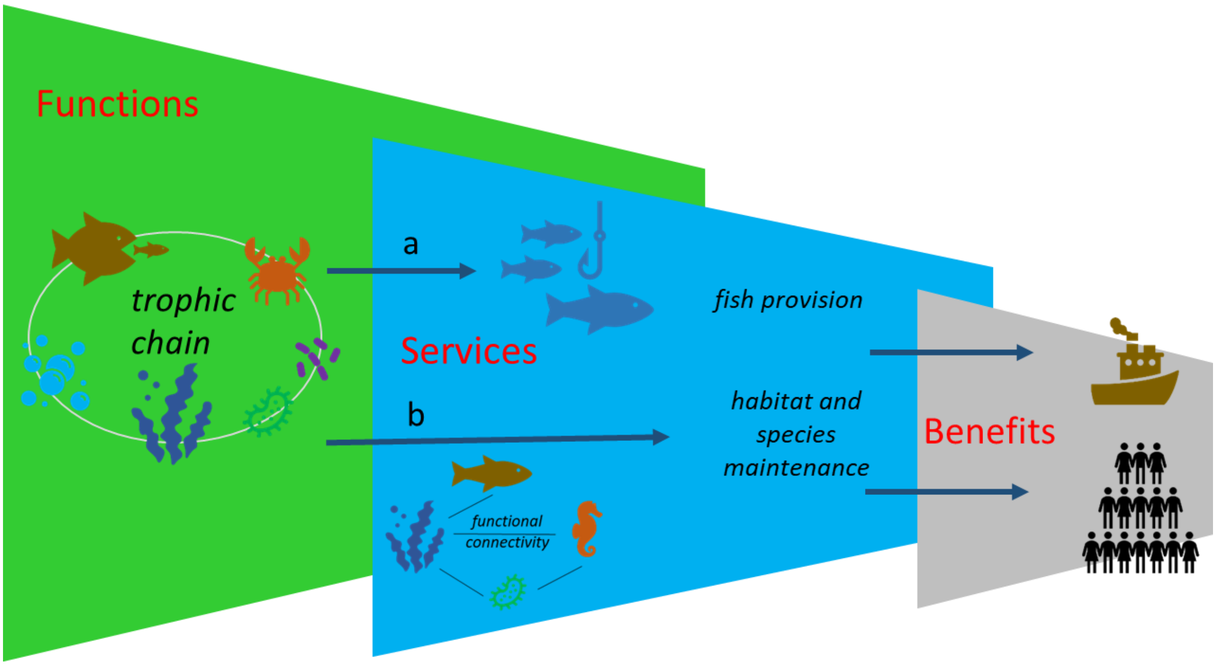
Extent and condition accounts mainly concern the biophysical structures and functions within the cascade model. However, the role of functional connectivity in this cascade may be double. It could in fact be: (i) the *intermediate service* e.g. "nursery population and habitat maintenance services" when they provide support to other services such as fish provision or nature-based recreation; (ii) the *final service* e.g. "ecosystem and species appreciation", already applied for terrestrial ecosystem as habitat and species maintenance (ref. chapter 3 in La Notte et al. 2021).

In the case of the former intermediate service, the ecosystem connectivity contribution will quantitatively adapt the assessment techniques that better fit the final service. For example, seagrass feed organisms that in turn feed fishes eventually caught by vessels. The role of seagrass could be disentangled as a proportion of the total catch (as done in Addamo et al., in submission*,* La Notte et al., in submission).

In the case of the latter final service, functional connectivity needs itself an *ad hoc* indicator that best reflects the existence value that people attribute to the survival of future generations linked to a well-functioning marine ecosystem. For example, genetic connectivity could provide the biophysical assessment to drive an economic estimate of marine habitat and species maintenance.

Figure 13 visually summarizes the two ecosystem services: the intermediate flow (a) and the final flow (b).

Figure 13. Role of functional connectivity as intermediate (a) and final (b) service. *Source:* own elaboration

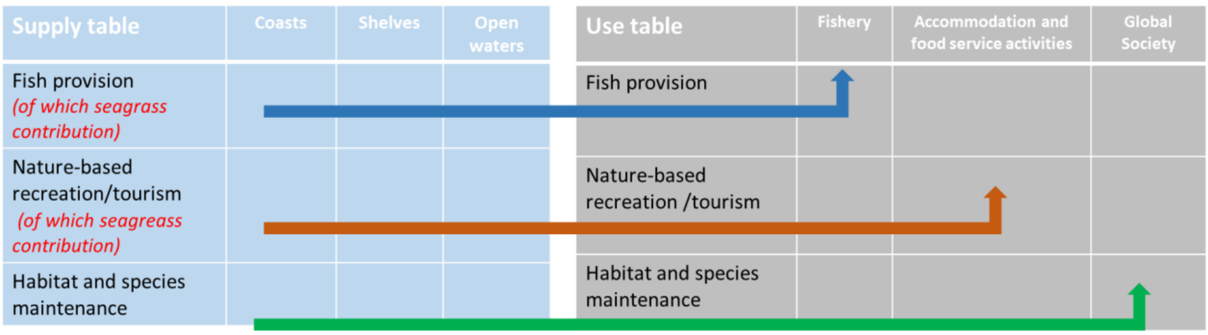


The combination of the two roles of functional connectivity does not represent a double counting because in the case of:

- intermediate service, the functional connectivity is not added, but disentangled from other ecosystem services, which are assessed and valued with their own techniques. Intermediate services are not accounted as a separate flow, but are considered embedded in what they contribute to generate;
- final service, the functional connectivity is added as non-use value to which people attribute a value, through *ad hoc* taxation or voluntary donations. This value if part of the utility function of people and therefore to be considered as a final service. Habitat and species maintenance can be considered as part of the overarching environmental target "halt biodiversity loss". In the Use table, this service is going beyond national boundaries are sectors, to what the INCA project[1] identifies as "Global Society".

Figure 14 shows the accounting table format for Supply and Use tables, with special reference to the above mentioned cases.

Figure 14. Allocation of different marine ecosystem services in the Supply and Use tables. *Source:* own elaboration



[1] Integrated Natural Capital Accounting https://ecosystem-accounts.jrc.ec.europa.eu/

The reader should note that the Use table only reports the primary user. This implies that for fish provision, only fishermen and vessels will be considered, and any further transformation and selling of sea food and products will be accounted in a sequence of Supply and Use tables that are already part of the System of National Accounts (the conventional accounts represented in Figure 11). In the case of nature-based tourism, the primary user is the accommodation and food service activities that eventually serve domestic and foreign visitors, and this is accounted in another Supply and Use tables. Finally, for habitat and species maintenance, the Global Society is already a final user because this ecosystem service addresses a global target involving the whole planet.

Although the number of ecosystem accounting application is rapidly growing, there is still a big gap about marine ecosystems. Present and future attempts need to quickly advance knowledge, methodology, and case studies to provide practitioners with clear guidelines and examples on such complex and policy key information tool.

5 Conclusions and recommendations

The importance of ecosystem-based studies is more than just large size (Mitsch & Day Jr. 2004). Indeed, including all processes of the ecosystem at spatial and temporal scale means to incorporate all direct and indirect, biotic and abiotic features that shape the interconnected dynamics of habitat and species.

An ecosystem-based approach would also allow reaching long-term achievements of the targets with a cost-benefit balance of the effort required by the EU legislations.

The attempt of mainstreaming ecological content into economic context takes advantages from integrated environmental and economic accounts, which in turn need indicators of functional connectivity for a variety of purposes including the economic assessment of overarching environmental goals, such as halting biodiversity loss.

As demonstrated in this report, several descriptors from genetic diversity and functional connectivity provide critical contribution to the conservation and sustainable use of marine biodiversity and resources.

Such measurements could in fact be accounted as condition indicators of the marine ecosystems. The next actions foreseen as a step-wise process would be in fact include the following steps as recommendations for empirical applications of marine accounts:

- set a system for monitoring and assessment of such indicators and report them as an asset accounts (according to the SEEA EA standard, United Nations et al. 2021);
- operationalize the linkages with the ecosystem services that directly depend on functional connectivity. This step would require to identify all those services and make sure that input variable in modelling procedures include (directly or indirectly) functional connectivity indicators;
- account for ecosystem services depending on functional connectivity in the form of Supply and Use tables (according to the SEEA EA standard, United Nations et al. 2021), where the user of each service (whether an economic activity, households, government) is clearly identified;
- develop an additional service named “habitat and species maintenance” where functional connectivity is the key variable to assess the existence value people are willing to pay for the survival of future generations. Such service would address the overarching environmental target that is “halting biodiversity loss” and its beneficiary would the Global Society;
- translate the outcomes of biophysical models in monetary terms for the all ecosystem services, and eventually estimate a “virtual stock” value, by applying the Net Present Value formula;
- develop integrated environmental-economic indicators to stimulate an interdisciplinary policy discussion.

By developing this set of marine accounts, it would be possible not only to avoid the current underestimation of marine ecosystem contribution to economy and society, but also to introduce ecological principles (such as marine functional connectivity) that matter for the economy in a cause-and-effect relationship.

Any application on marine ecosystem condition and marine ecosystem services accounting would actively contribute to the on-going GOAP initiative and SEEA EA working group on Marine Accounts. There is a lot to develop for marine and ocean ecosystem accounts, and there is urgency to fill this important data gap.

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List of abbreviations and definitions

CBD	Convention on Biological Diversity
GBF	Global Biodiversity Framework
EMODnet	European Marine Observation ad Data Network
EOV	Essential Ocean Variable
FST	F-statistics (i.e. a measure of population differentiation due to genetic structure)
GIS	Geographic Information System
GOAP	Global Ocean Accounts Partnership
HSM	Habitat Suitability Model
INCA	Integrated Natural Capital Accounting
MFC	Marine Functional Connectivity
MTI	Mixed Trophic Impact
SDG	Sustainable Development Goal
SEEA EA	System of Environmental-Economic Accounting Ecosystem Accounting
TL	Trophic Level

List of boxes

List of figures

List of tables

Annexes

Annex 1. List of marine species used in genetic diversity and connectivity maps (see Figure 10).

List of species analysed in Repullés et al. (2022), with their corresponding trophic level, functional group and functional group number (#) to which they belong, according to Piroddi et al. (2017). The column headed "Used for" indicates whether the species are considered for genetic diversity analysis (GD) and/or genetic connectivity analysis (GC). *Source*: Repullés et al., 2022

N	***Species name***	**Trophic Level**	**Functional group**	**#**	**Used for**
1	*Posidonia oceanica*	1	Seagrass	26	GD, GC
2	*Arbacia lixula*	2	Benthos	23	GD, GC
3	*Astroides calycularis*	2	Benthos	23	GD, GC
4	*Botryllus schlosseri*	2	Benthos	23	GD, GC
5	*Calanus helgolandicus*	2	Zooplankton	24	GD, GC
6	*Cerastoderma glaucum*	2	Bivalves	20	GD, GC
7	*Cladocora caespitosa*	2	Benthos	20	GD
8	*Clavelina lepadiformis*	2	Benthos	23	GC
9	*Coscinasterias tenuispina*	2	Benthos	23	GD, GC
10	*Desmophyllum dianthus*	2	Benthos	20	GD
11	*Eunicella cavolini*	2	Benthos	23	GD, GC
12	*Eunicella singularis*	2	Benthos	23	GD
13	*Gibbula divaricata*	2	Benthos	23	GD, GC
14	*Hediste diversicolor*	2	Benthos	23	GD, GC
15	*Meganyctiphanes norvegica*	2	Zooplankton	24	GD, GC
16	*Paracentrotus lividus*	2	Benthos	23	GD, GC
17	*Paramuricea clavata*	2	Benthos	23	GD, GC
18	*Petrosia ficiformis*	2	Benthos	23	GD, GC
19	*Pinna nobilis*	2	Benthos	23	GD, GC
20	*Pseudodistoma crucigaster*	2	Benthos	23	GD, GC
21	*Sagitta setosa*	2	Zooplankton	24	GD, GC
22	*Dendropoma lebeche*	2	Benthos	23	GD, GC
23	*Aristeus antennatus*	3	Crustaceans	23	GD, GC
24	*Mullus barbatus*	3	Small demersals	14	GD, GC
25	*Mullus surmuletus*	3	Small demersals	14	GD, GC
26	*Sparus aurata*	3	Medium demersals	13	GD, GC

N	*Species name*	Trophic Level	Functional group	#	Used for
27	*Pelagia noctiluca*	3	Jellyfish	22	GD, GC
28	*Scomber scombrus*	3	Medium pelagics	7	GD, GC
29	*Scomber japonicus*	3	Medium pelagics	7	GD
30	*Sarda sarda*	3	Medium pelagics	7	GD, GC
31	*Trachurus trachurus*	3	Other small pelagics	10	GD
32	*Engraulis encrasicolus*	3	European anchovy	9	GD, GC
33	*Dentex dentex*	3	Medium demersals	13	GD, GC
34	*Dicentrarchus labrax*	3	Medium demersals	13	GD, GC
35	*Auxis rochei*	3	Medium demersals	13	GD
36	*Sardina pilchardus*	3	European pilchard	9	GD, GC
37	*Gobius niger*	3	Small demersals	14	GD, GC
38	*Pagellus erythrinus*	3	Small demersals	14	GD, GC
39	*Homarus gammarus*	3	Crustaceans	21	GD, GC
40	*Palinurus elephas*	3	Crustaceans	21	GD, GC
41	*Palaemon elegans*	3	Crustaceans	21	GD, GC
42	*Caretta caretta*	3	Sea turtles	5	GD, GC
43	*Raja asterias*	3	Ray and skates	17	GD, GC
44	*Delphinus delphis*	4	Piscivorous cetaceans	1	GD, GC
45	*Merluccius merluccius*	4	European hake	12	GD
46	*Octopus vulgaris*	4	Benthic cephalopods	20	GD, GC
47	*Prionace glauca*	4	Sharks	16	GD, GC
48	*Scyliorhinus canicula*	4	Sharks	16	GD, GC
49	*Thunnus alalunga*	4	Large pelagics	6	GD
50	*Thunnus thynnus*	4	Large pelagics	6	GD, GC
51	*Tursiops truncatus*	4	Piscivorous cetaceans	1	GD, GC
52	*Xiphias gladius*	4	Large pelagics	6	GD
53	*Monachus monachus*	4	Pinnipeds	3	GD

Annex 2. References cited in Box 1 and Box 2

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