Innovative and practical tools for monitoring and assessing biodiversity status and impacts of multiple human pressures in marine systems

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Abstract
Human activities at sea can produce pressures and cumulative effects in ecosystem components, that need to be monitored and assessed, in a cost-effective manner. Five Horizon European projects have joined forces to collaboratively increase our knowledge and skills to monitor and assess the ocean in an innovative way, assisting managers and policy-makers in taking decisions to maintain sustainable activities at sea. Here we present and discuss the status of some methods revised during a summer school, aiming at a better management of coasts and seas. We include novel methods to monitor the coastal and ocean waters (e.g. environmental DNA, drones, imaging and artificial intelligence, climate modelling and spatial planning) and innovative tools to assess the status (e.g. cumulative impacts assessment, multiple pressures, Nested Environmental status Assessment Tool (NEAT), ecosystem services assessment, or a new unifying approach). As a concluding remark, some of the most important challenges ahead is assessing pros and cons of novel methods, comparing them with benchmark technologies, and integrating these into long standing time series for data continuity. This requires transition periods and careful planning, which can be covered through an intense collaboration of current and future European projects on marine biodiversity and ecosystem health.

Keywords: cumulative effects; environmental status; eDNA; drones; artificial intelligence; imaging

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Not applicable, since no data are associated to this manuscript.

Introduction

Human needs (e.g., food, energy, recreation) result in an increasing number of activities at sea, including fishing, aquaculture, shipping, infrastructure construction (e.g., coastal defence, renewable energies, oil and gas platforms, etc.), as well as on land-based activities (Dailianis et al., 2018). These human activities lead to pressures affecting various ecosystem components; this may result in the degradation of ocean health (Halpern et al., 2008a, 2008b; Reker et al., 2019; Korpinen et al., 2021) which in turn can result in a diminution in ecosystem services supply and, ultimately, affect human well-being and health (Borja et al., 2020).

Despite efforts to minimize human pressure effects in marine ecosystems and their services (e.g. European Commission et al., 2022), these effects will most likely increase with increasing use of the ocean (Nash et al., 2020). Among the main legal instruments available to prevent and mitigate human pressures and impacts, and promote the sustainable use of European seas, are the Maritime Spatial Planning Directive (MSPD; European Union, 2014) and the Marine Strategy Framework Directive (MSFD; European Commission, 2008). Their implementation should support a sustainable blue economy, which is crucial to achieving the Biodiversity Strategy targets for 2030 (European Commission, 2020) and the Good Environmental Status (GES) under the MSFD, by 2026. Also, when those objectives are not achieved, the Nature Restoration Law defines targets to restore degraded habitats in terrestrial, aquatic and marine systems, by 2030 and 2050 (Hering et al., 2023). To make development and conservation compatible, instruments are needed (European Commission, 2021), including innovative and practical monitoring and assessment methods that can assist managers in decisions making.

Several examples of reviews of methods for monitoring (Danovaro et al., 2016; Mack et al., 2020), modelling (Lynam et al., 2016; European Commission, 2022) or status assessment already exist (Borja et al., 2016). Materializing some of the developed ideas, in the last 2-3 years, some Horizon Europe projects have been initiated, providing the opportunity to dramatically increase our knowledge and skills to monitor and assess the ocean with new tools (Borja et al., 2024). Five of such projects (GES4SEAS[1], OBAMA-

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NEXT[2], BIOcean5D[3], ACTNOW[4], MARBEFES[5]) joined forces in a summer school held in San Sebastián (Spain), from 5th to 7th June 2023, to revise and discuss with more than 50 attendees, the innovative methods that are being developed and the ways in which these can assist managers and policy-makers. This manuscript presents the status of the most important methods reviewed during the school and discusses the progress they represent towards a better management of the seas, in Europe and beyond. The manuscript is divided in three sections: (i) novel methods to monitor coastal and ocean waters (e.g. environmental DNA (eDNA), drones, imaging and artificial intelligence, climate modelling and spatial planning); (ii) innovative tools to assess the status (e.g. cumulative impacts assessment, multiple pressures, Nested Environmental status Assessment Tool (NEAT), ecosystem services assessment, or a new unifying approach); and (iii) discussion on the next steps in monitoring and assessing marine biodiversity and ecosystem health.

Some novel methods to monitor the ocean

Analyzing eDNA to monitor different ecosystem components at sea

eDNA refers to DNA collected from environmental samples (Pawlowski et al., 2020). A few liters of marine water contain organisms such as bacteria, microbial eukaryotes or small planktonic animals, but also traces of larger organisms such as fish or marine mammals in form or scales, tissues, cells or gametes (Rodriguez-Ezpeleta et al., 2021a). Thus, sampling and then analyzing eDNA from marine water allows studying the diversity and abundance of several ecosystem components. This approach has the potential to revolutionize marine monitoring, which often requires time consuming surveys involving expensive sampling gear not suitable for all locations and depths.

Using eDNA for biomonitoring involves several steps upon which the accuracy of the obtained results largely relies (Rodriguez-Ezpeleta et al., 2021b). Sampling can involve collecting water from the desired location and depth and then passing it through a filter of a given pore size. eDNA is then extracted from the filtrate and used for species specific assays or for metabarcoding (Figure 1). Species specific assays consist of

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applying quantitative or digital PCR (qPCR/dPCR) to detect and quantify the DNA from a species of interest for which the assay is specifically designed (Salter et al., 2019). This approach is particularly interesting for early detection of non-indigenous species (Nathan et al., 2014) or for monitoring movements and spawning activity (Erikson et al., 2016), and it has recently been proven as a promising approach for fisheries surveys, providing similar outcomes to those obtained through active acoustics (Shelton et al., 2022). Alternatively, DNA metabarcoding allows to study the whole community of a given taxonomic group. Using this approach, the DNA from a given ecosystem component (e.g. cephalopods, fish, …) is amplified and sequenced; the resulting DNA reads are then compared against a reference database that contains the correspondence between DNA sequences (barcodes) and species names. This approach is particularly interesting for monitoring diversity and relative abundance patterns, including changes across time and space. This method has proven to detect systematically more species than traditional approaches, such as plankton nets, trawling, video, etc. (Fraija-Fernández et al., 2020).

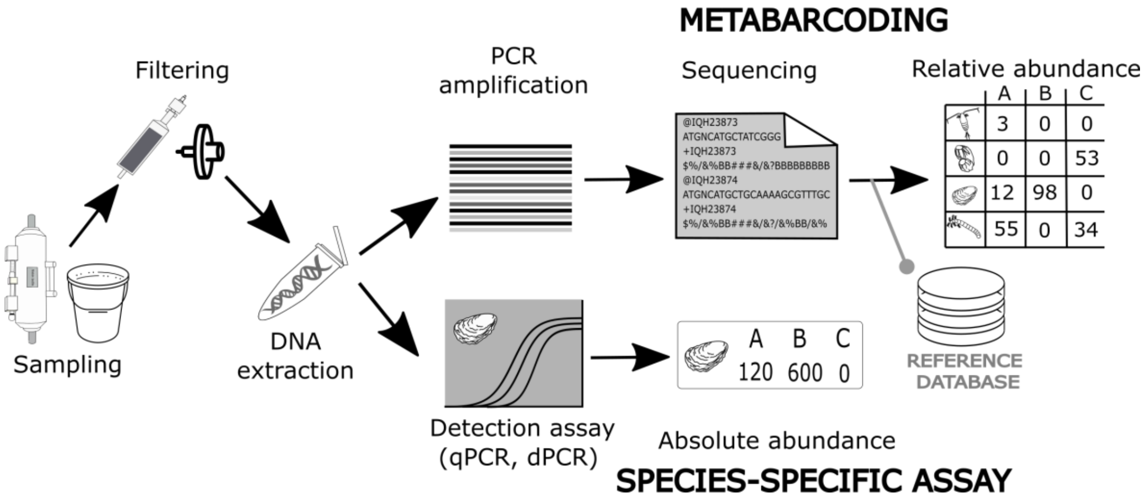

Figure 1: Schematic representation of the steps involved in the process of using environmental DNA for biodiversity assessment. Water samples are collected using Niskin bottles or less sophisticated gear and then filtered before environmental DNA extraction. For the metabarcoding process, environmental DNA is amplified using group specific primers and then sequenced; resulting sequences are compared against a reference database to obtain the relative DNA abundance (percentage of sequences) of each species. In the species-specific assay process, environmental DNA is amplified using a species-specific quantitative or digital PCR assay to obtain the absolute abundance (number of DNA copies per liter) of the target species.

Despite the demonstrated power of eDNA based analyses for biodiversity monitoring, there are several inherent challenges associated to the use of eDNA as a proxy of species presence. Several researchers have established the term "ecology of eDNA"

(Barnes and Turner, 2016) to refer to the origin, state, transport and fate of eDNA within the environment, factors that should be account when interpreting eDNA derived biodiversity assessments. Indeed, finding eDNA from a species at a given location is not always indicative that the species was there, alive at that moment (Burian et al., 2021). Yet, despite recognized uncertainties, experiments in controlled environments and open ocean suggest that eDNA detection is a good proxy of species presence (Holman et al., 2021), which is also supported by studies revealing the power of eDNA to detect species vertical structuring (Canals et al., 2021). In order to go beyond species lists with uncertain interpretation, it is now time that the knowledge accumulated over 15 years of using eDNA for studies of macroorganismal biodiversity is coupled with models that account for the factors that shape the "ecology of eDNA".

eDNA studies are also affected by the inherent biases of the species-specific and metabarcoding assays. Some of them, such as contaminations, can be controlled with appropriate laboratory practices (Burian et al., 2021), while others such as variable DNA extraction efficiency, inhibition in DNA amplification, low specificity of the developed assay (non-target species are also detected) or primer bias (under and over amplification of DNA of some species) are difficult to detect. An important aspect that mostly affects metabarcoding based studies is the lack of completeness and accuracy of reference databases, which can severely bias obtained results (Claver et al., 2023); thus, it is advised that a custom-curated database is used as reference and that, when not complete, an assessment of its completeness is performed.

Besides advances in improving laboratory and data analyses protocols as well as models for data interpretation, advances in eDNA research include the use of autonomous sampling (Preston et al., 2023) and *in situ* analysis (Hansen et al., 2020) devices, the use of eDNA for population genomics (Sigsgaard et al., 2019), improvements in the use of eDNA approaches for biomass estimation (Shelton et al., 2022) and other advances such as estimating the developmental status of the targeted species (Parsley and Goldberg, 2023). Thus, considering also advances in overcoming the challenges associated with standardization and reproducibility, it is anticipated that the position of eDNA as a game-changer in environmental monitoring will be further consolidated in the future.

Drones for mapping and monitoring shallow water benthic communities

In shallow water benthic habitats, ecosystem services are largely linked to the presence and livelihood of marine vegetation such as seagrass, rockweeds, kelp forests and tidal marshes (Krause-Jensen et al., 2022). Standard techniques to monitor and map the coastal zones include satellite remote sensing, research vessels, small boats, and diving operations. However, these methods often come short for shallow water coastal zones. Satellite remote sensing offers large spatial coverage but low spatial resolution, with RGB and spectral data acquisition at resolution of tens to hundreds of meter scales (Lønborg et al., 2022). Larger research vessels are limited to water depths deeper than ~10 meters and have high running costs. Smaller boats can access shallow waters at lower costs but with limited spatial coverage and a significant time investment. Similarly, SCUBA diving for underwater mapping and monitoring is also expensive and time-intensive (Féral and Norro, 2023). The latter two, in addition, often suffer from not being spatially explicit (Murphy and Jenkins, 2010).

Flying drones offer a novel technology and a strong complement to these existing methodologies and is filling a gap by providing cost-efficient data collection with a high spatial and temporal resolution (centimeter-scale and on-demand data collection), capabilities to operate below cloud cover, and facilitating systematic and repetitive data acquisition for shallow water research and monitoring purposes (Joyce et al., 2023).

Over the past few years, there has been an explosion in the technical development of flying drones and a following steep development of applications in research, mapping and monitoring of aquatic habitats (Joyce et al., 2023). Flying drones are scientifically often referred to as Unoccupied, Uncrewed, or Unmanned Aerial Vehicles (UAV). Or even Unmanned Aerial Systems (UAS) or Remotely Piloted Aircraft Systems (RPAS), as frequently used in the aviation industry. We suggest using the gender-neutral Unoccupied or Uncrewed Aerial Vehicles (UAV) in science and management context, as advocated for by Joyce et al. (2021).

UAVs are well underway changing the game of shallow water benthic habitat mapping, by facilitating efficient, precise, and reproducible data collection (Joyce et al., 2023). Using sophisticated light-weight sensors and high-resolution cameras, combined with artificial intelligence (AI) for data analysis and high-performance computers, research communities as well as management and mapping authorities now have tools at hand to capture detailed geospatial data of coastal and aquatic communities with ease. By providing a bird's-eye perspective on coastal ecosystems from a much lower altitude than satellites, drones can capture data at ~100 times better resolution than satellites, with a flexibility in timing and frequency allowing studies at certain tide stages, or before and after events (Casella et al., 2017), even in hazardous environments. Drones have already shown their efficiency and provide unique data and novel insight to a range of coastal subjects, such as mapping of seagrass (Duffy et al., 2019), kelp forests (Cavanaugh et al., 2021), coastal wetlands (Doughty and Cavanaugh, 2019), abundance and community dynamics of seabirds (Edney et al., 2023) and marine mammals (Álvarez-González et al., 2023), as well as anthropogenic substances such as plastic pollution (Torsvik et al., 2019) and oil spills (Adade et al., 2021).

Coastal habitat mapping typically employs three main categories of UAVs: rotor drones, fixed-wing drones, and vertical take-off and landing (VTOL) drones (Figure 2, Kvile et al., submitted). The VTOL drones integrate features from both fixed-wing and rotor systems, utilizing rotors for vertical take-off and landing of a fixed-wing drone. The type of drone to deploy depends on the objectives of the survey. Rotor drones excel in maneuverability, low-speed flight, and hovering capabilities. In contrast, fixed-wing drones are more energy efficient and thus facilitate extended flight durations, making them suitable for long-distance missions. However, their reliance on specified take-off and landing zones often poses challenges in coastal environments, a challenge subdued with VTOL drones for easier landings. See Kvile et al. (submitted) for an evaluation of these in coastal mapping applications.

Drone-based recognition of coastal habitats builds on optical identification of species and habitats, also referred to as an optical fingerprint. It describes a distinctive spectral signature of the target object (e.g. seagrass, macroalgae, etc.) and quantifies how the object interacts with light across different wavelengths of the electromagnetic spectrum, typically within the RGB bands of visual light and at slightly longer wavelengths (400 to 1000 nm). In habitat mapping, optical sensors like RGB, multispectral (MSI) and hyperspectral (HSI) imagery are applied enabling precise identification and differentiation of objects or elements (Figure 2), such as vegetation or animals by comparing the collected spectral data to established databases or by training of AI algorithms and recognition networks. This approach facilitates accurate classification and mapping in addition to uncertainty estimation of both quantitative (uncertainty in coverage and distribution) and qualitative (probability of prediction) parameters.

Figure 2. Novel drone and sensor types used for coastal habitat mapping: A) fixed-wing drone (eBee X mapping drone, AgEagle Aerial Systems); B) rotor drone (DJI Matrice 300, photo by SeaBee); C) vertical take-off and landing (VTOL) drone (DeltaQuad Pro, DeltaQuad, photo by SeaBee); D) Red, green and blue (RGB) sensor (Sony RXR II, Sony Group Corporation); E) multispectral imagery (MSI) sensor (MicaSense Altum, Ag Eagle Inc.); F) hyperspectral imagery (HSI) sensor (Specim AFX10, Specim, Spectral Imaging Ltd.). Adapted from Kvile et al. submitted.

Seagrass and kelp forests have been difficult to map with high levels of accuracy and precision with satellite imagery due to complex and mixed optical signatures from vegetative and abiotic spectra and near-coast optical distortion (e.g. Cavanaugh et al., 2021). The use of drones, however, enables mapping of such ecosystems with higher spatial resolution and more precise georeferencing, differentiating between biotic and abiotic matter using higher spectral resolution, and with minimal optical distortion. Figure 3 illustrates an example of seagrass mapping in a shallow bay area in the Oslofjord, Norway. Using a fixed wing drone and combined data from RGB and MSI cameras (as A, D, and E in Figure 2, respectively), around 500 single images were collected and subsequently stitched together to form a single orthomosaic which was georeferenced to a precision of +/- 3 to 5 cm (i.e. the precision of the orthomosaic relative to the real world). The orthomosaic holds a ground sampling distance (image resolution) of 3 cm/pixel. Major coastal components such as seagrasses, various macroalgae species, sandy sediments, and rocks were annotated from *in situ* (ground truth) observations, and subsequently used to train a machine learning (ML) algorithm (in this case random forest) for separating and predicting the habitat classes in the orthomosaic (see legend in Figure 3). The predictive accuracy of the classification (a so-called confusion matrix) showed that eelgrass (*Zostera marina*) was predicted with 86%, brown seaweed with 87%, sandy sediment with 79% accuracy, and so forth (Figure 3). More recently, Convolutional Neural Network (CNN) analysis has improved model performance (Liu et al., 2022) and has enabled habitats predictions at species level (Kvile et al., submitted). Ultimately, ML models may allow mapping and classification of known coastal habitats without the need for additional training of the ML/AI models, building on existing networks. However, ground-truthing will still be essential when training new algorithms for unknow species and habitats.

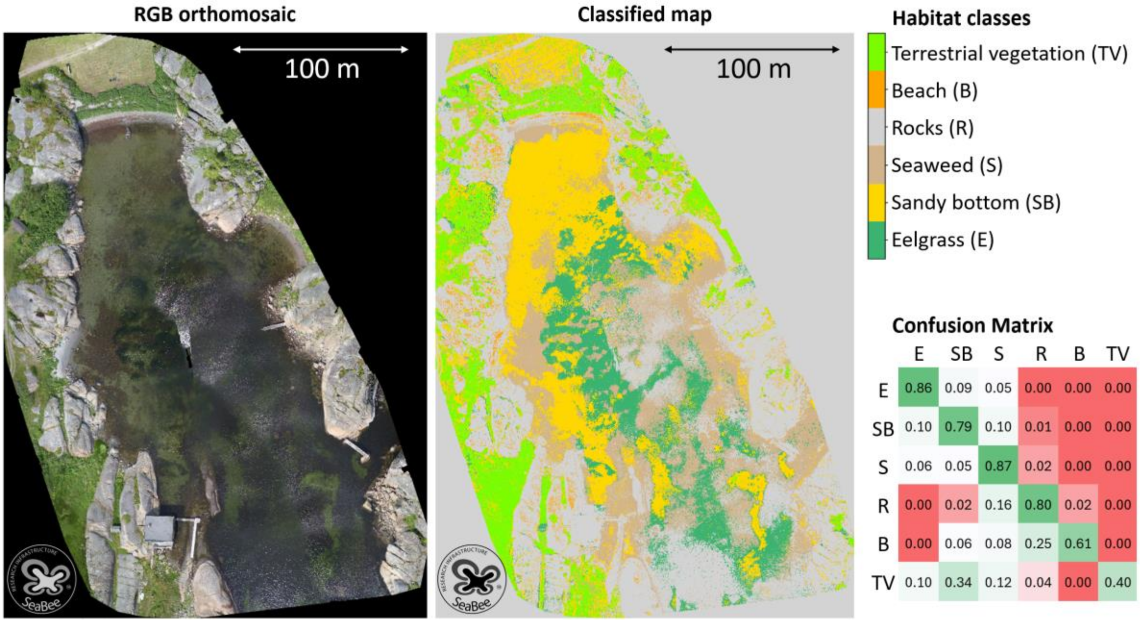

Figure 3. A drone-based map (orthomosaic) and the habitat classes it hosts in a typical coastal bay of the Oslofjord, Norway. The orthomosaic is stitched from ~500 single RGB images collected by a flying drone, and the corresponding habitat map is a result of an ML-based prediction algorithm. The predictive success is shown as the inserted confusion matrix, giving 86% prediction success for seagrass. Data by Hancke et al., unpubl., collected as described by Kvile et al., submitted.

Further advancement within drone technology for shallow water benthic habitat mapping includes higher resolution RGB sensors and multiband MSI and HSI data. It also involves the development of neural networks from foundation models and faster and larger drones with increased payload capabilities. Also, infrared (thermal) sensors will improve the recognition of animals and facilitate for studies of surface temperature variabilities of water and vegetation, with applications for land and ocean warming studies . We anticipate that LIDAR (Light Detection And Ranging) systems, now available for medium sized (<25 kg) drones will come to add a new range of applications for shallow water bathymetry and seafloor mapping, sedimentation and erosion assessments (Chust et al., 2010), and likely for coastal health and ecosystem services assessments too.

Thus, while drone imagery cannot replace traditional methods for coastal mapping completely, it offers a cost-efficient supplement, and, in concert with other new technologies like unoccupied surface and underwater vehicles, much needed opportunities for digitalized and reproducible ecosystem monitoring and research. In light of these advancements, drone-based methodologies may be recognized as recommended and protocol procedures for ecosystem mapping and monitoring under directives like the Maritime Spatial Planning Directive, the MSFD, and for biodiversity assessments in shallow-water coastal regions. This recognition could contribute to enhancing future coastal management and regulations by providing accurate, comprehensive and up-to-date information for decision-makers.

Use of imaging and artificial intelligence (AI) for monitoring pelagic communities

Within pelagic communities, plankton are plants and animals which are critical to all life on earth. Whilst phytoplankton play a key role in carbon flux as primary producers, absorbing more carbon dioxide than all trees on land, zooplankton occupy a central position in the food web, often controlling phytoplankton by grazing and providing food for many important larval and adult fish and seabirds (Pitois et al., 2012; Lauria et al., 2013). Decades of laboratory and field investigations have shown major impacts of changing oceans on zooplankton physiology, community composition, and distribution, and their resulting influence on both biogeochemistry and productivity of the oceans (Ratnarajah et al., 2023). It is therefore critical to further our understanding of how plankton community structures and abundances are likely to respond to climate change in the future, and their effects on ecosystem dynamics.

The implementation of the MSFD has resulted in increased volumes of plankton data needed to meet its requirements. However, the traditional collection of plankton samples, using bottles and nets followed by taxonomic analysis of the preserved samples using microscopes by a trained specialist, is a labor intensive and time-consuming process (Wiebe and Benfield, 2003; Benfield et al., 2007). Limited resources and budgets for monitoring have been a major driver for the development of cost-effective methods to gather plankton information (Danovaro et al., 2016). Nowadays there is a wide range of instruments including for example molecular, optical, remote sensing, and automated techniques (Pitois et al., 2023), most have been reviewed by Lombard et al. (2019).

Imaging instruments combined with ML tools to automatically classify the collected images (e.g., Weldrick, 2022) have received a high level of interest, due to their ability to provide rapid and unbiased data that can be stored digitally. Thus, they offer the opportunity to help overcome many of the limitations that characterize traditional methods of collecting and analyzing plankton samples. These can be used on collected samples (e.g. PlanktoScope (Mériguet et al., 2022; Pollina et al., 2022), FlowCam (Mériguet et al., 2022)), or *in situ* (e.g., Underwater Vision Profiler (Drago et al., 2022), Plankton Imager (Scott et al., 2023), In Situ Ichthyoplankton Imaging System (ISIIS) (Panaïotis et al., 2022), Scripps Plankton Camera (SPC) (Le et al., 2022), Imaging FlowCytobot (Kraft et al., 2022); Video Plankton Recorder (VPR) (Plonus et al., 2021).

A specific example of an all-in-one tool that combines zooplankton sampling and image analysis is the Plankton Imager (PI) (Figure 4).

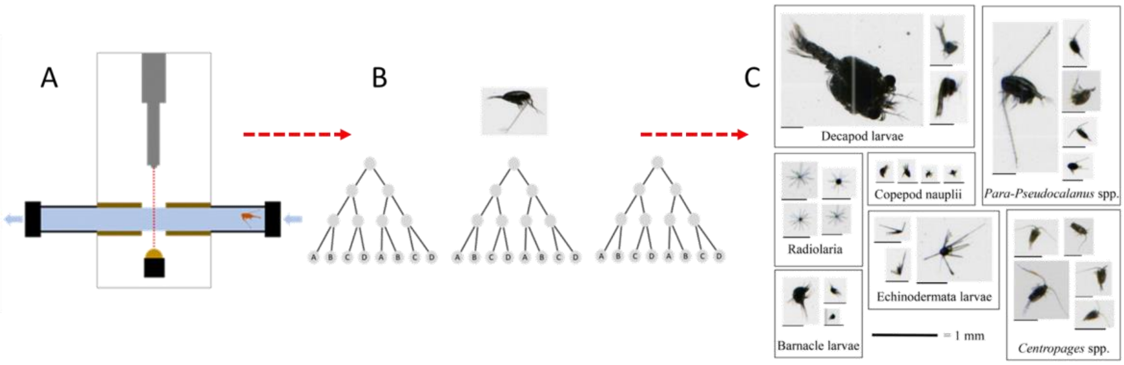

Figure 4. The Plankton Imager (PI) set-up onboard RV Cefas Endeavour. (A) The high-speed line scan camera images all particles (0.18-25 mm) continuously in a through-flow system connected to the ship water supply and can work in all weathers; (B) a machine algorithm (Data Study Group team, 2022) runs to automatically classify the (C) images are saved along with GPS, time and size information.

The PI was shown to provide a robust description of zooplankton abundance, distributions and community structures (Pitois et al., 2018; Scott et al., 2021). Its ability to capture *in situ* size information was applied to the development of ecological indicators (Pitois et al., 2021). Finally, the PI collects over 150 images per second, thus allowing for fine spatial resolution of the collected images (Scott et al., 2023).

The ability of the PI to continuously collect fine resolution data opens the door to data intensive plankton studies. But the pinnacle of pelagic studies will be the monitoring of plankton (phytoplankton and zooplankton) along environmental (e.g., temperature, salinity) and other biological (e.g., fish) variables, at high-frequency and in real-time. This offers the opportunity to quickly respond according to observed changes and adjust data collection strategy (during a survey for example), but also increases capabilities and potential towards building a digital twin of the ocean (Chen et al., 2023).

However, the concept of digital twin of the ocean, which seeks to integrate marine big data and AI, is in the initial stage of development and major data challenges need addressing first:

(i) An ever-increasing number of images is being produced. These need to be stored and made accessible. But the biggest bottleneck is in the processing and interpretation step. ML tools are key to automate this process and are reliant on comprehensive libraries of pre-classified images (training set). Whilst building training sets is a time-consuming exercise, these are key to developing high performing algorithms trusted by taxonomists in particular (Giering et al., 2022).

(ii) Another major bottleneck resides in the transfer of high numbers of relatively large data files (images) from offshore to land locations as soon as images are collected faster than they can be transferred.

(iii) Data is collected under different formats and at varying scales: for example, continuous temperature values, plankton images, acoustic signals, satellite imagery, etc. Integrating multimodal data is a challenge (Grossmann et al., 2022).

Addressing these challenges will require (i) substantial investments to build high quality and comprehensive training libraries, and (ii) a data driven approach to automate end-to-end data flows. The latter, which has only recently become possible thanks to the latest advances in ML tools, in par with high-performance computing (e.g. Kraft et al., 2022), whilst the use of Edge-AI allows for image processing to be done closed to the point of collection so that only the data output from the images is transferred to the cloud rather than the image itself (Schmid et al., 2023).

The use of imaging and AI tools is still in its infancy, but early results suggest that technological advances in this field have the potential to revolutionize how we monitor our seas.

Climate modelling as a decision support tool for marine spatial planning and biodiversity conservation

The pressures on marine resources are increasing to the extent that societal demand for goods and services often exceeds the capacity of the marine environment to meet them all simultaneously (Ehler, 2021). Without regulation, these demands can lead to over-use, conflicts and eventual degradation of marine ecosystems. To ensure the sustainable use and conservation of the world ocean, a public process must be undertaken to decide what, where and when human activities should take place in marine areas – e.g. marine spatial planning (MSP) (Frazão Santos et al., 2020). MSP has the potential to balance multiple, often conflicting, human demands and to protect marine ecosystems in a spatially explicit manner (Frazão Santos et al., 2020). As a management process, MSP has spread widely over the last 15 years and marine spatial plans are in development in around 70 countries, encompassing six continents and four ocean basins (Frazão Santos et al., 2021). However, despite this widespread acceptance, development and implementation of MSP still faces some challenges, not least of all global climate change. Anthropogenic climate change is altering marine ecosystems at unprecedented rates. Altered environmental conditions (e.g. warming, acidification, deoxygenation, changes in salinity and hydrodynamics, among others; Pörtner et al., 2021, 2022) and habitat suitability cause marine species to respond by tracking their optimum habitat (when possible), resulting in changing distributions (Dulvy et al., 2008). This is expressed locally as changes (and often loss) of species biodiversity (Pecl et al., 2017), impacting on conservation efforts as well as on sectors such as capture fisheries and aquaculture. Such changes may increase the potential for conflicts between users of marine space, and increase cumulative impacts on the marine environment, all of which will have implications for human well-being and prosperity (Frazão Santos et al., 2020). Designing MSP that addresses the effects of climate change (climate-smart MSP) is therefore a pressing ambition for ocean managers and marine planners. However, it remains - with very few exceptions – an ambition without practical implementation (Queirós et al., 2021).

The sensitivity of marine habitats and species to climate change can be heterogenous in time (Hawkins and Sutton, 2012) and space, and this fact can be utilised by planners and policymakers when considering possible uses of marine space into the future. . Designing effective climate-smart MSP therefore requires an objective means to assess how ecosystem components may respond to climate change. Ocean climate modelling (e.g. physical-biogeochemical modelling and species distribution modelling) has been identified as an essential decision support tool for planners seeking climate-smart design (Queirós et al., 2023). The benefits of such models are clear – plans can be tested under different climate change scenarios, and assessments of possible trade-offs can be made; these in turn can inform those stakeholder consultations, which are so critical to planning (Pınarbaşı et al., 2017; Frazão-Santos et al., 2020). To date, however, the uptake of climate modelling projections has been slow. This is largely due to the mismatch between traditional climate modelling analyses (which tend to focus on long-term climate impacts and individual ecosystem components separately) and the time-scales and interests of planners, who see climate change as a holistic process that may affect the activity of sectors reliant on those species and services impacted by changing ocean conditions (Queirós et al., 2021). To address this mismatch, the “bright-spots framework” was developed.

Climate change drives concurrent changes in multiple ecosystem attributes, which change spatially and temporally at different speeds and with different magnitudes (Pörtner et al., 2022). The bright-spots framework uses techniques established in the meta-analysis statistics field to identify the emergence of ecosystem-level climate change signals (cf. pressure by pressure as in traditional analyses), which allows for a mapping of the emergence of: (i) climate change refugia - i.e. where environmental conditions or species abundances may be resilient to climate change, and where current uses may be sustainable; (ii) climate change hotspots – i.e. areas where climate driven trends force ecosystem components or species abundances into a new state beyond their natural variability; or (iii) climate change bright-spots, indicating a change in environmental conditions or species abundances (outside of present day variability) in opposition to predicted climate change trends, generally as a result of climate cycles (Queirós et al., 2021). Capitalising on refugia and bright spots in climate-smart planning design may be beneficial in the mid-term delivery of sustainability goals, and for increasing the climate resilience of important conservation sites and maritime sectors. It may also function as a time buying strategy while we reduce global emissions and slow the pace of climate change. This method has already been used in several globally distributed research programmes including those informing marine management in Ireland (Queirós et al., 2021), The UK (Queiros et al., 2023) Philippines (Talbot et al., 2024) and Vietnam (Queirós et al., 2022). It is also in use in a number of case studies across the European Union ().

To further illustrate, a case study using the bright spots framework was recently conducted to inform the development of proposed new marine protected areas in Irish waters (Queiros et al., 2024), in support of the move towards the Global Biodiversity Frameworks 30x30 target. That study identified areas in the Irish marine planning area which consistently (>80% of the time) appeared as climate refugia between 2026 and 2069 across two emissions scenarios (RCP4.5 and RCP8.5), for habitats and species of conservation interest. Specifically, four separate analyses were conducted: two used modelling datasets that described the spatial distribution of climate resilience and vulnerability in the environmental conditions and prey species required by benthic and pelagic megafauna. A further two analyses used datasets that described benthic and pelagic habitats directly (Queiros et al., 2024). Figure 5 shows the locations of those refugia, along with the current Natura 2000 network of marine protected areas (MPAs) in Irish waters, and the proposed locations of new ones. While identified long-term refugia for pelagic habitats and megafauna were rather sparse, a substantial proportion of the proposed new MPA sites fell in refugia for benthic habitats, particularly those in deeper offshore waters (Queiros et al., 2024). The identification of such consistent, long-term refugia can provide marine planners and managers with a better idea of those areas which may support effective conservation into the future, despite climate change pressures in the broader region.

Some novel methods to assess the status of the ocean

Cumulative Impacts Assessment to guide Ecosystem-based Management

The Ecosystem Approach (Kirkfeldt, 2019) offers a powerful strategy for the integrated management of living resources and was adopted as the primary framework for the conservation and sustainable use of biodiversity under several EU policy frameworks (e.g. the MSFD). The many definitions, though broadly similar, vary in their emphasis but could be structured into three broad themes representing the main aspects of Ecosystem-Based Management (EBM) using a review of Long et al. (2015), providing a set of key principles: (i) capturing the integrity, functioning and dynamics of the marine ecosystem in its environmental context; (ii) accounting for all relevant human activities and their interconnections with the marine ecosystem as part of the wider socio-economic context; and (iii) organising the process regarding governance and management, taking account of the institutional context.

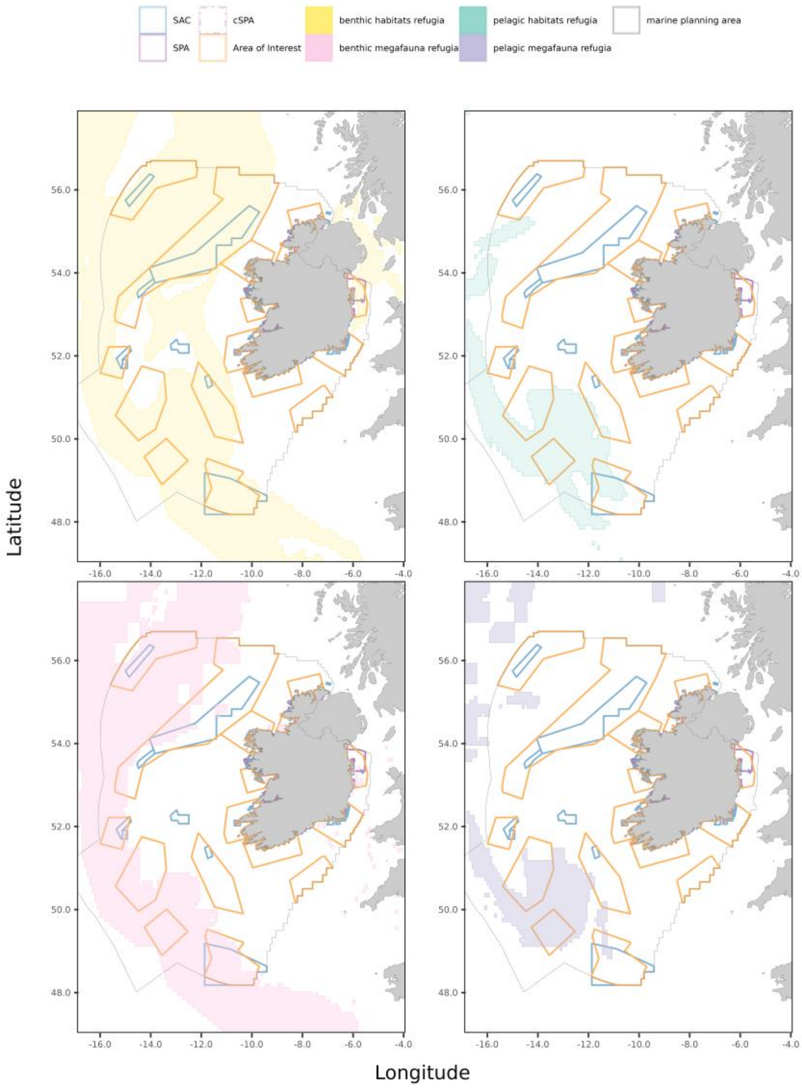

Figure 5. Areas in which long-term climate change refugia were identified for benthic habitats, pelagic habitats, and megafauna reliant on either benthic or pelagic species and habitats. Refugia appeared consistently across both emissions scenarios (RCP4.5 and RCP8.5), between 2026 and 2069. Proposed Marine Protected Areas locations (Areas of Interest) were identified as being important conservation sites by the FairSeas coalition of Irelands leading environmental NGOs and environmental networks. Figure reworked with permission.

The first requires the monitoring and assessment of the multitude of indicators put forward in the MSFD and other frameworks intended to capture the many different aspects and interactions encapsulated in the ecosystem integrity, functioning and dynamics biodiversity concepts (Teixeira et al., 2016). The second follows the rationale that EBM is about managing human activities, not the ecosystem itself (Long, 2012; O'Hagan, 2020). An assessment of the potential impacts of all relevant human activities and their pressures is therefore key to guide management towards the achievement of the policy objectives for the marine ecosystem (Borja et al., 2024).

A study by Piet et al. (2023) introduced SCAIRM (Spatial Cumulative Assessment of Impact Risk for Management), a spatially explicit cumulative impact assessment (CIA) method that has been specifically developed to guide an ecosystem approach. CIAs are considered key to policymaking and planning in governance and management but their actual implementation in MSP and management processes is often limited (Stelzenmüller et al., 2018; EC-CINEA, 2021). Management is required to reduce the effects of human activities on ecosystem components and their functioning to achieve policy objectives in support of societal and environmental goals (Cormier et al., 2017). However, it is the implementation of management measures developed by authorities that should 'carry into effect' the objectives identified at the onset of the process required as part of an ecosystem approach (Elliott et al., 2020).

Risk-based approaches have often been at the basis of CIAs for marine management (Stelzenmüller et al., 2018). SCAIRM combines elements from both a likelihood-consequence approach (i.e., calamities; Williams et al., 2011), and an exposure-effect approach, which is considered more suitable when assessing existing and (more or less) continuous or frequently occurring pressures (Smith et al., 2007; Knights et al., 2015) into a method able to handle pressure events as well as continuous pressures. Exposure and Effect are estimated separately using risk-based approaches based on (i) qualitative categories and expert-judgement scores, or (ii) quantitative information applying actual data. The former has the advantage that it allows comprehensive assessments that covers all potential impact chains (i.e. a causal chain connecting every single human activity-pressure-ecosystem component) and can (and needs to) be applied in data-poor areas (Knights et al., 2015; Borgwardt et al., 2019). The latter has the advantage of higher accuracy and more confidence in the outcome; however, is limited to only those impact chains for which adequate information is available and is therefore more appropriate in data-rich areas, such as the North Sea (Piet et al., 2021a). The SCAIRM method is essentially a harmonization of this work, allowing to mix qualitative and quantitative information.

An ecosystem approach requires policy objectives to be achieved through what are essentially sectoral operational controls, which are not designed to achieve ecosystem scale outcomes (Cormier et al., 2019). While sectors may individually fully conform to sector-specific operational standards and regulatory requirements, integration across sectors is lacking (Garcia and Charles, 2014). Thus, instead of a lack of legislation, regulations or standards, the problem may be caused by a lack of coherence and alignment between sector-specific operational practices and environmental legislation and policy objectives (Cavallo, 2018). Some studies show how an advanced CIA method like SCAIRM can be used to provide the type of operational-centric approach (Murawski, 2007; Cormier et al., 2017, 2019) required to achieve environmental policy objectives.

Impact Risk is the key output of SCAIRM and can be estimated per impact chain as Exposure multiplied by the Effect Potential. Impact Risk reflects the expected pressure-induced loss of a specific ecosystem component relative to an undisturbed situation based on state-of-the-art (qualitative or quantitative) information. To guide (an ecosystem approach to) MSP or management first a recent baseline situation needs to be established which can be evaluated against any future scenarios. Aggregation of the impact risk from cumulative pressures across ecosystem components then reflects the degree to which biodiversity or ecosystem integrity is at risk from anthropogenic activities.

Here, we provide a North Sea example of such scenarios for 2030 and 2040 (Table 1). The main driver for MSP in the North Sea is the transition towards renewable energy, notably offshore wind (van de Pol et al., 2023). For an ecosystem approach to MSP, the implementation of offshore windfarms needs to be considered as part of scenarios involving all the main human activities.

Table 1. Scenario values (% of study area) for the main human activities in the North Sea study area (Piet et al., 2021a).

Activity	**Phase**	**Baseline (2022)**	**Scenario for 2030**	**Scenario for 2040**
Aquaculture: fish	Operation	0.00356	0.00625	0.10861
Aquaculture: fish	Set-up	0.00036	0.00027	0.00512
Aquaculture: macroalgae	Operation	0.00105	0.00184	0.03194
Aquaculture: macroalgae	Set-up	0.00105	0.00184	0.03194
Aquaculture: shellfish	Operation	0.00545	0.00956	0.16611
Aquaculture: shellfish	Set-up	0.00054	0.00041	0.00783
Fishing: benthic trawling	Mooring/anchoring	0.89099	0.83975	0.78081
Fishing: benthic trawling	Operation	89.09860	83.97510	78.08124
Fishing: Nets	Operation	6.86605	6.47122	6.01704
Fishing: Nets	Set-up	6.86605	6.47122	6.01704
Fishing: Pelagic trawls	Mooring/anchoring	0.17532	0.16524	0.15364
Fishing: Pelagic trawls	Operation	17.53183	16.52368	15.36396
Oil and Gas	Construction	0.00538	0.00538	0.00000
Oil and Gas	Operation	0.10760	0.10760	0.00230
Sand/gravel mining	Operation	2.55918	3.37578	4.09468
Sand/gravel mining	Disposal	2.55918	3.37578	4.09468
Shipping	Mooring/anchoring	0.07972	0.07972	0.11081

Shipping	Operation	20.86627	20.86627	29.00411
Telecoms and Electricity	Operation	0.04168	0.05602	0.07762
Telecoms and Electricity	Laying cables	0.00104	0.00143	0.00108
Wind farms	Construction	0.06144	0.17251	0.27723
Wind farms	Operation	0.73154	2.11163	4.88391

SCAIRM was then applied to assess the consequences of these scenarios in terms of (changes in) Impact Risk (Figure 6).

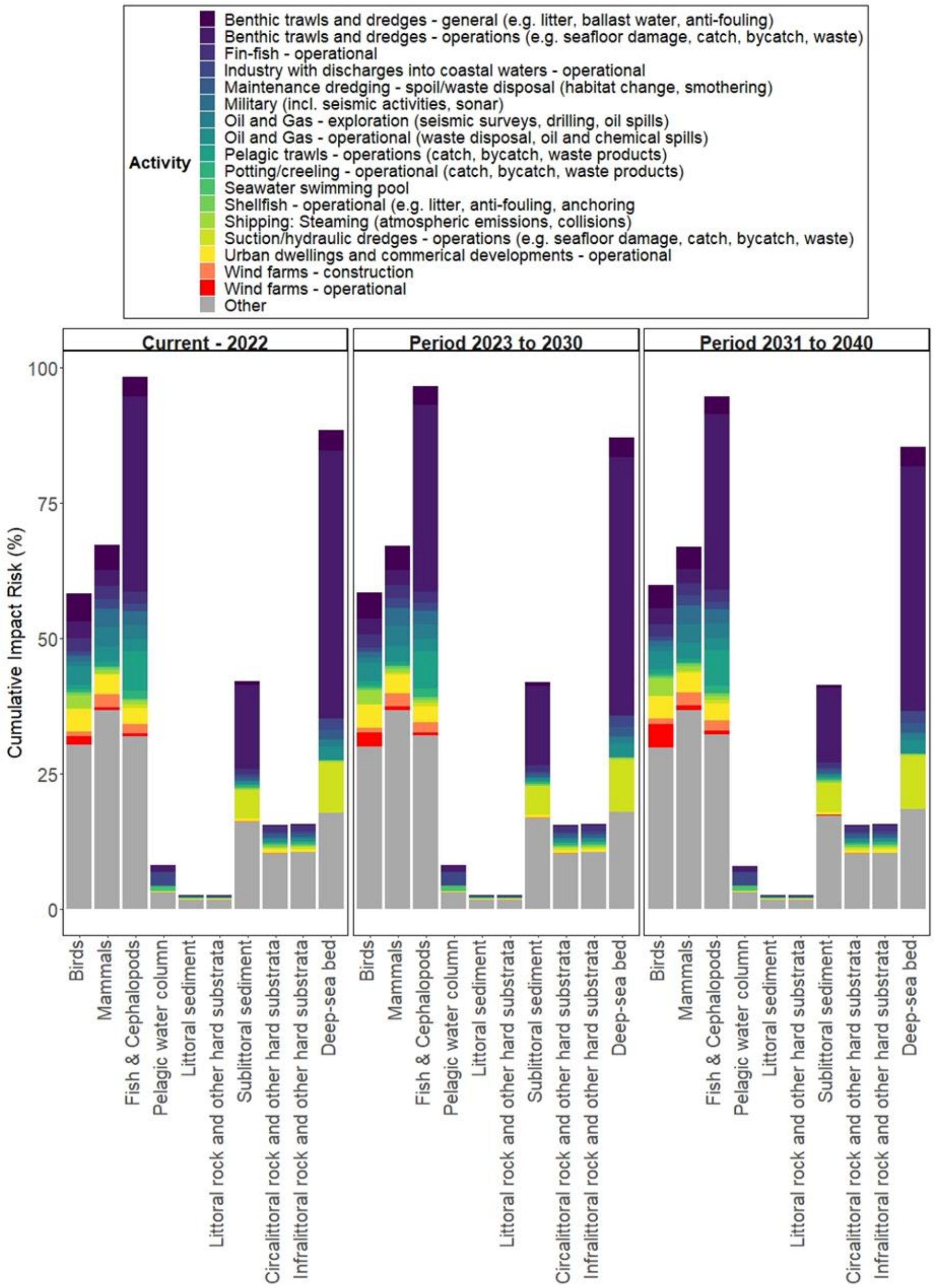

Figure 6. Baseline situation and two future scenarios (2030 and 2040) from a North Sea application of SCAIRM indicating the main ecosystem components at risk with the 16 main stressors causing this risk. The remainder are grouped under “other” for presentation reasons but can be specified if needed.

Table 2 shows which ecosystem components are likely to suffer from an increased Impact Risk under these scenarios.

Table 2. Change in cumulative impact risk (%) of human activities on ecological components in two future scenarios (2030, 2040) relative to the baseline (2022). An increase is shown in bold.

Ecological component	Change in impact risk (%) 2030 vs. 2022	Change in impact risk (%) 2040 vs. 2022
Birds	**0.4%**	**2.9%**
Mammals	-0.3%	-0.4%
Fish & Cephalopods	-1.7%	-3.7%
Pelagic water column	-0.8%	-1.7%
Littoral sediment	0.0%	**0.2%**
Littoral rock and other hard substrata	0.0%	**0.2%**
Sublittoral sediment	-0.6%	-1.6%
Circalittoral rock and other hard substrata	-0.2%	0.0%
Infralittoral rock and other hard substrata	-0.2%	0.0%
Deep-sea bed	-1.6%	-3.6%

This assessment thus shows that notably for birds and to a lesser extent for seabed habitats in the littoral zone, additional management measures are required in order not to (further) compromise the achievement of the MSFD objectives.

Assessing multiple pressures and cumulative impacts for Europe's seas

The European Union's policy objective to achieve GES of marine environment under the MSFD, cannot be successful without understanding which anthropogenic pressures are preventing GES and where that takes place. This becomes more complex when multiple simultaneous pressures act together exerting cumulative impacts. Tools to identify, map and assess the cumulative impacts of multiple pressures have been developed for several Europe's marine areas (Korpinen and Andersen, 2016) and CIAs have been published since 2010s (Korpinen et al., 2012, 2021; Coll et al., 2012; Andersen et al., 2013; Micheli et al., 2013).

A key challenge in CIAs is the integration of tens or hundreds of ecosystem impacts into a common scale (Halpern and Fujita, 2013). In simpler models, such as fishery impacts on marine populations (Coll et al., 2016) or impacts of multiple pressures on a single population (Marcotte et al., 2015), the integration can be made with data-based pressure-population response functions, but the integration of tens of different pressures on tens of very different ecosystem features is beyond the possibilities of ecosystem models. The framework to address the complex CIAs was introduced by Halpern et al. (2008b). It represents the pressure-ecosystem response functions as simple sensitivity estimates. These are scores representing the sensitivity of an ecosystem feature to a single pressure and they are surveyed from a wide pool of experts (Halpern et al., 2007).

While the use of expert judgement is not ideal, the approach has served tens of peer-reviewed CIAs worldwide (Korpinen and Andersen, 2016) and many other ecosystem studies (e.g. Isbell et al. 2022). So far, only a few studies have been made to assess how well the experts agree with each other in environmental studies. It is also unclear if such sensitivity estimates should be local or regional or whether they can be global (Halpern et al., 2007). Korpinen et al. (2021) carried out a pan-European survey of marine sensitivity estimates and concluded that the expert elicitation was relatively reliable: 394 out of 450 combinations of pressure-ecosystem estimates did not differ from each other by the experts. The remaining sensitivity estimates represented cases where there were too few responses, the knowledge of the sensitivity was arguably weak, or for reasons not known for the authors. In that study, the CIA was made for the whole European Sea area using the same sensitivity estimates. To our knowledge, more comprehensive analyses of the coherence of the sensitivity estimates have not been made.

The pan-European CIA by Korpinen et al. (2021) was carried out to support the European Environment Agency (EEA) in locating the areas of too high cumulative impacts (which can potentially prevent achieving GES) and identifying the human activities and pressures causing them (hereafter the 'EEA study'). The objective of this study was to map, on a European scale, the main anthropogenic pressures and estimate their level of intensity in each assessment grid cell, which were 10 km x 10 km in size. From the management point of view, it is important to use standardized definitions of 'pressures' and 'human activities' as given by the respective policy, in the case of the EEA study the MSFD definitions were used (European Commission, 2017). Therefore, the EEA study categorized the pressures into 14 classes, following the MSFD Annex III (European Commission, 2017). The CIAs certainly have different objectives which determine how detailed data – human activities, pressures or ecosystem features – are necessary to use. In the EEA's CIA, the objective was not to enable managing detailed activities but to get an overall picture of the 'problem areas' and identify the problems. For that objective the 14 pressures were a sufficient level of detail.

Visual and analytical assessments of integrated pressures is sometimes enough for management, but more accurate results may be produced by estimating their cumulative impacts, i.e., producing a full CIA. The EEA study estimated the cumulative impacts of 14 pressures on 31 ecosystem features, which were both habitats and species groups. These needed 14 x 31 estimates for ecosystem sensitivity. Thus, a practical limiting factor for a large-scale CIA is also the determination of the sensitivity estimates. There is no guideline on how widely and deeply marine biodiversity should be included in CIAs. In the EEA study it was, however, ensured that the ecosystem features should cover benthic and pelagic habitats and the main mobile species groups (mammals, birds and fish). A CIA in the Baltic Sea compared the influence of increasing or decreasing the number of ecosystem features and concluded that the higher number of features adds fine-scale details in the spatial presentation of the results but the main conclusions of the CIA – on the marine region scale – were not jeopardized (Korpinen et al., 2012).

The EEA study was a necessary landmark in the EU ambition to map and identify the preventing factors to achieve GES in Europe. It showed that there are multiple pressures and cumulative impacts in Europe's seas that probably prevent GES and that the main pressures in the shelf areas are physical disturbance by bottom-contacting fishing gears, invasive alien species and species disturbance by human presence, and in the whole sea area, increased surface water temperature and underwater noise. The coastal and shelf areas had significantly higher cumulative impacts than the offshore areas. Do these pressures cause such high impact that GES is not achieved? There are a few studies which have compared the CIA results with environmental monitoring data. Bevilacqua et al. (2018) showed how the condition of coralligenous algae assemblages depended on the level of a CIA score, and Clark et al. (2016) showed correlation with benthic communities in an estuary. The EEA's Marine Messages II report successfully correlated the CIA with an integrated assessment of marine ecosystem status on the pan-European scale (Reker et al., 2019), and Korpinen et al. (2021) showed how the CIA explained the ecological status of coastal waters.

Even if the large-scale CIAs such as the EEA study are only rough estimations of the cumulative impacts, they are useful tools to guide marine management into the right direction. The work is far from building a model to predict good or not good status of marine ecosystem, which is still best to assess with biological monitoring, but it is unarguable that the spatial and temporal resolution of CIAs is much sharper than in biological monitoring. The CIAs can indicate areas of high risks of failing policy objectives such as GES. Such an approach is already adopted with the MSFD assessment of the benthic habitats where pressure impacts on seabed habitats are seen as the major data source informing of the need for management measures (European Commission, 2022b).

Assessing the status of marine systems using NEAT

Among the tools and methods for assessing the status of the sea, in an integrated way, one is NEAT (Borja et al., 2016). This method was developed for assessing the marine environmental status under the MSFD and was first validated in 10 locations in the four European regional seas (in the Black Sea, Mediterranean Sea, Atlantic Ocean, Baltic Sea) and the Arctic (Uusitalo et al., 2016). Later, it has been applied in different locations and considering different pressures, e.g. (i) in the Iranian Caspian Sea, to assess the status of bathing waters (Nemati et al., 2017); (ii) in the Mediterranean, for assessing the spatio-temporal recovery of a Greek bay (Pavlidou et al., 2019); to compare NEAT and Maltese official MSFD assessments (Borja et al., 2021); or to identify the contribution of MPAs to the status of this sea (Fraschetti et al., 2022); (iii) in the Atlantic, to assess the status of MPAs in deep-sea (Kazanidis et al., 2020); and testing the use of multi-source and multi-scale data (Menchaca et al., 2022); (iv) in the Black Sea, to assess the status of Romanian (Marin et al., 2020) and Turkish waters (Tan et al., 2023); and (v) in the whole Europe, to test some MSFD descriptors (Borja et al., 2019).

As shown in all the above-mentioned studies, NEAT (version 1.4) is a software designed to assess the environmental status of marine waters, which can be freely downloaded from https://www.azti.es/en/productos/neat-software-nested-environmental-status-assessment-tool/. The central principle of NEAT is its hierarchical and nested structure of the environmental status of a given area, or Spatial Assessment Unit (SAU), and the associated habitats. This means that information belonging to a SAU contributes to the specific assessment of this SAU, as well as to the assessment of any other larger SAU that encompasses it, and therefore, to the overall assessment. The order of these hierarchies is such that the assessment begins with the hierarchically nested SAUs. In the example below (Figure 7), information belonging to the "Sardinia" will contribute to the specific assessment of the "Sardinia", as well as that of the "Western Mediterranean" and the "Mediterranean Sea".

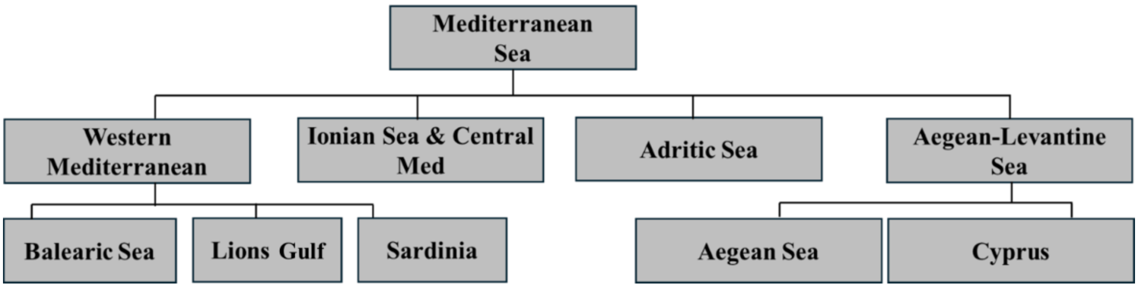

Figure 7. Hypothetical example of Spatial Assessment Units (SAUs) for the Mediterranean Sea, that are nested and hierarchically structured. The SAUs on the higher level always comprise all the units below in the hierarchy.

This tool aggregates the values of a set of indicators that are specific to a SAU, habitat, ecosystem component and MSFD descriptors in a comparable and systematic way (Uusitalo et al., 2016; Borja et al., 2019, 2021). Each indicator must have at least: (i) a range (i.e., a minimum and a maximum value), in which the values can be found, (ii) the class boundary values between the different quality classes (e.g. the boundary between good and non-good status), (iii) the actual measured indicator value, and (iv) its standard error value (Borja et al., 2016, 2021). After selecting the indicators to be used, the software normalizes them to a uniform scale ranging from 0 (worst environmental status) to 1 (best environmental status) with equidistant classes, allowing for integration across different ecosystem components and indicator types. (Borja et al., 2016, 2019, 2021).

The resulting NEAT value can be calculated at different levels, e.g. it can contain all indicators belonging to a certain set of descriptors. The NEAT value is interpreted as the environmental status for this specific descriptor(s). Another result can be the NEAT value of a specific ecosystem component, habitat or SAU (Figure 8). The final NEAT values are calculated as a weighted mean of all indicator values assigned to a certain defined SAU, habitat or combination of both, and the value can be visualized as SAU + habitat (+descriptor) or SAU + ecosystem component (+descriptor) (Figure 8).

The initial version of the software only matched the original MSFD criteria within the 11 descriptors, and Borja et al. (2021) implemented a version adapted to the ecosystem components of the Directive (e.g., pelagic and demersal commercial fish species, type of nutrients or contaminants). These authors redefined the hierarchies following the hierarchical structure for habitats and considering relevant species groups, habitat types, ecosystems and pressures, as well as criteria, according to the updated European Commission (2017) Decision. In this sense, it can be confirmed that NEAT is an adequate and efficient tool to aggregate multiple indicators from different ecosystem components (and multiple sources) to assess (spatially and temporally) the health status of marine waters. NEAT was found to be useful for managers, policy-makers and scientists in taking decisions for future management measures and assessment approaches.

Spatial Assessment Units	Area	SAU weight	NEAT value	Confid.	Ecosystem component & Descriptor						Habitat & Descriptor	
	(km²)			(%)	WC	Phyto.	Fish	Moll.	Crust.	Com Fish	Pelagic	Dem/Ben
					D8	D5	D3	D3	D3	D3	D3, D5, D8	D3
Mediterranean Sea	2,520,033	0.080	0.513	100.0	0.893	0.944	0.513	0.466	0.315	0.495	0.561	0.383
Western Mediterranean	844,630	0.000	0.435	100.0	0.639	0.956	0.413	0.420	0.382	0.410	0.486	0.383
Balearic Sea	468,191	0.015	0.442	100.0	0.530	0.949	0.431	0.374	0.382	0.421	0.508	0.376
Lions Gulf	62,702	0.002	0.438	100.0		0.967	0.418	0.393	0.393	0.413	0.500	0.376
Sardinia	313,737	0.010	0.423	100.0	0.698	0.964	0.382	0.454	0.382	0.392	0.451	0.394
Ionian Sea & Central Med	846,345	0.026	0.414	100.0	0.909	0.921	0.393	0.447	0.380	0.397	0.444	0.383
Adriatic Sea	98,666	0.002	0.558	100.0	0.786	0.976	0.447	0.466	0.473	0.454	0.638	0.417
Aegean-Levantine Sea	730,392	0.012	0.464	100.0	0.951	0.943	0.346	0.487	0.200	0.376	0.641	0.382
Aegean Sea	286,427	0.002	0.701	100.0	0.974	0.917	0.411	0.535	0.388	0.418	0.796	0.417
Cyprus	443,965	0.006	0.423	100.0	0.883	0.945	0.323	0.300		0.321	0.507	0.319

Figure 8. Example of NEAT results from different Spatial Assessment Units (SAUs), corresponding to Mediterranean Sea and aggregation at different levels by ecosystem component, habitat or descriptor of the Marine Strategy Framework Directive (modified from Borja et al., 2019). [D3: Commercial (Com) fish; D5: eutrophication; D8: contaminants], and habitat [Pelagic; Demersal/Benthic: Dem./Ben.]. BoB, Bay of Biscay. The colors indicate the status: High: blue; Good: green; Moderate: yellow; Poor: orange. Confidence (Confid.) refers to the certainty (in %) that the status class corresponds to that calculated.

Ecosystem services mapping and assessment

In everyday life, individuals constantly make choices depending on their needs. Individuals' choices depend on the limited, scarce resources the individual possesses, and the priority each option assumes for that individual (Daly and Farley, 2004). Scarce resources to consider include, for example, time and money. Policy and decision makers will also have to make choices for the benefit of society, e.g., decisions on how to invest the available budget within a country will include considering trade-offs between building a new road to connect villages or conserving a forest for recreational purposes. These kinds of trade-offs, where natural resources are involved, consider potential alternative uses of the natural capital and the ecosystem services that it provides. Environmental economics methods and techniques, including natural capital accounting and environmental valuation, are particularly useful to support decision making on policy and management of natural resources (Bagstad et al., 2021).

When considering the economic valuation of natural capital (Natural Capital Committee, 2014), it becomes important to subdivide ecosystem services in two categories: intermediate and final ecosystem services (Fisher et al., 2009). This distinction is important to avoid double counting the same ecosystem service, considering only those final ecosystem services which provide a benefit to society. Some final ecosystem services can coincide with the benefits, although usually benefits are provided with the support of other capital inputs such as human capital, infrastructure, etc. For example, all fish in the sea is a *final ecosystem service*, whereas the commercial fish to be sold in the market, that is caught by fishers using vessels, is a *benefit*. The economic valuation is in fact performed only on the benefits, e.g., valuing nutrient cycling on its own and also valuing water quality could be problematic as it may double count the nutrient cycling service as this is already part of the water quality assessment. Similarly, valuing places and seascapes separately from recreation may also double count the value of the beauty of the environment (Figure 9).

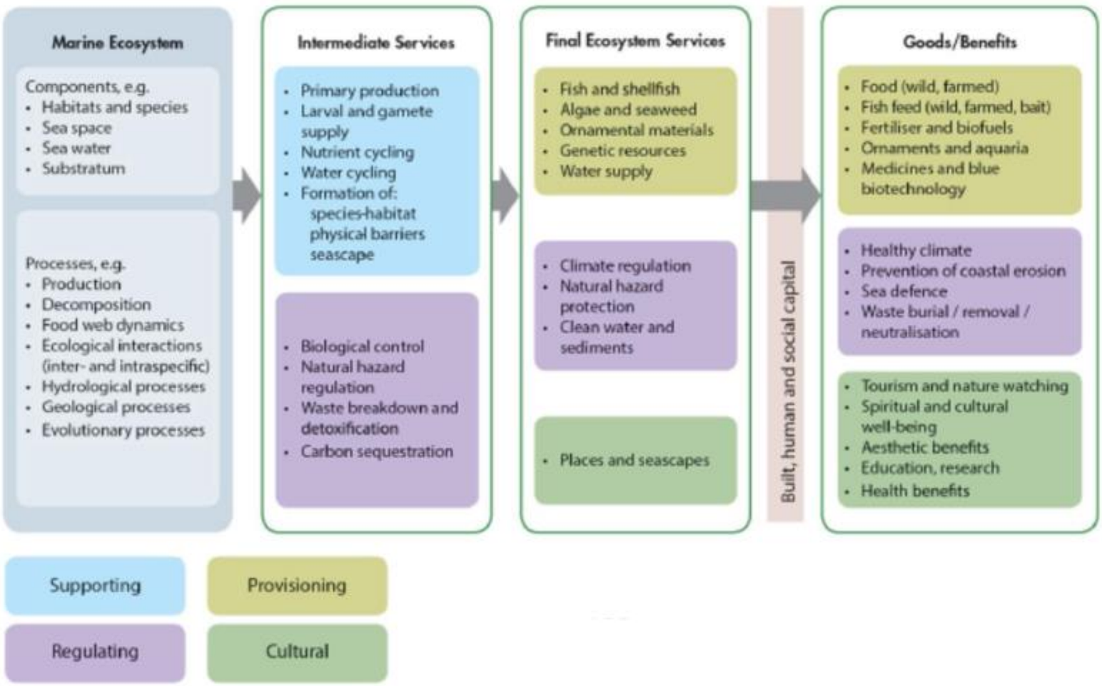

Figure 9. Ecosystem services classification (Source: Turner et al., 2014).

Ultimately, we want to be sure that the natural capital assets are managed in a way that they will continue to be productive and provide current and potential benefits to society.

There are well-established steps that need to be followed to perform an appropriate environmental economic assessment. Since ecosystem services are context dependent (Morse-Jones et al., 2011), their valuation must be spatially explicit; this means that the area where the ecosystem service arises must be known. The extent of the spatial area of the natural capital asset (i.e. the habitat) may be calculated using geographic information system (GIS) mapping. The process and function of the ecosystem service to be valued has also to be known as well as their economic value (€) per unit of measure (e.g. tCO_{2eq} km^{-1}). These data will allow initial calculations of the economic value of the ecosystem service. Natural capital accounting, which follows a more restrictive set of rules to value the natural capital and its ecosystem services, requires further information on the condition of the natural capital (United Nations, 2021). Condition refers to the health of the natural capital assets and if they are still providing relevant ecosystem services and they are in the condition to continue to do so in the future. However, data collection and understanding on condition are demanding and therefore this information is currently often missing from natural capital accounts.

As an example, Luisetti et al. (2011) illustrate a step-by-step approach to support decision making when dealing with coastal management assessment and evaluation. Steps are sequential, although at times overlapping, and deal with well-known issues such as the need of spatial explicitness when dealing with ecosystem services valuation, the need to avoid double counting of ecosystem services and non-linearities in services provision (Morse-Jones et al., 2011). The Luisetti et al. (2011) paper is concerned with managed realignment, an ecological engineering technique that provides a natural sea defense through saltmarshes. The project is assumed to be developed in the Blackwater estuary (UK) and for which an estimate of the economic value of the benefits provided by newly re-created saltmarshes in the area was investigated. The benefits estimated were the provision of a *healthy climate* through carbon sequestration and storage, *recreation in saltmarsh areas* including nature watching and spiritual wellbeing, and *fish nursery benefits* for commercially valued wild fish (seabass). For the *healthy climate* in blue carbon ecosystems (i.e. the saltmarshes), following best practice, GIS maps estimating the areal extent of the new intertidal habitats created with managed realignment were produced and an estimate of the tons of carbon stored each year per hectare was provided. These data are used to know the amount of carbon stored within the area of interest. The economic values of carbon used considered a sensitivity analysis of several values using the social cost of carbon method. More recently, values estimated using other methods are usually included as well, such as the abatement cost method (Luisetti et al., 2019). The carbon price from regulated carbon markets is still considered too volatile to be included in this kind of assessment. For the benefit of *recreation in saltmarsh areas*, in the Blackwater project a specific primary data collection exercise to obtain the willingness to pay to estimate the value of that benefit was undertaken. The willingness to pay data was elicited with a choice experiment method and provided an estimated economic value for the saltmarshes around the Blackwater estuary in the case of potential saltmarsh re-creation. When primary data collection is too onerous, it is possible to use a benefit transfer technique, although that technique has many challenges and limitations (Luisetti et al., 2014). Finally, the *fish nursery benefits* were estimated looking at the potential increased production in the Blackwater newly re-created saltmarshes of seabass. For this valuation, it was important to take account only of the fish that were not going to leave the estuary so to estimate only the contribution of the saltmarshes and not confound the data with other, bigger, seabass entering the estuary. All these benefit values were then used in a cost-benefit analysis (CBA) to assess the viability of undertaking such a policy through this project. For this reason, assumptions under different scenarios were developed; from a hold the line and a business-as-usual scenario to greener scenarios deeply rooted in ecological restoration.

This example is interesting as it shows a trade-off analysis and calculations of opportunity costs between the use of the land for agriculture or for coastal protection, flooding agricultural land and re-creating saltmarshes.

Environmental economics methods and techniques provide decision makers, of either policy design or project development, with tools to evaluate their decision from an economics perspective. The interdisciplinary nature of environmental economics and of natural capital approaches enable the decision maker to gather both ecological and economic information (Bagstad et al., 2021). Natural capital accounting, although requiring a more restricted set of economic rules, can provide comprehensive time-series monitoring of marine biodiversity and natural resources within a country’s national waters while scrutinizing any increment or decrement in their economic value and contribution to economic activities (Luisetti and Schratzberger, 2023). Environmental valuation techniques encompass a wider set of options to value the marine natural resources compared to natural capital accounting (Bartkowski et al., 2015). These techniques can also support the decision maker in evaluating a post-implementation policy or project. Both sets of methods and techniques support the decision maker with the level of information needed depending on the economic analysis needed, including trade-offs analysis such as those for marine spatial planning, and are therefore encouraged.

Towards a unifying framework for assessing ’cumulative effects’ and ’environmental status’

The basic method of doing CIA (see Section 3.1) or cumulative effects assessment (CEA) (Halpern et al., 2008a; Korpinen et al., 2013, 2021; Piet et al., 2021a) is to develop a so-called linkage framework or mental model of all the relevant links (or impact chains) between Activities, Pressures and State (or ecosystem) components, i.e. A-P-S from DAPSI(W)R(M) (Elliott et al., 2017a) that may potentially be impacted and apply a risk-based approach to assess the cumulative pressures on the ecosystem and estimate the relative contributions of each of those stressors. A typical risk assessment consists of two aspects of risk: exposure and effect. The estimation of exposure requires the collection of spatial data from as many as possible of the pressures (e.g. nutrients, pollution, trawling, shipping, acidification) and ecosystem components (e.g. mangroves, reefs, seagrass, open waters) to estimate the spatial overlap. Estimation of effect requires a numerical sensitivity score expressing the severity of the effect the pressure can have on the component. Additionally, the effect can be translated into a potential impact using the population dynamics of the ecosystem component. Finally, all individual results are added up (cumulation by addition) where the difference between CEA and CIA lies in whether or not the final step is included (Piet et al., 2021a).

This method is simple to implement. While the results may give a good estimate of the overall additive effects of human pressures, it still has a number of drawbacks. Some of them were recognized early on by Halpern et al. (2008b) but have not yet been implemented in typical applications of CIA/CEA. We have identified these six most important issues:

(i) Flat structure: Every ecosystem component makes the same (numerical) contribution to the final result. As an example, using one species of five different phyla, plus 10 bird species, will mean that the assessment result will be largely determined by the birds and not by any one of the species of some other phylum, regardless of their functional role or importance in the ecosystem.

(ii) No uniform scaling: Typically, the minimum and maximum observed pressure intensities are rescaled to a common scale of 0–1, possibly doing some logarithmic transformation to keep numbers within a few magnitudes (e.g. Halpern et al., 2008a, 2008b). This, however, does not consider that the minimum/maximum pressure intensities of any two pressures from the dataset do not necessarily result in the same biological effect on the ecosystem component. The sensitivity score alone is not able to compensate for this. Further, ecosystem components are not on the same scale just because the same numeric value was measured for whatever parameter was used to quantify them. As an example, a species richness of five benthic fauna species in a given area is not the same as 5 g m^{-2} biomass of the same or another species or the whole community in that same area.

(iii) Weighting factors: Adding weighting factors to certain ecosystem components as a proxy for their importance will likely just increase the bias introduced by the flat structure (see #1) as some species gain even more influence on the final result of an assessment.

(iv) Static sensitivity scores: Typically, sensitivity scores are derived from expert surveys and are static, i.e. represented by one single value. This leads to a bias. With low levels of a pressure, the reaction of an ecosystem component is often less pronounced than in situations where the pressure level is higher. The reaction does not follow a linear relationship with respect to the pressure level. Situations where pressure levels exceed a specific threshold value, leading to the death of a species, are not considered either.

(v) No uncertainty information: The amount of uncertainty in the final result is often unknown and typically underestimated. This can easily lead to misinterpretations, e.g. when using the data for management decisions where one area must be prioritized over another for conservation. An area might be chosen that would not be considered when the potentially high amount of uncertainty was known.

(vi) No interaction: In the sectoral approach all individual effects are treated independently from each other and cumulation occurs by addition. In reality, however, an effect from one pressure on a specific species may be altered by that of another pressure e.g. when disturbed by noise it is more susceptible to collision or when mortality increases in a species suffering from contaminants and it is additionally disturbed from its preferred feeding grounds. Another type of interaction is a pressure on one species influencing the effect of that same pressure on another species (often called indirect or knock-on effects). For example, when a mammal species gets scared away by underwater noise and leaves a specific area, this will mean it will not feed on e.g. the fish species present in that area anymore. These fish might not be so sensitive to the underwater noise and the larger number of mammals in the other area may result in more fish entering the noisy region, thus increasing the number of fish affected by the noise. Thus, the sole usage of adding effects per component over all pressures may lead to misinterpretations. Lack of information on these interactions and the conceptual simplicity of many CIA/CEAs then lead to ignoring synergistic and antagonistic interactions.

For a reliable and transparent CIA/CEA, these six issues should be considered in any method aiming to assess the cumulative effects of pressures. Here, we outline solutions to every one of these, which we are developing in a new framework to assess cumulative pressures, environmental status and supply of ecosystem services (Borja et al., 2024):

(i) Hierarchical structure: Instead of using an approach where each ecosystem component is treated identically, the structure of the assessment can be made hierarchical, e.g. reflecting the taxonomic relationships of the ecosystem components or using biological traits or other structural information important to the ecosystem. The structuring by taxonomy was already developed in NEAT (see Section 3.3). In this approach, the weight of every ecosystem component (e.g. a species) in the assessment is determined by its position within the taxonomic tree. The specific distribution of weights ensures that every taxonomic level of the taxonomic tree get the same share of the total weight. In practice, taking the example given above, all phyla in the assessment share the weight equally and when there is more than one species in a specific phylum, the weight of that phylum is again distributed evenly among itself and its lower taxonomic groups. Thus, a set of 10 bird species are not able to out-rule the other ecosystem components. This hierarchical structure can also be applied to a hierarchy of habitats or biotopes (e.g. using the European Union Nature Information System

(EUNIS; Davies and Moss, 2002).

(ii) Uniform scaling: Instead of a pure mathematical scaling, a benchmark scale can be used. Such a scale means that the same value of the overall magnitude of a given pressure will result in a similar (biological) effect on an ecosystem component. For this, the scale needs to use common anchor or reference points that define specific reactions of ecosystem components that can be compared, e.g. 0 = no pressure; 0.4 = pressure is starting to have reproductive consequences on the ecosystem component; 0.6 = pressure is starting to have increased mortality; and 1 = the ecosystem component is lost. In order to determine these points, the overall pressure magnitude can be deconstructed into its constituents: intensity, frequency, range, type (direct or indirect), and recovery time after loss. Regarding the ecosystem components, it must be ensured that the same parameter is used for representing the ecosystem component, for example abundance or biomass only. When only mixed data are present, the data can either be reduced to only presence/absence (reducing the level of information by a large margin) or one parameter can be translated into the other by estimation or using a (numerical) translation function.

(iii) Weighting: A natural weighting can be achieved by utilizing the hierarchical structure from #1. The importance of an ecosystem component can thus be determined by its position within the taxonomical tree. A manual weighting factor can then be applied on top of the natural weighting. This should, however, only be done when a special kind of importance needs to be applied. This may be the case when legal settings require to put more weight on e.g. protected or rare species.

(iv) Dynamic sensitivities: Non-linear dose-response relationships can be used instead of a single sensitivity score. A single sensitivity value is equivalent of having a direct linear relationship between the magnitude of the pressure and the resulting (biological) effect on the ecosystem component. This can be modified in various ways, to reflect the typical reactions observed in nature:

- low levels of pressure may not increase the reaction until a threshold point from which the linear response starts;
- the response may be linear but with different values in specific ranges of the pressure magnitude, leading to a piece-wise linear relationship with

different slopes;

- the response can be logarithmic, quickly increasing the response in the first place but then levelling off at higher pressure levels;
- the response may show a sigmoid behaviour with a slow increase of the reaction followed by a (tipping-point like) sharp increase and then slowly levelling off again.

  Hence, instead of using a single value for the sensitivity, the procedure will thus use a sensitivity function that describes the dose-response relationship depending on the magnitude of the pressure on the uniform scale from issue ii.

(v) Uncertainty framework: A dedicated uncertainty framework can be applied to the data. Each pressure value and each value representing the presence of an ecosystem component should have an accompanying uncertainty value capturing the range of uncertainty about the measurement. As an example, this can be a standard error. The same can be applied to sensitivity scores or other measured and estimated data used in the assessment. This enables the application to use e.g. a bootstrapping procedure that will propagate the uncertainty through the whole assessment procedure, resulting in an overall uncertainty for the assessment result. With this, it can be quantified how large the range of uncertainty around the final assessment value is (Carstensen and Lindegarth, 2016).

(vi) Interaction: Known important interactions can be parameterized to support synergistic and antagonistic interactions between ecosystem components, in addition to the additive cumulation being used in the assessment procedure. In its easiest form this can be done by applying a matrix which gives a factor of 1 for all ecosystem component pairs that have no known interactions. Numbers above 1 represent synergistic interactions, increasing the total value of the effect for the paired components. Numbers below 1 represent antagonistic interactions, decreasing the total value. Ideally, these factors would only be applied in one direction, identifying a source and a recipient ecosystem component. Then, the factors can have two different values for each pair of ecosystem components, depending on for which components the cumulative effect is calculated (which component is the recipient component). In a more detailed assessment, the interaction values could be pressure-specific. To derive such detailed information about the interactions between components, it can be helpful to build a conceptual ecosystem model of the ecosystem at hand. If more detailed models are needed, a specific food web model may be used to quantify interactions. This will allow to analyze the relationships of the constituent ecosystem components in a systematic and transparent way.

Given this framework of parameters of a CIA/CEA, other types of assessments can be applied. Instead of using ecosystem components the procedure can be applied to ecosystem functions instead. The estimated sensitivities/vulnerabilities of the various ecosystem components can be translated to the effect on the ecosystem functions which the components have. Ultimately, this is a proxy for their capacity to supply ecosystem services resulting from those functions. To make such assessments, a translation needs to be done, transferring the spatial and quantitative distribution of the ecosystem components into their ecosystem functions as an indication of their capacity to supply ecosystem services.

Often, a CIA/CEA is one of the indicators as part of a status assessment. E.g., CIA/CEA is one of the pillars for achieving good environmental status, *sensu* MSFD (Borja et al., 2024). It is important to note the semantic difference between an effect and a potential impact (Piet et al., 2021a) as estimated in respectively a CEA and CIA as presented here. These potential impacts are different from the concrete impacts as they are measured with monitoring programs that are the basis of environmental status assessments. CIA rather is intended to guide management towards minimizing and mitigating pressures in order to improve environmental status and/or predicting the risk of further deterioration given the current monitored situation. Thus, not every effect is automatically an impact. Further, a potential impact as estimated by CIA does not directly translate into a "status" in the sense of a status assessment (Borja et al., 2013). These considerations show the necessity of a unifying framework for assessing cumulative effects and applying these together with environmental status assessments as part of EBM.

We here propose a systematic approach to CIA/CEA to form such a unified framework. Not every aspect of the framework must be employed every time an assessment is made, but the users should be aware of them, assessing their influence on the final result, transparently documenting what is left out from the analysis and to which degree this increases the uncertainty of the result on top of the pure uncertainty imposed by the data. As a common foundation of the framework, we propose to use DAPSI(W)R(M) (Elliott et al., 2017a), for structuring the problem and determining the implementation of the management cycle. DAPSI(W)R(M) links the human system of Drivers, Activities, Pressures, Status, Impacts, Welfare, Responses and Measures to the natural system of environmental status. The proposed framework follows a typical EBM cycle and starts with the environmental status of the natural system (see Elliott et al., 2017a): (1) Assess the environmental status, using state indicators alone; (2) identify the problematic ecosystem components (the ones not allowing to achieve good status); (3) do cumulative effects assessment; (4) identify the pressures that likely prevent achieving good status for the identified ecosystem components (to inform management); (5) take measures, set or strengthen the thresholds for the respective pressure and wait for the environmental response to happen; and (6) repeat.

This cycle of actions and assessments will allow management to iteratively advance closer and closer to a good environmental status and thus to good ocean health. It uses distinct management phases and separates status assessment from CIA/CEA by putting them into different stages of the cycle. Nonetheless, both status and impact analysis are still integral parts of the cycle and need to be used to complement each other.

Discussion: looking forward in monitoring and assessing the ocean

Long-term marine monitoring has demonstrated high value for informed management decisions addressed to reduce the impacts of human activities and pressures (Borja and Elliott, 2019). Therefore, maintenance of these long-term monitoring networks is critical, and relies on the development of cost-effective implementation approaches (Borja and Elliott, 2013) and methods (Hyvärinen et al., 2021). These methods can be either traditional or rely on novel technologies, with a broad range of sampling, observation and analytical techniques (Danovaro et al., 2016), such as: instruments (e.g. buoys to deploy sensors, physico-chemical sensors, remotely operated vehicles, autonomous vehicles, biosensors, biologgers, passive sampling), molecular approaches (e.g. metabarcoding of eDNA), acoustic devices (e.g. multibeam), flow cytometry, imaging, remote sensing (optical, thermal and radar images from airborne, including drones, and satellite sensors), artificial substrata, etc.

Traditional methods are often time consuming, and costly, which typically translates into pronounced limitations of the spatio-temporal resolution and coverage that is required by policy-makers, stakeholders and decision-makers (Borja and Elliott, 2013; Mack et al., 2020). To improve cost-efficiency, reproducibility and spatio-temporal resolution, novel methods are being proposed, i.e. as proof-of-concept, or already operational but with little application in the marine realm, using five basic criteria (Mack et al., 2020): (i) technology readiness level of seven or higher; (ii) comprehensive and standalone techniques; (iii) filling a gap in the current monitoring frameworks; (iv) novel and not in general use; and (v) evaluated as cost-efficient in terms of their cost-benefit ratio (including investment costs, monitoring costs, reliability, environmental impact, added value, limitations, required expertise). These criteria were all identified and rated for 22 novel methods relevant to marine monitoring, with some of them being included in the presented review (e.g. molecular methods, drones, etc.).

For novel methodologies to prove their usefulness and value to monitor and assess marine ecosystem status and health of coasts and seas, they need to score well on relevant criteria, as those proposed by Mack et al., (2020) and in addition be properly communicated to end-users, especially to stakeholders as management agencies who will be using data generated by such innovative tools for improved management actions (Seys et al., 2022). Some marine organizations have proposed step-by-step tailored guides for an accurate and efficient communication (Seeyave et al., 2017). Usually, this communication requires (i) identifying the target audience (e.g. end-users of monitoring and assessment methods, policy-makers, decision-makers, managers); (ii) break down the communication objectives into relevant messages for each target audience, starting with those with highest priority (e.g. those deciding on monitoring and assessment infrastructures); (iii) for each target, decide the most appropriate channel to communicate (e.g., hard copy, digital, in-person, etc.); and (iv) build the communication implementation plan, including the objectives, target audience, tasks and methods, timing and key performance indicators (Seeyave et al., 2017). An example of a communication plan, from one of the projects participating in the summer school (GES4SEAS), can be seen in Figure 10 (Borja et al., 2024). It includes the outcomes and key messages to be transmitted (“What?”), the target audience (“To whom?”), the multiple channels used, which can include social media, videos, infographics, summer schools, specialized workshops, etc. (“Which channel?”), and also the expected effects on the audience

("What effect?").

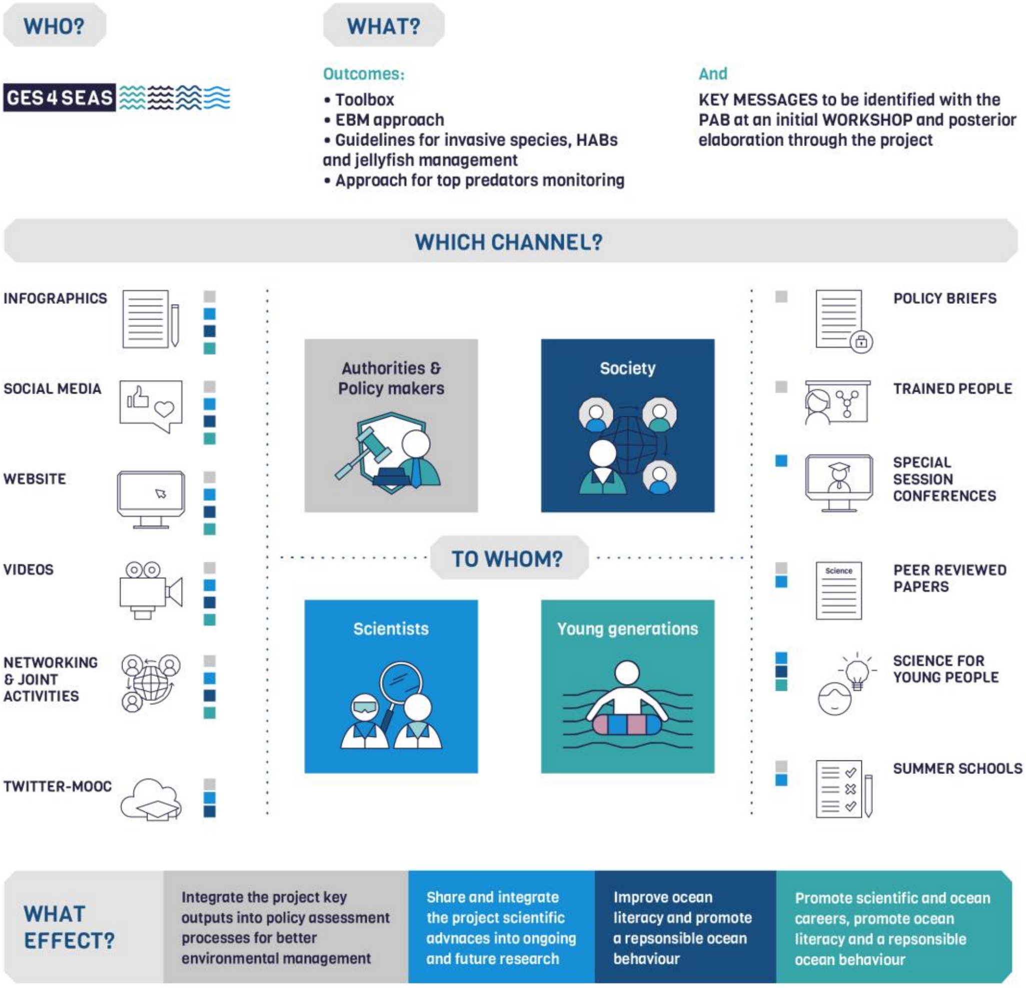

Figure 10. Graphical example of the communication plan from Horizon Europe project GES4SEAS. EBM: Ecosystem-Based Management; PAB: Practitioners Advisory Board; HAB: Harmful Algal Blooms; MOOC: Massive Open Online Courses. The colours associated to each communication channel refer to the target audiences that each channel can reach. Modified from Borja et al. (2024).

Research programmes on monitoring and assessment, based on individual and unconnected projects, can lead to research fragmentation, wasting resources through duplication, dispersion and overlap (Elliott et al., 2017b). To minimize these deficits, the Horizon Europe programme offers an instrument for collaboration among "sister" projects (i.e. those under the same call or addressing close topics), facilitating synergies in their outcomes, communication, stakeholders' participation and increase of knowledge. This has been the case of the summer school which has resulted in this contribution, in which five projects (i.e., GES4SEAS, OBAMA-NEXT, BIOcean5D, ACTNOW and

MARBEFES) have collaborated to bring to the scientific and stakeholders' community the most recent advances in marine monitoring and assessment and the way forward, to better manage the ocean and reduce the effects of multiple human pressures on ecosystem components (Borja et al., 2024).

Regarding those advances, in the summer school discussions the following tools emerged and were evaluated to be important and useful for future monitoring, including remote sensing, drones, eDNA metabarcoding, video, biological sensors, imaging and ML. The applicability and their and pros and cons for marine coastal and marine monitoring have been discussed in the sections above. In the case of assessment methods, the availability of integrative and flexible tools was discussed, highlighting the importance of having capacity to be feed by large open access datasets (e.g. Lowndes et al., 2017; Borja et al., 2019) and having an associated confidence of status classification system. Regarding the attendees comments about the expectations from a marine monitoring tool, they included minimizing the environmental impact (i.e., non-destructive methods, or related to the animal ethics, as in the case of biologging), maximizing repeatability throughout time and space, having reasonable affordability for operation and maintenance, demonstrating the ability to monitor spatially and work autonomously for a long time-period, standardizing measurements, and being cheap, easy to use, precise and efficient.

Conclusion

As a concluding remark, the attendees agreed that one of the most important challenges ahead is associated with integrating the novel methods presented here (or others) into long standing time series for data continuity, alongside current benchmark methodologies. This requires dedicated and well-planned transition periods allowing for quantitative and qualitative comparison of methods to secure consistent and data-driven ecosystem assessments. Such actions can be cover through intense collaborations between current and future European projects on marine biodiversity and ecosystem health.

Author Contributions

AB coordinated the group and wrote the first draft. NRE wrote section on eDNA, KH, AGH, HG, and JES wrote section on drones, SP wrote section on AI and pelagic environments, ET wrote section on climate modelling, GJP wrote section on CIA, SK wrote section on assessing multiple pressures in Europe, IM wrote section on NEAT, TL wrote section on ecosystem services mapping, TB and CM wrote section on a unifying framework, MCU and MCL contributed to communication parts. All authors collaboratively wrote the introduction, discussion and conclusions, contributed to the revision of the article, and approved the version submitted for publication.

Statements and declarations

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Ethics approval and consent to participate

All authors have read the last version of the manuscript prepared and approved it before submission.

Ethical Responsibilities of Authors

All authors have read, understood, and have complied as applicable with the statement on "Ethical responsibilities of Authors" as found in the Instructions for Authors of the journal.

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