Supplemental Material: Papadopoulou et al., 'Horses for courses' – an interrogation of tools for marine Ecosystem-based Management (EBM)

S1. Linking EBM Principles to essential elements and assessment methodologies/tool groups

Table S1. Core EBM Principles for GES4SEAS, key elements within each principle and Methodological approaches/tool groups to assess each key EBM element. MSFD: Marine Strategy Framework Directive; GES: Good Environmental Status; BHD: Birds and Habitats Directives; NEAT: Nested Environmental status Assessment Tool; OHI: Ocean Health Index; BBN: Bayesian Belief Network; HAB: Harmful Algal Blooms, NIS: Non-Indigenous Species; HSM: Habitat Suitability Models; GIS: Geographical Information System.

Core EBM Principles	Key EBM elements within each core EBM principle	Methodological approaches/tool groups to assess key EBM elements
Stakeholder Involvement	co-creation, participation, representation, consultation	iterative and incremental development approach; social science methods soliciting expert views or tools using judgement (e.g., on sensitivity to pressures in Halpern's 2008 global cumulative impacts analysis)
Ecological integrity and biodiversity	overall environmental status, MSFD GES assessments, ecosystem components, species and habitats and assessments e.g., BHD and OSPAR's specific effects and impacts including for example HABs/NIS/eutrophication, threatened species and habitats, biotic effects/ecosystem functions/sex segregation issues, cumulative and in combination effects -– key work theme of GES4SEAS, relevant to MSFD and Regional Sea Conventions and EEA's Assessments	combination of simple/multimetric indicators, risk based approaches and spatial/temporal pressure-impact-mapping and models, integration tools like NEAT, OHI, conceptual models, BBN probabilistic approaches, bow tie risk based approaches exposure-effect-hazard-vulnerability, cumulative impact spatial mapping (Halpern et al. 2008, HOLAS, CUMI), impact risk ranking through linkage-chain-frameworks (ODEMM & SCAIRM approach, Knights et al. 2025, Piet et al. 2023), specific effects and impacts with ecosystem models, food web models, HSM models, multispecies models (e.g., looking at tipping points).
Distinct boundaries	spatial maps and reporting units, relevant for GES4SEAS case studies work and MSFD/EEA/Regional Sea Conventions Assessments	GIS, spatial planning and systematic conservation planning models (MARXAN with zones, prioritize, zonation), VAPEM
Acknowledge Uncertainty	explicitly addressed in various models and tools used and developed by GES4SEAS	various (e.g., in NEAT the calculations are made using Monte Carlo iterations; uncertainty is also assessed by some of the modelling tools)
Consider cumulative impacts	assess cumulative and in combination effects, assess activities-pressures-impacts linkages and spatial and temporal footprints, assess risk and spatial overlap of activities/pressures with vulnerable habitats/ important areas – key work theme of GES4SEAS, relevant to MSFD and Regional Sea Conventions and EEA's Assessments	conceptual models, BBN probabilistic approaches, bow tie risk-based approaches exposure-effect-hazard-vulnerability, cumulative impact spatial mapping (Halpern et al. 2008, HOLAS, CUMI), impact risk ranking through linkage-chain-frameworks (ODEMM & SCAIRM approach), GIS.

Core EBM Principles	Key EBM elements within each core EBM principle	Methodological approaches/tool groups to assess key EBM elements
Use of all forms of knowledge	scientists and stakeholders using all forms of knowledge, implicit in GES4SEAS working with an Advisory Board and local stakeholders in case studies	tools combining quantitative, monitoring, local ecological knowledge data (e.g. on presence and distribution of specific activities or pressures) with expert judgement (e.g. on severity of impacts), social science methods

S2. Narratives on Ecosystem Based Management Elements

S2.01 Cumulative effects assessments:
Cumulative Effects Assessment (CEA) (often used interchangeably with Combined Effects Assessment, Cumulative Impact Assessment (CIA); In combination Effects Assessment; Cumulative Pressure and Impacts Assessment) is defined as the assessment of ecosystem changes that accumulate from multiple stressors, both natural and manmade (Dubé et al., 2013). CEAs are holistic evaluations of the combined effects of human activities and natural processes on the environment, constituting a specific form of environmental impact assessments (ICES (2019a). MSFD (EC, 2008) Article 8 requires Member States (MS) to perform assessments and an analysis of the predominant human activities, pressures and impacts including the main cumulative and synergetic effects on the environmental status of their waters.
The first global assessment was by Halpern et al. (2008), soon to be repeated by Regional Sea Commissions (RSC) and pan-European processes (e.g., HELCOM, 2010, 2023; EEA, 2019; Korpinen et al., 2019, 2021, presenting multiple pressures and their combined effects across the European seas; and Halpern et al., 2019, the latest global assessment). Since then, numerous studies follow covering large regions to small scale applications (e.g., by Micheli et al., 2013a and Coll et al., 2012, for the Mediterranean and the Black Sea; Andersen et al., 2015, 2020b, for the North Sea and the Baltic Sea; Holon et al., 2015, for French Mediterranean coasts; Fernandes et al., 2017, for Portugal; Menegon et al., 2018, for the Adriatic Sea; Hammar et al., 2020, for Sweden, Quemmerais-Amice et al., 2020, for France; Loiseau et al., 2021, for an island of French Polynesia). Studies also deviate by addressing different ecosystem components, instead of whole ecosystems to specific habitats and/or species (see Bevilacqua et al., 2018, for an example on Mediterranean coralligenous outcrops; Vaher et al., 2022, for an example on reef and sandbank habitats in the northeastern Baltic Sea; Carlucci et al., 2021, for an example on cetaceans; Maxwell et al., 2013, for example on marine predators; Coll et al., 2015, for marine vertebrates in the Mediterranean Sea).
Korpinen and Andersen (2016) reviewed 40 marine cumulative pressure and impact assessments (CPIA) around the world and found similar approaches: many of the assessments were based on habitats, mainly benthic but also pelagic, while some focused on species; methodologies including same assumptions (e.g. additive pressures, long lasting effects), a lack of benchmark for the pressures (i.e., a quantitative definition of a certain level of pressure, for which the impact, adverse effect occurs or sensitivity is estimated), a relationship between pressure and impact, based on expert judgment in a categorical way and with few empirical validations (but see Bevilacqua et al., 2018). Borja et al. (2016) also review five commonly used approaches and their commonalities and differences, in terms of use of the Ecosystem Approach; inclusion of multiple ecosystem components and pressures and impacts in the assessment; use of reference conditions; and determining uncertainty among others. Major assumptions and limitations of the CEAs, including uncertainty, have been identified (Halpern and Fujita, 2013), and methodologies have been proposed to assess the inherent uncertainty associated with CEAs, helping to identify data gaps, ensure transparency in decision-making and facilitate adaptive management (Gissi et al., 2017; Jones et al., 2018; Stock, 2016; Stock et al., 2018). Quemmerais-Amice et al. (2020) have proposed several modifications to the Halpern et al. (2008) method. These include a demonstrator tool able to map the Risks of Cumulative Effects (RCE) of different pressures on benthic habitats, the calculation of the Risk of Cumulative Effects' Confidence Index and other aspects such as spreading the pressure effect from point sources for each activity–pressure pair.
Recent assessments with a focus on ecosystem-based marine spatial planning applications are provided by Andersen et al. (2020a) and Fernandes et al. (2017). Additionally, Menegon et al. (2018) and Hammar et al. (2020) show how cumulative impact assessments (CIA) can support ecosystem-based MSP in practice through MSFD pressure-driven CEA, CEA-based marine ecosystem service

threat analysis, maritime use conflict (MUC) analysis, elaboration of scenarios, while developing and applying methodological advancements (e.g., by using the Symphony-tool and the Tools4MSP GeoPlatform, and by addressing uncertainty and some pressures differently) to traditional assessments (e.g., Halpern et al., 2008).

Stelzenmüller et al. (2018, 2020) develop and operationalize a framework for risk-based cumulative effect assessments in the marine environment, thereby working with 11 national, subregional and regional case studies. They conclude with a key recommendation to differentiate CEA processes and their context in relation to marine spatial planning or governance and regulatory advice purposes.

Risk-based approaches have often been at the basis of CEAs for marine management (Tamis et al., 2016; Stelzenmüller et al., 2018). For estimating the risk of a rare or unpredictable event (i.e., calamities) such assessments follow a likelihood-consequence approach (Williams et al., 2011), whereas an exposure-effect approach is considered more suitable when assessing existing and (more or less) continuous or frequently occurring pressures (Smith et al., 2007; Knights et al., 2015). The estimation of exposure and effect can be based on (i) qualitative categories and expert-judgement scores (Knights et al., 2015; Borgwardt et al., 2019), (ii) a fully quantitative approach applying actual data (Piet et al., 2021) or (iii) a mix of qualitative and quantitative information (Piet et al., submitted 2024). This latter approach can be readily applied using a categorical approach, e.g., in a new assessment or in case of a data-poor ecosystem but can be gradually improved over the longer term by incorporating quantitative information as it becomes available. Piet et al. (2023) builds on previous CEA/CIA approaches and their applications to inform management (Knights et al., 2015; Piet et al., 2015, 2017, 2019; Borgwardt et al., 2019; Culhane et al., 2019a). It uses the comprehensive categorical risk-based approaches but modified such that their outcome (Impact Risk from cumulative pressures) is conceptually identical to the outcome from the quantitative approaches.

In terms of indicator-based assessment methods, CumI ("Cumulative impact from physical pressures on benthic biotopes"), is a HELCOM core indicator (HELCOM, 2021) to be used in HOLAS III assessments. CumI evaluates the cumulative potential/expected impact of several physical pressures on the benthic biotopes of the Baltic Sea, (partly) based on pressure-specific sensitivities. The method works with spatial data (e.g., VMS and fisheries SAR) and grids and is applicable to all pressure gradients and to Marine Strategy Framework Directive (MSFD) D6 criteria D6C3 and D6C5. Thresholds for adverse effects are based on a categorial approach (6 disturbance categories from no to high). CumI is comparable with similar indicator development under OSPAR (i.e., BH3). Future developments include a more rigid approach to assess the uncertainty in CumI (WKBENTH3; ICES, 2022b). BH3 is operational at an OSPAR Region-scale, with spatial outputs generated for OSPAR assessments (see OSPAR, 2017b; Elliott et al., 2018) and applications for the MSFD by some Member States. BH3 also has relevance to components of D1- biodiversity (benthic habitats) and applies to D6C3 and informs D6C5 seafloor integrity. Thresholds considered for adverse effects are based on a categorial approach (9 disturbance categories). Outputs are developed with accompanying confidence maps to indicate uncertainty in component data layers used in assessments. Future developments include further exploration of threshold values and inclusion of more human activities (ICES, 2022b).

The OSPAR North-East Atlantic Environment Strategy 2030, Strategic Objective 7 acknowledges the need to "ensure that uses of the marine environment are sustainable, through the integrated management of current and emerging human activities, including addressing their cumulative impacts" (OSPAR, 2021). Taking this forward, in its QSR 2023 report (expected to be published online in July 2023 in the OSPAR website) new and further elaborated approaches to quantifying and assessing cumulative effects include:

a) Modified Bow Tie Analyses (mBTA) undertaken describing the connectivity of the DAPSIR components for each Thematic Assessment (for DAPSIR see section below on linking activities pressures and impacts).

b) Weighting of the Activity-Pressure-State (APS) connections in the mBTA following an adaptation of the ODEMM approach (Robinson et al., 2013; Knights et al., 2015) as an indicative assessment of those activities and pressures of greatest potential concern and meriting priority action.
c) Weighting of the State Impact (SI) connections as an indicative assessment of impacts on ecosystem services following a methodology developed by the OSPAR group on Economic and Social Analysis (Cornacchia, 2022).
d) Sankey diagrams generated for the biodiversity thematic assessments (pelagic habitats, benthic habitats, fish, marine birds, marine mammals) combining the outputs from (b) and (c) for APSI as an indicative assessment of cumulative effects.
e) Confidence in the cumulative effects assessment is also assessed (in line with guidance on communicating the degree of uncertainty in key findings of OSPAR assessments) using two criteria on (i) the level of evidence, and (ii) degree of agreement of the underlying assessments (OSPAR 2019).

S2.02 GES MSFD assessments:
An assessment is both a process and a product. As a process, an assessment is a procedure by which information is collected and evaluated following agreed methods, rules and guidance. It is carried out from time to time to determine the level of available knowledge and to evaluate the environmental status. As a product, an assessment is a report that synthesizes and documents this information, presenting the findings of the assessment process, typically according to a defined methodology, and leading to a classification of environmental status in relation to the determination of Good Environmental Status (GES) (CSWD, 2020).
In the MSFD term 20 defines Good Environmental Status (GES), *"The environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations"* (EC, 2008). European research projects (e.g., DEVOTES), attempted to define the operational meaning of GES *''Good Environmental Status is achieved when physico-chemical (including contaminants, litter and noise) and hydrographical conditions are maintained at a level where the structuring components of the ecosystem are present and functioning, enabling the system to be resistant (ability to withstand stress) and resilient (ability to recover after a stressor) to harmful effects of human pressures/ activities/ impacts, where they maintain and provide the ecosystem services that deliver societal benefits in a sustainable way (i.e. that pressures associated with uses cumulatively do not hinder the ecosystem components in order to retain their natural diversity, productivity and dynamic ecological processes, and where recovery is rapid and sustained if a use ceases)''* (Borja et al., 2013). Numerous EU/DG Environment projects have worked/are working on various aspects of MSFD assessments (e.g., EcApRHA, NEA PANACEA, MEDCIS, MEDREGION, ABIOMMED).
Diverse studies have reviewed and investigated the official assessments undertaken by Member States (Magliozzi et al., 2021, 2023; Palialexis et al., 2014, 2019, 2021). These studies compare approaches and provide recommendations aiming to improve the consistency and representativeness of the MSFD assessments. Most publications on single country MSFD assessments focus on method applications or comparisons and a selected set of descriptors (e.g., Pavlidou et al. 2019; Borja et al., 2021).

S2.03 Whole ecosystem assessments:
Whole ecosystem assessments like MSFD do exist but without the same strict structure and requirements of the MSFD e.g. in the Baltic Sea (HELCOM, 2010, 2023), in the Atlantic Ocean (OSPAR Commission, 2010, 2017a, 2019; OSPAR QSR2023 in preparation https://www.ospar.org/work-areas/cross-cutting-issues/qsr2023), in the Mediterranean (UNEP-MAP, 2017), or in the Black Sea (Todorova et al., 2019).

S2.04 Ecosystem services (delivery, impacts, valuation):
This includes assessments of ecosystem services (ES) in terms of delivery and impacts as well as of value (see also tools narrative section: valuation methods and natural capital accounting methods). *"Ecosystem services are the final outputs or products from ecosystems that are directly consumed, used (actively or passively) or enjoyed by people"* (CSWD, 2020). The Common International Classification of Ecosystem Services (CICES) is the 'EU reference' typology for all ecosystem services. To this end Culhane et al. (2019a, b; 2020) developed an EU-level marine ecosystem-based assessment approach (i.e., a concept framework and method) that considers how ecosystem state affects its capacity for the supply of ecosystem services. The approach provides a way to translate marine ecosystem state into marine ecosystem capacity for service supply. Teixeira et al. (2019) identified the flow from biodiversity to ecosystem services supply for eight case studies across European aquatic ecosystems covering freshwater, transitional, coastal and marine waters realms. Recently, Piet et al. (submitted 2024) conducted a cumulative impact assessment for the North Sea and combined the outcome with an assessment of the Ecosystem Service Supply Potential. Moving away from purely qualitative and score-based assessments, van de Pol et al. (2023) propose a 4-tiered method to aid quantitative assessments of ecosystem services supply linking the supply to the human activity and pressure of interest.
In line with the Biodiversity Strategy 2030 (EC, 2020) and the Climate Adaptation Strategy (EC, 2021), the EU is taking steps ensuring that the risks stemming from climate change and biodiversity loss do not jeopardise the availability of the goods and services that healthy marine ecosystems provide to fishers, coastal communities and humanity at large. By the end of 2023, the Commission will start developing a modelling tool to incorporate the concept of 'natural capital' in economic decisions. This will involve assessing and quantifying both the economic value of marine ecosystem services and the socio-economic costs and benefits derived from keeping the marine environment healthy (EC, 2023). Natural capital was first mentioned in the Biodiversity Strategy to 2020 under the headline motto "our life insurance, our natural capital" with further support for valuing and accounting of natural capital given in the EU Biodiversity Strategy for 2030 (EC, 2020). Although the application of the concept and the progress so far varies significantly between countries and types of ecosystem accounts (e.g., with more carbon accounts being developed than biodiversity accounts) a significant increase in completed ecosystem accounts is expected in the foreseeable future (Lange et al., 2022). Ecosystem accounts are needed to provide consistent information on extent and condition of ecosystems and on the flows of services from these ecosystems to society, and to monitor progress towards a green economy in line with the EU Green Deal (EC, 2019) and towards the Sustainable Development Goals in a Union context (EC, 2022a).
Tables 19 and 20 include a comprehensive description and examples of tools and methods for natural capital accounting, ecosystem services valuation and societal goods and benefits valuation, some of which will be applied by the ongoing EU project MARBEFES (MARine Biodiversity and Ecosystem Functioning leading to Ecosystem Services). MARBEFES has ES in its core objectives along with understanding the causes and consequences of the maintenance, loss and gain of biodiversity and ecological and economic value and the repercussions of this for the management of European seas (https://marbefes.eu/).

S2.05 Special biotic effects/impacts:
One major biotic effect concerns sex segregation i.e., when sexes of a species live apart, either singly or in single-sex groups. Various forms of sexual segregation, due to multiple factors involving natural but also anthropogenic pressures, have been documented for cetaceans, marine pinnipeds, marine birds, marine reptiles, marine birds, and marine teleost and elasmobranch fish (for a review see Wearmouth and Sims, 2008). Documenting the underlying causes of sexual segregation is important for management and conservation reasons as differential exploitation of the sexes by humans (e.g., by spatially focused fishing in key areas or hunting) can lead to population declines (Wearmouth and

Sims, 2008; Mucientes et al., 2009). Understanding the factors that influence the distributions of species is also crucial for implementing effective EBM and conservation practices. Many shark species for example exhibit either spatial and/or temporal sexual segregation, requiring sex-specific habitat suitability modeling (Drymon et al., 2020). Habitats may be selected differentially by the sexes for social, foraging or thermal related reasons (Wearmouth and Sims, 2008). In a recent study, the impacts from climate change in the Mediterranean turtles, that received the highest scores from experts, included: (i) risk to the sex ratios of hatchlings and (ii) risk of current nesting sites becoming unviable in the future (i.e. location of nesting sites), which are very important in conservation decisions for these species (Mazaris et al., 2023). This is because it is well-known that turtles are species with temperature-dependent sex determination (i.e., the sand temperature determines whether male or female offspring are produced) (Hays et al., 2014). Hence, with increasing temperatures under climate change, populations are under threat as existing incubation temperatures are leading to predominantly female-skewed hatchling sex ratios (Patrício et al., 2021).

On the other hand, many studies have highlighted that pollution can alter population sex ratios or the female intersex condition in certain species, e.g., the Tributyltin (TBT) effect on imposex in gastropods (Gibbs et al., 1988; Bauer et al., 1995; Sousa et al., 2005). Similar sex ratio changes have been described for PAHs, PCBs and other pollutants.

More subtle effects have been described in the case of fishing on the demography and population ecology of sex-changing fishes. Hence, species with female-first sex change often have naturally skewed sex ratios in the adult population, and fishing pressure can alter this natural bias, limiting egg production and fertilization success (Chong-Montenegro and Kindsvater, 2022). For two species of groupers (*Epinephelus quinquefasciatus* and *Mycteroperca olfax*), these authors consider how variation in growth rates and fertilization rates interact with fishing selectivity to affect age structure and sex ratios, predicting a decrease both in spawning potential and biomass, causing rapid depletion of sex-changing species.

Changes in temperature at critical developmental stages can induce biases in primary sex ratios in some fish species. Hence, most studies under controlled conditions, conclude that if temperature affects sex ratios, elevated temperatures under climate change warning mostly led to a male bias (Geffroy and Wedekind, 2020). Extreme sex ratios resulting from high rates of environmental sex reversal (either due to climate change, pollution or other human pressure) reduce effective population sizes considerably. This may limit any evolutionary response to the deleterious effects of sex reversal and populations losing genetic sex determination may quickly go extinct if the environmental forces that cause sex reversal cease (Cotton and Wedekind, 2009).

S2.06 Specific ecosystem functions (and impacts on functions)

The biotic and abiotic assets of the marine environment constitute the natural capital held in Europe's seas, i.e., 'marine natural capital'. Part of this capital is depletable, such as marine ecosystems and the services they can supply to people. Ecosystem structures (such as biotic elements as species and habitats and light, dissolved carbon and other physico-chemical elements), processes (such as nutrient uptake, photosynthesis, respiration, excretion, decomposition, biological/ecological and food web interactions) and functions (such as primary production, carbon sequestration, nutrient cycling, ecosystem metabolism, physical engineering; see Strong et al. (2015) for a comprehensive review of functions and their relation to biodiversity) are the foundation of ecosystem services (Costanza et al., 1997; EEA, 2019). The MSFD (EC, 2008) in line with its requirement for 'good environmental status' and 'clean, healthy and productive oceans and seas within their intrinsic conditions', additionally requires that 'the structure, functions and processes of the constituent marine ecosystems allow those ecosystems to function fully and to maintain their resilience to human-induced environmental change'. More specifically it dictates for D6: 'sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected' (EU, 2017a), and for criterion D6C5: *'the extent of adverse effects from anthropogenic pressures on the condition of the habitat type, including alteration to its biotic and abiotic structure and its functions (e.g. its typical species composition and their relative abundance, absence of particularly sensitive or fragile species or species providing a key function, size structure of species), does not exceed a specified proportion of the natural extent of the habitat type in the assessment area.'* A key functional role within the component (e.g., high or specific biodiversity, productivity, trophic link, specific resource or service) or certain life history traits (age and size at breeding, longevity, migratory trait) is noted of particular ecological relevance for the choice of habitats and species for the assessments is (EU, 2017a). In terms of policy relevance (e.g., EU Green Deal, EU Nature Restoration Law) carbon sequestration by blue habitats is a major topic (for restoration and as natural climate solution) and drive for research and assessments (Macreadie et al., 2021; Duarte et al., 2020).

S2.07 Pressures-activities footprint:
Pressure and activities footprints are used to demonstrate the extent of an activity (e.g., Eigaard et al., 2017; Amoroso et al., 2018; showing the extent of the towed gears footprint in European and world shelf and slope areas; Galparsoro et al., 2022a; Gușatu et al., 2021, showing the extent of offshore wind sector; Korpinen et al., 2019, showing the extent of multiple activities and pressures across Europe; the overlap of an activity with a sensitive or important ecosystem components (offshore wind and birds of high conservation concern: Garthe et al., 2023; cetacean distributions overlap with maritime traffic: Awbery et al., 2022; Pennino et al., 2017) and consequently the risk from this interaction (depending on its extent and severity, Knights et al., 2015). A review article by Elliott et al. (2020a) discusses the differences between the Activity-footprints, the pressures-footprints and the effects-footprints (see also the EBM element below). Determining the overall effects of human activities on the estuaries, seas and coasts, as a precursor to marine management, requires quantifying three aspects: (i) the area in which the human activities take place, (ii) the area covered by the pressures generated by the activities on the prevailing habitats and species, and (iii) the area over which any adverse effects occur. These three features correspond to the activities-footprints, the pressures-footprints and the effects-footprints. The first two might or might not have the same footprint (e.g., fishing or aquaculture can cause localised near field effects such as mortality or physical habitat changes within the activity's footprint but they can also cause effects some distance away from operations by extensive resuspended sediment clouds, increased nutrient and carbon loads, biogeochemistry changes, spread of aliens and disease; Puig et al., 2012; Bradshaw et al., 2021; Weitzman et al., 2019). The effects footprints in turn incorporate both the effects on the natural system and the effects on ecosystem services from which society extracts goods and benefits. Quantifying the three footprints has repercussions for marine governance, is essential for EBM at national and regional levels and for cumulative and transboundary assessments. Cormier et al. (2022) discuss further the difference in scales and the links of these footprints with the management response footprints.
The term activity footprint has been defined as *"The area, and/or time, based on the duration, intensity and frequency of an activity which ideally, has been legally sanctioned by a regulator in an authorisation, license, permit or consent, and which should be so clearly defined and mapped in order to be legally-defendable; it should be both easily observed and monitored and attributable to the proponent of the activity"* and Pressures Footprint as *"The area and time covered by the mechanism(s) of change resulting from a given activity or all the activities in an area once avoidance and mitigation measures have been employed (the endogenic managed pressures)"* Cormier et al. (2022). It does not necessarily coincide with the activity footprint and may be larger or smaller. It also needs to include the influence and consequences of pressures emanating from outside the management area (the exogenic unmanaged pressures); given that these are caused by widescale events (and even global developments) these are likely to have larger scale (spatial and temporal) consequences (Cormier et al., 2020). Important considerations for the pressures-footprint include the frequency of the activity as well as the spatial extent and temporal duration.

S2.08 Effects footprints (and/or impacts footprints):
As with the category activity-pressure footprint this category brings in a spatial element in the assessments of impacts and effects which is an essential part of EBM and EU conservation policies (e.g., HD and MSFD). As effects and impacts are more difficult to identify and assess activity and pressure footprints are often used as proxies. For example, in simple regional risk screening exercises, the spatial extent of trawling is often used as a proxy for the extent of the pressure seabed abrasion which then is assumed to represent the extent of the fishing impacts on the benthic communities as accounting for other impacts (e.g., smothering sub-lethal effects on growth) is more difficult (ICES, 2019b, c). Spatial distribution (extent) in combination with fishing intensity layers are used to assess benthic impacts footprint for example underpinning the OSPAR 'Extent of Physical damage indicator' (BH3) (OSPAR, 2017b; ICES, 2021b). Specific impacts such as macrofauna depletion or impacts on mean benthic longevity or changes in biological traits are also investigated closely linked with the spatial extent of fishing (de Juan et al., 2009, 2020; Hiddink et al., 2017, 2020; Mazor et al., 2021; Pitcher et al., 2022; Smith et al., 2023). HELCOM and OSPAR are also developing indicator and risk-based assessment approaches based on species and habitats sensitivities and various disturbance layers (Elliott et al., 2018; González-Irusta et al., 2018, Serrano et al., 2022).
The effects footprint is defined as: The spatial (extent), temporal (duration), intensity, persistence and frequency characteristics resulting from (i) a single pressure from a marine activity, (ii) all the pressures from that activity, (iii) all the pressures from all activities in an area, or (iv) all pressures from all activities in an area or emanating from outside the management area. They will have adverse consequences on the natural ecosystem components, but also are likely to affect the ecosystem services from which society gains goods and benefits. Hence, the determination of the effects-footprint needs to include the near-field and far-field effects and near- and far-time effects because of the dynamics and characteristics of marine areas and the uses and users of the area. Similarly, the effects footprints may be larger in extent and more persistent than the causing activity-footprint and the resulting pressures-footprints. They also need to encompass the effects of both endogenic and exogenic pressures operating in that area (Cormier et al., 2022). Beyond the spatial and temporal extent of the effects of pressures arising from an activity, their magnitude may also be considered (ICES, 2019a). The activity and effect footprints might differ as the effects may be near-field (within the immediate vicinity of the pressure) or far-field (at distance as the result of physico-chemical dispersion or biological migration).

S2.09 Linking activities, pressures and impacts:
Linking activities, pressures and impacts is an essential part of the MSFD (EC, 2008) and a crucial element of EBM. Numerous variants of the original DPSIR frameworks exist (Patricio et al., 2016; Smith et al., 2016) covering multi-space cycles, from the original simple P-S-R chains (e.g., as used by OECD (2003) and required as a minimum by the MSFD) to tetrahedral DPSIR and addressing various issues or actions relevant to EBM. Two such examples include Ecosystem Goods and Services being directly linked to EBM (through EBM-DPSER: EBM- Driver-Pressure-State-Ecosystem Service-Response (Kelble et al., 2013) or the Ecosystem Services and Societal Benefits (ES&SB) linked-DPSIR approach (Atkins et al, 2011). A recent variant offering more clarity on aspects of the EBM cycle is DAPSI(W)R(M) (Drivers, Activities, Pressures, State change, Impacts on human Welfare, Responses by Measures). This framework explicitly refers to human activities and separates these from both drivers (i.e., basic human needs such as food, shelter, security, and goods) and pressures resulting from human activities (Elliott et al., 2017).
The EC is using the similar DAPSES-MMM (socio-economic Drivers, human Activities, Pressures, State of environment, Ecosystem Services – Management (policies and governance), Measures, Monitoring) (CSWD, 2020) framework for the MSFD linking not only activities pressures and impacts but also relating to the various EBM elements and selected articles of the MSFD (especially Article 8: Assessments; Article 11: Monitoring programmes; and Article 13: Programmes of measures).

OSPAR (OSPAR, 2019) is using the DAPSIR (drivers, activities, pressures, state, impacts, response) framework. For the QSR 2023 a suite of Thematic Assessments is being produced describing human activities, pressures and biodiversity in the context of, an adapted from Elliot et al. (2017) and Judd and Lonsdale (2021), DAPSIR framework embodying all the components of the ecosystem approach. This framework incorporates the Bow tie analyses undertaken for risk management and is fully compatible with the DAPSES-MMM framework (OSPAR 2019). HELCOM is using the DAPSIM framework (drivers, activities, pressures, state, impact, measures) and its Spatial distribution of Pressures and Impacts Assessment (SPIA) addresses the cumulative burden on the environment caused by human activities in the Baltic Sea region (HELCOM, 2023). In addition, HELCOM is using the ACTION tool (Ahtiainen et al., 2021), which is a probabilistic tool to estimate the effectiveness of actions in reducing pressures (or improving directly the ecosystem) and ultimately leading to improved ecosystem state (and the agreed environmental policy targets). This way, sectorial outlook assessments were used to predict changes in human activities for a near future ("D"), reduced pressures were estimated on the basis of measure impacts ("R") on human activities exerting them ("P"), the reduced pressures improved the state ("S") or the measures improved this directly, the measures had costs for the society ("I").

All of these EBM approaches and related DPSIR derivatives (DAPSIWRM, DAPSES-MMM, DAPSISR and DAPSIM), aim to link activity/pressures (e.g., in single and cumulative assessments), to environmental impacts/effects (e.g., addressing key structures and functions and effects on selected species/habitats), and to adverse effects impairing the capacity of ecosystems to function properly (i.e. below GES) and deliver the ecosystem services linked to societal goods and benefits. Importantly these include human health and human survival and welfare and as such there is an urgent need to close the gap in assessing them and to quantify the relationship between compromised and degraded ecosystems and the delivery of services (EEA, 2019; Liquete et al., 2013a, b).

To track and assess the impacts of all the human activities on the ecosystem and its components use can be made of a linkage framework (Knights et al., 2015). The basic elements of the linkage framework are activities (note: sectors in the original, but conceptually closer to human activities) pressures and ecosystem components and how these are connected: activities can cause a range of pressures which, in turn, may impact one or more ecosystem components (Knights et al., 2013; Tamis et al., 2016; Piet et al., 2017). Knights et al. (2015) referred to this linear interaction between a "sector, pressure, and ecological component" (e.g., species or habitats) as an "impact chain".

Within the EU project ODEMM, impact chains were defined following an extensive review of the peer-reviewed scientific literature and published reports (Knights et al., 2013) resulting in a pre-pressure assessment matrix of 4,320 potential impact chains. Knights et al. (2015) identified and evaluated 3,347 sector – pressures linkages that can affect the ecological components of Europe's four regional seas noting that the number of impact chains varied between regions as a result of differences in the types of economic activities operating in each sea, the spatial extent of their operations (e.g. over a different range of habitats) and the type and number of pressures introduced.

In a following EU project AQUACROSS, Borgwardt et al. (2019) identified impact chains that link 45 human activities through 31 pressures to 82 ecosystem components. In this linkage framework >22,000 activity-pressure-ecosystem component interactions were found across seven European case studies (Borgwardt et al., 2019). Pedreschi et al. (2023) working in the Irish Sea have identified 1,592 impact chains while Paramana et al. (in press) working in the EU project ABIOMMED with 4 Mediterranean countries and 5 pressures linked to Descriptor 6 identified 1,680 impact chains linked to 24 activities and 14 habitats. These linkage frameworks go beyond the total risk as seen by the number of impact chains and working with 5 criteria and risk scores can inform on risk to specific ecosystem components (e.g., species groups or habitat types or bathymetric zones) by activity and pressure helping thus management measure prioritization. The framework is used by ICES (ICES, 2021a) to produce standardised comprehensive ecosystem overviews for the ICES areas using 3 criteria: spatial extent, and degree of impact with a slightly modified risk scoring (e.g., including future threats).

Sustaining the supply of ecosystem services requires understanding the activity-pressures-impacts chains and how these can lead to changes in the supply of services. A way to assess this is by assessing the total potential for ecosystem service (ES) supply by an ecosystem component (EC) based on EC-ES linkage matrices (working with expert judgement on presence-absence of a link and strength of a link) and the 'risk to ecosystem service supply' approach developed by the EU project Aquacross (Culhane et al., 2019b; Teixera et al., 2019).

The application of the linkage frameworks requires extensive expert judgement (especially when determining the sensitivity to impacts and the resilience/time to recovery to pre-impact conditions), and comparability between studies is not guaranteed as is dependent on the categorization of pressures and activities. Steps are taken however in the most recent applications (Pedreschi et al., 2023; Paramana et al., in press) to refine and align the approach with the revised MSFD criteria and by using the MSFD elements and pressure lists (EU, 2017a, b). Other refinements are being made to reconcile the use of accurate geospatial data with expert knowledge (Piet et al., 2023).

S2.10 Single MSFD descriptors/single issues:

This includes primarily assessments on major regional issues such as IAS, HABs, jellyfish blooms and eutrophication.

Invasive alien species (IAS) are non-native organisms that cause significant harm to the ecosystems they invade, their ecosystem services, and human health (Tsirintanis et al., 2022). These species can disrupt native ecosystems by outcompeting native species for resources, altering food chains, and degrading habitats. Invasive species can also cause economic and social damage by impacting fisheries and aquaculture, spreading diseases, and interfering with human activities such as recreation and tourism. In addition, invasive species can threaten biodiversity by causing the local extinction of native species, altering genetic diversity, and reducing the overall resilience of ecosystems. Overall, the impacts of invasive alien species can be severe and long-lasting, making their prevention and management critical for the health and well-being of both natural and human systems. The ecological and social impacts of IAS have been reviewed at the European level by Katsanevakis et al. (2014) and recently in the Mediterranean Sea by Tsirintanis et al. (2022). Horizon scanning can be conducted to identify emerging IAS, e.g., Tsiamis et al. (2020) developed a scoring tool that aims at identifying the most likely invasive species in European waters. Katsanevakis et al. (2016) developed a standardized, quantitative method for mapping cumulative impacts of invasive alien species on marine ecosystems, based on a conservative additive model; their model is called CIMPAL ('Cumulative IMPacts of invasive ALien species') and has been applied in the Mediterranean Sea (Katsanevakis et al., 2016), all European Seas (Korpinen et al., 2019; but also in the freshwater environment for all European catchments (Magliozzi et al., 2020).

Jellyfish blooms can have significant impacts on marine ecosystems, as well as on human activities such as fishing and tourism. When jellyfish populations increase rapidly and uncontrollably, they can consume large amounts of plankton and fish eggs, which can lead to a decline in fish populations and affect the entire food chain. Moreover, large blooms of jellyfish can clog fishing nets, damaging fishing equipment and reducing the catch. Jellyfish can also sting swimmers and beachgoers, which can lead to discomfort and allergic reactions. The economic impacts of jellyfish blooms can be substantial, particularly in areas where tourism and fisheries are important industries. Jellyfish is a key group for the evaluation of the ecological condition of marine waters, as it is considered as an indicator taxon for the health of marine ecosystems. Jellyfish have been proposed by OSPAR for inclusion in the indicator "Changes in Phytoplankton and Zooplankton Communities" that was used towards the assessments of pelagic habitats and food webs in the Intermediate Assessment of 2017 (OSPAR, 2017a). The Black Sea Commission includes a section on the "State of Gelatinous Plankton" within their "Chapter 1: State and Dynamics of The Black Sea Ecosystem" (BSC, 2019). Jellyfish blooms are attributed to several natural and anthropogenic causes. The latter were reviewed by

Purcell et al. (2005), who examined cases where jellyfish blooms have increased and analyzed how several human activities may have contributed to that.

Harmful Algal Blooms (HABs) occur when certain species of algae grow rapidly, often producing toxins that can harm aquatic animals and humans. These blooms can have significant impacts on the environment, fisheries, and public health. For example, HABs can cause mass mortalities of many wild or farmed species (Hsia et al., 2006; Katsanevakis et al., 2014). HABs render coastal waters unsuitable for recreational activities due to health risks and may cause the temporary closure of beaches (e.g., Figgatt et al., 2017). HAB-related mucilage outbreaks may have serious impacts on maritime operations by clogging sea chest filters and causing overheating and damages to the main engine, generators, compressors, or the cooling systems, and increasing operational costs (Uflaz et al., 2021). Many HAB-causing species produce toxins, which accumulate in shellfish and other invertebrates and may then be transmitted to other commercial species through the food chain (Costa et al., 2009; Lage and Costa, 2013), and eventually to humans (Pérez-Linares et al., 2009; Katsanevakis et al., 2014). Understanding the causes and effects of HABs is critical for managing and mitigating their impacts on aquatic ecosystems and human activities. So far, HABs are primarily managed as a public health problem (Food Hygiene Regulations (EC) No. 853/2004 and (EC) No. 854/2004) to address their toxicity hazards. Nevertheless, HABs are linked to various MSFD descriptors, such as Descriptor 1 - Biodiversity is maintained, Descriptor 4 - Elements of food webs ensure long-term abundance and reproduction, Descriptor 5 - Eutrophication is minimized, Descriptor 6 - The sea floor integrity ensures functioning of the ecosystem, and Descriptor 9 - Contaminants in seafood are below safe levels.

Guérin and Lizińska (2022), working for the NEA PANACEA EU project review MSFD Descriptor 6 (sea floor integrity) assessments on physical disturbance and physical loss as well as links with Regional Seas' Conventions and D4 (food webs integrity) and D5 (eutrophication). HELCOM (2023) presents the thematic Baltic Sea assessments, the spatial maps of potential impacts of physical disturbance on sea floor (the 'CumI tool'), NIS and hazardous substances, the potential effect of continuous noise to mobile species and their habitats, and various single pressures maps and human activities distribution maps. Especially the CumI tool is a spatial sea-floor specific CEA tool estimating potential cumulative impacts on benthic habitats (HELCOM, 2021).

The perhaps best assessed and mapped single issue is eutrophication, where HELCOM and OSPAR regularly have published indicator- and tool-based assessment (HEAT and OSPAR COMP; see EEA, 2019a for summary). Another single issue is contamination and indicator- and tool-based assessments (e.g., CHASE) have also been published regularly by HELCOM and OSPAR (see EEA, 2019 for summary).

S2.11 Single species, ecosystem components state change:

This includes assessments of single species status (e.g., under CFP for commercial species, under MSFD for descriptor D1 Biodiversity or D3 Commercial species) or assessments of a habitat (e.g., for reporting for HD or within an MPA or within a region) and looking at changes of status due to pressures. Ecosystem components are constituent elements of an ecosystem, particularly its biological elements (species, habitats and their communities), or of marine waters (CSWD, 2020). An ecosystem component can also be defined as an attribute or set of attributes of the natural environment (Cooper, 2013). Alternative terms used may be valued ecosystem component (VEC), ecological component, receptor, indicator (Tamis et al., 2016). Ecosystem components should be comprehensive in the sense that they are expected to represent biodiversity and all its different life forms and should thus cover all the biota of the ecosystem. At the most basic structural level ecosystem components include species groups (e.g., birds, mammals) and habitat types including their pelagic and benthic biota (see EU, 2017a, for the MSFD groups of ecosystem components). Each of these may be split up into increasingly smaller ecosystem components depending on the aim of the assessment (e.g., MSFD, CEA), as identified in a scoping exercise (MacDonald, 2000; Therivel and Ross, 2007; Tamis et al., 2016). The likelihood of change of state of an ecosystem component when a pressure is applied can be referred to as sensitivity or effect potential, and is a function of the ability of that receptor to avoid interaction, tolerate or resist change (resistance), and/or its ability to recover from impact (resilience) (Tillin et al., 2006; Piet et al., 2024 submitted).

S2.12 Threatened habitats and species:
This includes documenting status and distributions of species and habitats at risk at global (e.g., O'Hara et al., 2019, for species; Gubbay et al., 2016, for habitats), regional (see OSPAR 2020 threatened habitats and species list and spatial dataset now available in EMODNET: https://emodnet.ec.europa.eu/en/ospar-threatened-andor-declining-habitats-spatial-dataset, the IUCN red list: https://www.iucnredlist.org/regions/europe) or local levels (Nebot-Colomer et al., 2021; Zotou et al., 2020).

S2.13 Climate change:
Climate change is relevant to EBM to both planning mitigation and adaptation actions and evaluation phases from both a managers' and a scientists' point of view. Modelling tools can address changes in species distributions due to climatic effects (e.g., Fabbrizzi et al., 2023; Antão et al., 2020; Hodapp et al., 2023; Pennino et al., 2020) and these insights are relevant to the planning phase to conservation spatial planning and restoration prioritization. For the evaluation and assessment phase it would be useful to know whether the outcome of the evaluation is affected by climate change as well as the trajectory of change. Recent works investigate the combined effects of climate change and other human pressures (Gissi et al., 2021; Korpinen et al., 2021; Vilas et al., 2021).
Doxa et al. (2022) developed a depth-specific prioritization analysis to inform the design of networks of protected areas, further including metrics of climate-driven changes in the ocean (planning phase). Climate change was captured in this analysis by considering the projected future distribution of >2000 benthic and pelagic species inhabiting the Mediterranean Sea, combined with climatic stability and heterogeneity metrics of the seascape to identify climatic refugia.

S2.14 Pressure and impact reduction/mitigation:
This includes targeted and specific pressure and impact reduction or mitigation measures. For example, for mitigating the impacts of IAS in the marine environment a variety of options have been proposed and assessed (Thresher and Kuris, 2004; Giakoumi et al., 2019). Management options ranked high by experts and stakeholders include physical removal, promotion of commercial exploitation, and environmental rehabilitation. On the other hand, biological control, with alien agents or genetically modified pathogens, generally ranked the lowest. The only two species for which large-scale control efforts in the EU have been implemented are the lionfish *Pterois miles* and the silver-cheeked toadfish *Lagocephalus sceleratus.* In Cyprus, to control the population of *L. sceleratus*, targeted, intense fishing pressure on the species by the coastal professional fleet has been promoted by a bounty (3 €/kg) since 2012; there is anecdotal information by fishers of effective reduction of its population and mitigation of its socio-economic impacts (A. Petrou, pers. comm). For the lionfish, physical removal methods have been extensively applied in the western Atlantic and the Mediterranean, based on culling by divers, often with public participation (e.g. by organizing lionfish tournaments), physical removals involving fishers, promoting the targeted fisheries of the species and human consumption (or other uses), developing specific fishing gear such as the 'Gittings' traps (Harris et al., 2020), and recently developing UW robots, which may harvest lionfish (Sutherland et al., 2017).
Other pressure and impact reduction methods include rebuilding fish stocks to Maximum Sustainable Yield levels (in line with CFP and EU SDG goals) and above to reduce negative impacts on marine ecosystems; increasing the selectivity of the fishing gears (e.g., with technical measures increasing the net mesh size, fitting escape panels or with various technical modifications and devices: Mytilineou et al., 2020, 2021; Squires et al., 2021) and implementing the landing obligation to reduce bycatch and unwanted catches contributing to the decline of marine resources and of

protected species (Lucchetti et al. 2019; Fauconnet et al., 2023); improving the CFP governance, ensuring compliance with the obligations stemming from the EU environmental legislation and the Technical Measures Regulation and by Strengthening the ecosystem-based approach through better science taking into account the various types of human impacts and management systems on the use of natural resources and the marine environment, and, vice versa, the impacts of the state of natural resources on the fisheries sector (EC, 2023b). McConnaughey et al. (2020) review nine measures and industry actions under four classes: (i) *technical measures* (e.g., changes in gear design and operations), (ii) *spatial controls* (e.g. gear-specific prohibitions, freezing the trawling footprint, nearshore restrictions and coastal zoning, prohibitions by habitat type including real time (i.e. 'move-on rules') and multipurpose habitat management, (iii) *impact quotas* (i.e., output controls that include invertebrate bycatch or habitat-impact quotas) and (iv) *effort controls* that affect the overall amount and distribution of trawling. Based on this review they offer insights on best practices for managing impacts of trawl fishing on seabed habitats and biota. Area-based fisheries management measures ABFMs) can be related to the sustainable use of resources but are increasingly considered broader conservation measures and include the placement of Vulnerable Marine Ecosystems (VMEs) closures, fisheries closures, species-specific closures, fishing bans and conservation areas among others (Petza et al., 2023).

Inputs of contaminants, substances and of energy and litter may require pressure and impact reduction measures to levels that do not adversely affect the marine environment and are or remain in line with levels established by Union legislation or other relevant standards (EU, 2017a, Descriptors D8-D11). In the case of marine litter (descriptor D10) relevant measures include preventive measures by reducing the occurrence or reducing at source (e.g., production, waste management and recycling, establishment of port reception facilities, gear marking to tackle lost and discarded gears, EU policies such as the MSFD establishing threshold values for litter, the EU Single use plastics directive introducing several bans and restrictions on different uses and materials including for plastics and derelict fishing gear, the EU Green Deal and Zero pollution ambitions); mitigating measures through reducing debris disposal and dumping regulations; and curative measures to remove litter from the marine environment through clean up technologies and actions and fishing for litter campaigns (Gallo et al., 2018; Madricardo et al., 2020; Van Loon et al., 2020; Gilman et al., 2021). Various mitigation measures depending on affected species groups (e.g., mammals, birds or fish) and the phase of operation (e.g., construction, installation, operation, maintenance, decommissioning) are given in relation to type of impact (e.g., from noise, electromagnetic fields) and the offshore renewable energy generation sector by Soukissian et al. (2023).

As with marine litter, thresholds values are also introduced, through cooperation at Union level, for D6 Seafloor integrity restricting the extent of physical loss allowed but also the maximum allowed adverse effects from anthropogenic pressures on the condition of the habitat type, including alteration to its biotic and abiotic structure and its functions (e.g. its typical species composition and their relative abundance, absence of particularly sensitive or fragile species or species providing a key function, size structure of species) (EU, 2017a). These thresholds will require additional measures to attain GES including by reducing both the extent (e.g., by closures, exclusion zones, protected areas) and the severity of operations (to remain within the light impact categories by lighter operations or through stricter licensing) and will work in synergy with the EU Biodiversity Strategy and the proposed EU Nature restoration Law aims for both increased protection and restoration of degraded habitats. Pressure and impact criteria are set out in the MSFD conceptually linked to GES as an EBM to assessment of GES follows the main elements of the system: the state-based Descriptors (such as D1 Biodiversity and D6 Seafloor integrity) and the adverse effects of pressures from human activities via their environmental impacts (e.g., by the pressure-based descriptors such as D2 alien species, D6 seafloor integrity and D10 litter) (EC, 2022b; CSWD, 2020).

S2.15 Spatial and other measures:

This includes spatial and other measures related to the management response and management footprint. Well known spatial measures include the ban of trawling in the deep habitats in the Mediterranean (with a depth operations limit to 1000 m although the GFCM is currently discussing proposals to revise this to depths between 600-800 m depth) and the North East Atlantic (with a depth limit of trawl operations at 800); the ban of trawling within 3 nm- 1.5 nm distance from coast/shallower than 50 m depth and over protected habitats like *Posidonia oceanica* beds, coralligenous habitat, and maërl beds in Mediterranean (EU Mediterranean Regulation (EC) No 1967/2006); and various regional 'Fisheries Restricted Areas' (FRA) that aim to protect Essential Fish Habitats (EFH) and/or sensitive habitats of high ecological value, such as Vulnerable Marine Ecosystems (VME) such as the recent Jabuka/Pomo Pit FRA in the Adriatic with different measures implemented by one or both of the two main countries, Croatia and Italy (FAO, 2020). Giménez et al. (2020) build on a spatial planning approach with MARXAN to design a proactive area-based protection strategy towards elasmobranch conservation in the Western Mediterranean.
Various spatial measures and move on rules are implemented in the NEA and other regions to protect VMEs from fishing operations (e.g., as set out at the EU Regulation (EU) 2016/2336, the so-called the "Deep-sea Access Regulation" (DSAR); Auster et al., 2011; Geange et al., 2020; Walmsley et al., 2021). Various networks of MPAs (e.g. SPAMIs in the Mediterranean) and numerous national MPAs, marine reserves and local spatial closures (and temporal closures) also fall within this category aiming to protect species and habitats particularly important in terms of biodiversity or services and/or threatened/protected habitats and species (e.g. mammals and sea turtles; typically MPAs such as the international Sanctuary PELAGOS combine spatial measures with maritime speed limitations for passing vessels, https://www.sanctuaire-pelagos.org/en/). Other spatial measures are related to licensing and restricting the footprint of certain human activities for example by designating areas for offshore wind farm development, for aquaculture development, for aggregate extraction (e.g., the Belgian MSP at the EU MSP platform at https://maritime-spatial-planning.ec.europa.eu/countries/belgium, and the EMODNET portal of Human activities https://emodnet.ec.europa.eu/en/human-activities). The effectiveness of these protected areas is as important as their extension (Arneth et al., 2023) and currently no-take areas or highly protected areas cover very small areas of European regional seas, such as the Mediterranean (Claudet et al., 2020).

S2.16 Uncertainty:
Uncertainty applies to all phases related to assessments in that the managers prefer to get advice that includes uncertainty/confidence both in terms of consequences of planning scenarios or management plans and in assessment outcomes. However not all tools address assessment uncertainty.

S2.17 Risks:
Different tools address different aspects of risk, including risk from action or inaction, risks to ecosystem components due to spatial overlap with activities and pressures (e.g., EU ODEMM and AQUACROSS projects approaches), risks related to specific biotic effects (e.g., sex related fishing impacts or fishing impacts on nesting or nursery areas; UNEP-MAP-RAC/SPA 2010; Ogburn, 2019). Some examples can be seen in the tool narratives section in Supplementary Material S3.

S2.18 Other policy requirements e.g., MSPD, BHD, Biodiversity Strategy:
Different tools will address different policy needs, e.g., conservation planning tools and habitat suitability modelling can provide outputs relevant for the MSPD or the Biodiversity Strategy and the proposed EU Nature Restoration Law (Michelli et al., 2013b; Korpinen et al., 2021; Fabbrizzi et al., 2020) while threatened species and habitats assessments will support the BHD or the needs of RSC (e.g., OSPAR, HELCOM). Food web models can support the assessment of nutrient reduction policies (Piroddi et al., 2021).

S3. Narratives on assessment methodologies/tool groups

***S3.01.* Conceptual models (including argument mapping, mind-mapping, horrendograms, organograms, etc)**
A. Description of the methodological approach: EBM relies on an ability to structure the analysis of the ecosystem in relation to the ecological, cultural and social systems that drive governance and management processes to ultimately gain a better understanding of their interplay. Addressing any problem should start by creating an actual or virtual conceptual model (analogous to argument mapping, mind mapping, horrendograms and organograms) which are graphical representations and as such, they are used extensively for defining, interrogating and communicating cause, consequence and response sequences (see below an example schematic of a horrendogram by Elliott et al., 2014. They are of particular value in representing the continuum from ecosystem physical and chemical attributes to ecological structure and functioning, to ecosystem services flow and to societal goods and benefits provision; they can include the pathways whereby each of these elements get degraded and recovered through human actions. Hence, they show components, processes, and linkages that form a social–ecological system. Their use encompasses the following definitions: a system is a set of things working together as parts of a mechanism or an interconnecting network; a set of principles or procedures according to which something is done; an organized scheme or method. A process is a series of actions or steps taken in order to achieve a particular end. Management is the process of dealing with or controlling things or people, and governance is the action or manner of governing through authority, decision-making and accountability.

The term framework is used as a basic structure underlying a system, concept, or text and approaches can be separated into frameworks, theories and models – hence EBM requires a framework that aims to identify the necessary components and the links between them prior to analysing, interrogating and then using the framework. This also allows the questions and hypotheses in sustainable resource management to be defined and addressed (see Elliott et al., 2020b for a discussion of systems analysis).

Systems analysis relies heavily on diagrams to show the complexity of the natural and human systems and its management; this could be taken further as there are possible mapping approaches of the system to represent system elements and connections, e.g.: Actor maps – covering which individuals and/or organizations are key players in the space and how they are connected; Mind-maps – which highlight various trends in the external environment that influence issues; Issue maps – covering the political, social, or economic issues affecting a given geography or constituency (often used by advocacy groups), and Causal-loop diagrams – interrogating the feedback loops (positive and negative) that lead to system behaviour or functioning.

Mind-mapping or conceptual models are regarded as an integral part of summarizing complex relationships within a system, what social scientists may call 'wicked problems' (Rittel & Webber, 1973). They aim to present graphically (visually) the main components (nodes) of a system, the pathways (vectors) linking those components and thus the nature of changes in one part of the system having repercussions for other parts of the system. While these are often qualitative diagrams, they form the basis of a quantitative assessment and thus future mathematical descriptions and predictive models; for example, conceptual models combined with Bayesian Belief Network Modelling can create descriptive and predictive systems. The boxes or nodes in a model may be regarded as the structural elements whereas the arrows (vectors) connecting these are functioning (rate processes) or cause-and-effect relationships.

Owing to their inevitable complexity they have often been referred to as 'horrendograms' which may be regarded as being of greater value for the constructor than the reader (e.g., see example below, schematic from Elliott et al., 2014). It is often more illustrative to present individual parts to the reader than to present the whole, for example to create a composite diagram showing all features and hence the complexity, but then deconstruct this in order to discuss, explain and tackle the individual parts. As a related technique, the use of 'Rich Pictures', a branch of Soft System Methodology (Avison et al., 1992), is increasingly used both as a tool for communicating complex ideas to stakeholders and the public but also for determining, interrogating and summarizing those ideas in a pictorial/semi-pictorial form. Similarly, there is an increasing amount of software to allow complex problem visualization leading to solving a problem, for example bCisive (http://bcisive.austhink.com/).

Argument mapping is a method of producing a diagram of an argument, in the form of reasoning, inferences, debates or cases. Pictures and diagrams (infographics) are thought to be a clearer way to illustrate and understand complex themes, and Davies (2010) gives a summary of research and development into the visual representation of information, while Farnham Street (2019) focusses on mind mapping and conceptual mapping across the natural and social sciences. There are various systems of mapping, such as concept mapping, mind mapping and argument mapping, which are sometimes used interchangeably but which Davies (2010) differentiates between; argument mapping allows the display of inferential connections and evaluation of the structure and basis of the argument.

B. Application of the tool: Conceptual models are usually created merely using drawing packages (in WORD or Powerpoint) although there are more sophisticated techniques. Various software packages are available for computer-aided argument mapping (CAAM), e.g., KUMU (https://kumu.io/). See below example schematic using software Rational (after Davies, 2010) demonstrating the structure of an argument map; at the top, there is a contention, followed by a supporting claim and an objection. Further claims, objections and rebuttals follow these, and the terminal boxes require evidence. A further example (below) of argument mapping is using the software bCisive™ to keep track of arguments used in environmental case-making. It allows all the threads of a case to be tracked and provides an audit-trail for any environmental position adopted in the power station planning process of how it has been used, type of assessment it has been used for and, to what level.

C. Tool requirements: The data requirements are low in preparing the conceptual models but they require either a high individual or collective knowledge. They can address single chains of cause-consequence-response or they can address cumulative effects/impacts. They do not explicitly consider uncertainty (potentially giving a false sense of precision). They either do not produce spatially-dependent outputs or temporal trends or at least give these in general terms for a particular area. They can be designed to apply both to data rich/skill rich and data poor/skills poor areas.

D. Key example References or Resources: Avison et al., 1992; Broszeit et al., 2019; Davies, 2010; DePiper et al., 2021.; Elliott, 2012, 2014; Farnham Street, 2019; Gray and Elliott, 2009; McLusky and Elliott, 2004; Twardy, 2003.

S3.02. Semi-quantitative mental models – Fuzzy Cognitive mapping

A. Description of the methodological approach: Mental models are another name for a conceptual model (see above for conceptual models). Both consist of a graphical representation of a system (e.g., natural ecosystem, socio-economic system, socio-ecological system). Because these systems are so complex, these models reflect simplifications that can be useful for improving common understanding and communication tool, particularly when working with stakeholders, helping to identify common understanding, goals and objectives, and highlighting differences in perspective, values and priorities. The main difference with this approach is that linkages are not just documented, but the direction and strength of interaction should be specified, allowing for simple scenario investigation. Perhaps the most used Fuzzy Cognitive Mapping (FCM) tool is Mental Modeler (Gray et al., 2012, 2013a, 2013b) which is freely available at: https://www.mentalmodeler.com/ (with registration). Another type of qualitative mathematical model in this category would be the sign-directed graph model (Dambacher et al., 2009 and 2015).

B. Application of the tool: As outlined for conceptual models, this process can be extremely useful for identifying what elements are relevant/should be included/prioritised in an otherwise extremely complex system. It highlights which elements are related to each other, and how they are connected. It is an incredibly useful first step in processes such as Integrated Ecosystem Assessments, where it can be useful to identify key questions to be answered by such an assessment, and to build up the socio-economic understanding around the identified ecological issues.

C. Tool requirements: There are no data requirements. This extremely adaptable tool can be built by an individual or a group, and completely sourced from their innate knowledge, similarly, what results will be influenced by who contributes, and limited by their knowledge (although data can be used to inform decisions if/when relevant). It can also be used to enhance understanding around a specific issue (e.g., core elements can be specified a priori) and for a range of uses, purposes and potential outcomes (see WKCCMM report; ICES 2022a). They are limited in the outputs (e.g., if scenarios are investigated), and it does not allow self-looping, nor calculate uncertainty. Thus, FCM represents knowledge by defining three characteristics of a system: the components of the system, the positive or negative relationships between the components, and the degree of influence that one component can have on another, defined using qualitative weightings (e.g., high, medium, or low influence).

Almost no skills are needed to operate the tool – it is a simple click and place interface which can be quickly mastered in an afternoon. However, skills are needed to facilitate discussion, manage group interactions, and focus the exercise on the task at hand (including keeping the model manageable!). Additionally, outcomes rely on the level of expert judgement or data available to quantify relationships (see Lazăr et al. 2018).

D. Key example References or Resources: Gray et al., 2012, 2013a, b; ICES, 2020, 2022a; Lazar et al., 2018

S3.03. Knowledge graph

A. Description of the methodological approach: a knowledge graph is a structured representation of knowledge that encapsulates information on entities, their attributes, and the relationships between them. It consists of 'nodes' (representing entities) and 'edges' (representing the relationships between them), and can be visualized as a network or a graph. It is a type of information source that is used by search engines and other applications to provide more relevant and accurate information to users. A knowledge graph might be usefully viewed as a combination of a graphical network diagram, allied to a database providing information on each of the nodes and links. Knowledge graphs are useful in a variety of applications (e.g., search engines, recommendation systems, virtual assistants). By capturing the relationships between entities, knowledge graphs can provide a more comprehensive understanding of a topic, and enable more sophisticated reasoning and decision-making. There are many options for software to develop and apply a Knowledge graph approach – e.g., Neo4j, TigerGraph, GraphDB – with a variety of subscription levels (and costs) dependent on the user's specific needs.

B. Application of the tool: A causal loop network (CLN) is a type of system dynamics model that represents the feedback loops and causal relationships between variables in a complex system, and a CLN-based approach is often used to understand the behaviour of complex systems (e.g., social, economic, or environmental systems). A knowledge graph can be used to represent the entities and relationships within a system, including the variables and feedback loops that are modelled as part of a causal loop network (CLN). Hence, by representing the knowledge about the system in a structured and standardized format, a knowledge graph can facilitate the development of a CLN by providing a clear and comprehensive view of the system's components and relationships. For example, a knowledge graph could be used to represent the entities and relationships in a cause-effect-consequence chain, such as the causal links between human Activities, the resulting Pressures that are generated and that cause State changes in the environmental system and consequential Impacts (on societal welfare) – i.e. an A-P-S-I(W) chain. Such a knowledge graph could then be used to develop a detailed CLN model representing the feedback loops and causal relationships between these entities, facilitating analysis of the system's behaviour under different scenarios.

Knowledge graphs are commonly used across a variety of IT system applications, including internet search engines, recommendation systems, chatbots, and virtual assistants. By capturing the relationships between entities, knowledge graphs can provide a comprehensive understanding of a topic, and so enable more sophisticated reasoning and decision-making. The knowledge graph approach is primarily a means of organising and presenting information, and is not a statistical model per se. In this sense, whilst a knowledge graph approach may provide valuable insights into an environmental system's behaviour (for example, supporting a CLN analysis), it does not represent a means by which a parameterised analytical or predictive model of aspects of the system might be developed. Knowledge graphs have been used in environmental science for data integration, biodiversity assessment, and environmental monitoring (Page, 2016; Babalou et al., 2022). Knowledge graphs can provide a powerful tool for organizing and integrating diverse environmental data sources, enabling more comprehensive analysis and understanding of complex environmental systems, but are not applicable as parameterised analytical or predictive models.

C. Tool requirements: A knowledge graph application is, by its nature, data-rich and therefore places a considerable demand on data availability. However, the knowledge graph approach is able to accommodate a wide variety and combination of data types (quantitative numerical, qualitative, graphical) that apply to each of the entities (nodes) that make up its structure.

D. Key example References or Resources: Page, 2016; Penev et al., 2019; Sachs et al., 2019; Babalou et al., 2022

S3.04. Bayesian belief networks

A. Description of the methodological approach: Bayesian Belief Networks (BBNs) are models that graphically and probabilistically represent correlative and causal relationships among variables and which account for uncertainty (McCann et al. 2006). Hence, BBNs are based on two structural model components: (1) a directed acyclic graph (DAG) that denotes dependencies and independencies between the model's variables (referred to as nodes); and (2) conditional probability tables or distributions (CPTs/CPDs) denoting the strengths of the links in the graph. The DAG consists of a structured set of variables or nodes that represent the modeled system. Directed arrows that are often designed to represent cause effect relations between the system's variables indicate the statistical dependencies between the different nodes. Each arrow starts in a parent node and ends in a child node. The absence of a link between two variables indicates statistical independence between them. The graph is acyclic and therefore no feedback paths from child nodes to parent nodes exist. The DAG can be developed by experts and based on system understanding or can be learned by empirical observation. The resulting BN structure forms the basis for developing operational BNs. Each node has a probability distribution that encodes the current belief in its state; i.e., it shows the uncertainty there is about the value of the variable. The probability distribution can be either continuous or discrete, consisting of several mutually exclusive states, which each have a certain probability of occurrence. Since they denote a probability distribution, the sum of state values makes a total of 1. BBNs allow for the integration of different data types and are a valuable tool for modelling complex systems with many uncertain variables (Marcot and Penman, 2019; McCann et al., 2006). BBNs are probabilistic models that are defined in relation to the problems or question at hand. There are commercial software products like Netica (www.norsys.com), HugIn (www.hugin.com) or GeNIe (https://www.bayesfusion.com/genie/) as well as numerous R libraries such as e.g., bnlearn (www.bnlearn.com).

B. Application of the tool: BBNs have been successfully applied to natural resource management to address environmental management problems and to assess the impact of alternative management measures (Marcot et al., 2019; Coccoli et al., 2018). Thus, BBNs can be used for probabilistic scenario analysis (Hosack et al., 2008; Coll et al., 2019; Kaikkonen et al., 2021; Pihlajamäki et al., 2020). One of their strengths lies in their ability to transparently integrate different types of data sources such as expert judgement, data, or modelling results (Uusitalo, 2007) and even encode expert knowledge into numeric form (Uusitalo et al., 2005). This allows the development of BBN models even in data-poor cases if theoretical understanding of the phenomenon exists. Marine BBN applications have primarily addressed the assessment of alternative management options and choices (Naranjo-Madrigal et al., 2015; van Putten et al., 2013; Rambo et al., 2022) or the risk of not achieving management targets (Stelzenmüller et al., 2010; 2011; Bastardie and Brown, 2021). More recently BBNs have been used to integrate complex models into a decision-support system in the context of evaluation of the attainment of different MSFD goals (eutrophication, Descriptor 5, and fisheries, Descriptor 3) under different scenarios (Uusitalo et al., 2022) or to identify suitable areas for offshore wave energy farms, in the framework of ecosystem approach to marine spatial planning (Maldonado et al., 2022). BBN can easily integrate new knowledge as new evidence becomes available (belief updating) allowing for an evaluation of existing management measures and their adaptations (McCann et al., 2006).

C. Tool requirements: One of the strengths of BBN lies in their ability to transparently integrate different types of data sources such as expert judgement, data, or modelling results (Uusitalo, 2007) and even encode expert knowledge into numeric form (Uusitalo et al., 2005), and different data qualities, as long as there are reasonable estimates of the related uncertainties. BNs can be built based on rich data alone, or scarce data together with domain knowledge. This allows the development of BBN models even in data-poor cases if theoretical understanding of the phenomenon exists. As described above, the quantitative information is expressed in the form of (conditional) probability distributions, reflecting the current level of knowledge about the state of the variable under each set of conditions. Acquiring this uncertainty information may be tricky, but there are possible ways of doing it (Uusitalo et al., 2015). The available commercial BBN packages are easy to use and require only moderate skills to use. Building BBN models that are theoretically solid and describe the phenomenon in the intended way requires some theoretical understanding on BBN modelling.

D. Key example References or Resources: Bastardie and Brown, 2021; Kaikkonen et al., 2021; Marcot and Penman, 2019; McCann et al., 2006; Naranjo-Madrigal et al., 2015; Pihlajamäki et al., 2020; Rambo et al., 2022; Stelzenmüller et al., 2010, 2011; Uusitalo, 2007; Uusitalo et al., 2005, 2015, 2022; van Putten et al., 2013.

S3.05. Risk based approaches exposure-effect-hazard-vulnerability (e.g., Bow-Tie)

A. Description of the methodological approach: Seas and coasts are subject to and may contribute to many hazards, each of which has causes and consequences. In essence, hazards may occur either naturally or by human actions and they become risks when they adversely affect something valued by humans such as health, welfare or property; in some cases, human responses to one hazard may make the consequences even more severe. Hazard is the cause of an adverse effect compared to risk which in contrast is the probability of effect (i.e., the likely consequences) potentially leading to even more severe consequences to humans. Natural risk can be defined as the damage expected from an actual or hypothetical scenario triggered by phenomena or events following natural events (Smith and Petley, 2009). Hazards may be natural or anthropogenic (Elliott et al., 2019) and require to be tackled using a rigorous approach covering technological, governance and economic approaches.

Bow-Tie diagrams (examples below) are a visual tool describing and analysing the pathways of a risk, from hazards to outcomes and reviewing controls (preventative and mitigation/compensation methods, the so-called Programmes of Measures). The approach shows the causes of a problem (to the left of the knot of a Bow-Tie), the hazard and element of main concern (the knot of a Bow-Tie) and the consequences of a hazard happening (to the right of the knot). Various controls can be placed on the left of the hazard to prevent the hazard from occurring, or on the right to reduce/mitigate/compensate for the magnitude of any consequences.

Bow-Tie is an analytical approach for risk assessment and management, which can be adapted for opportunity assessment and management (see Cormier et al., 2019; Elliott et al., 2020b). It is an industry-standard ISO-31000 compliant method for producing conceptual models. It addresses a risk or problem and indicates the causes of that problem, ways to mitigate and prevent these causes, and then consequences that occur because of the problem, and again, ways to prevent or mitigate the consequences. The central knot of the Bow-Tie represents a particular risk (e.g., loss of profits for a specific fishery due to climate change). The left side lists pathways of potential causes whereas the right-side lists consequences (and even opportunities) resulting from the event. Controls (as prevention in a Programme of Measures) are positioned along the pathways of risks on the left (solutions to prevent the central event) and, on the right are placed, other aspects of the Programme of Measures involving mitigation/compensation and recovery from the central event. Escalation factors which undermine or enhance the effectiveness of a given control, can also be added with additional barriers. Hence, the scheme accommodates uncertainty in risk management. The performance of a management control in managing or reducing these uncertainties relies on a suite of barriers that eliminate, avoid, or control the likelihood of a given risk to occur or mitigate or recover from the consequences of a given risk. Barriers implemented closest to the sources of the risk provide the greatest assurance in reducing uncertainty in achieving objectives. At the same time, determining the prevention or response to risk, allows the opportunities to be defined for the sector in question to successfully accommodate impacts and enhance growth.

A key advantage of this method is that the Bow-Tie concept visualises the risks being considered in one, clear and easy to read picture (see examples 1-3 below). Shaped like a bow tie, the diagram creates a clear differentiation between preventative and mitigation/compensation/adaptation measures – effectively ways to prevent an event from happening, and if it does, ways to mitigate any effects; the latter step will also include any compensation, adaptation and control measures. The power of a Bow-Tie analysis is that it shows a summary of numerous risk scenarios, in a single, easy to follow picture that can be understood by all levels of an organisation, as well as the general public. In short, it provides a simple, visual explanation of a risk that would be much more difficult to explain otherwise.

B. Application of the tool: Risk assessment and risk management techniques (for example using the ISO standard Bow-Tie analysis) such as IEC/ISO 31010 (IEC/ISO, 2009) are fundamental to industrial and commercial applications and have only since 2010 been applied to environmental challenges. The environmental manager will be responsible for carrying out operational controls to reach operational outcomes which in turn are required for fulfilling the management objectives and the higher national and international development goals (see diagrams in Cormier et al., 2019). The risk assessment technique of the

ISO 31000 risk management standard (ISO, 2018) is an efficient method well-suited to this role. Here it is emphasised the value to marine environmental managers of the risk management process of ISO 31000 given that an analysis of the measures and actions is needed both to reduce the risks and horizontally to integrate operational controls and conservation measures. Bow-Tie analysis has been used in many industrial applications and recently used in relation to fisheries and aquaculture (Elliott et al., 2020b) and offshore windfarms (Burdon et al., 2018).

C. Tool requirements: Bow-Tie analysis can be carried out either using proprietary software (see Bow-Tie XP and Bow-Tie Master in references, examples 1 and 2) or using a simple WORD drawing or Powerpoint technique; the latter was used as a stakeholder-led process in the CERES project (see Elliott et al., 2020b, example 3). In the latter, generic conceptual models were generated which were then made case-specific using stakeholder engagement.

D. Key example References or Resources: Bowtie Master – https://bowtiemaster.com/?gclid=CjwKCAjwitShBhA6EiwAq3RqA8SiNmfBeYGxchMPGzXZhSWK2iAmqBlZtIfHCbTJFZGmbjznRSkpDBoCRFgQAvD_BwE, Bow-Tie XP - https://www.wolterskluwer.com/en-gb/solutions/enablon/bowtie/bowtiexp, Burdon et al., 2018; Campos et al., 2015; Cormier et al., 2013 (eds); Cormier et al., 2015, 2019; Elliott et al., 2014, 2019, 2020b; IEC/ISO, 2009; ISO, 2018; Smith and Petley, 2009, www.decisionsciences.org/decisionline/vol30/30_3/pom30_3.pd, www.iso14000-iso14001-environmental-management.com/

S3.06. Cumulative impact spatial mapping (e.g., Halpern et al., 2008)

A. Description of the methodological approach: The main features of the global human impact assessment (Halpern et al., 2008) are (i) a grid of selected resolution for all the spatial data, (ii) spatial layers of pressures which are quantified and then normalized between 0-1 inside a grid cell, (iii) spatial layers of ecosystem components (e.g. species, species groups, habitats) which are similarly quantified and then normalized between 0-1 inside a grid cell, and (iv) weight scores representing the sensitivity of the ecosystem components to each of the pressures. Depending on the application, the three scores are summed, or a mean of impacted ecosystem components is taken (e.g., Stock and Micheli, 2018). There are also various ways in determining the weight scores (Halpern et al., 2007; Korpinen and Andersen, 2016).The global method is simple to use, a free software EcoImpactMapper is available to calculate it (Stock, 2016) and it is easily modifiable (Stock and Micheli, 2018). Simplicity, however, also makes it vulnerable to simplifications of reality which was first warned by Halpern and Fujita (2013) and later confirmed by many studies (e.g., Korpinen et al., 2021). While the use of expert judgment in determining the weight scores often receives the most critical comments, it is a relatively robust component (e.g., Korpinen et al., 2012), and the real caveats are in the selection of ecosystem components, data availability, the assumption of linear ecosystem response to pressures and the additive model of cumulative impacts (Halpern and Fujita 2013; Quemmerais-Amice et al., 2020).

B. Application of the tool: Following the first global assessment (Halpern et al., 2008), several regional and pan-European development processes were established and published in a couple of years' time span. The HELCOM holistic assessment in 2010 was the first regional cumulative effect assessment (CEA; HELCOM, 2010; Korpinen et al., 2012), followed by the Mediterranean and the Black Sea (Micheli et al., 2013a) and the North Sea (Andersen and Stock 2013). Korpinen and Andersen (2016) reviewed the use of this (and similar) methods worldwide and recognized the main similarities and differences among the tens of studies. Despite the simplifications, the global CEA method seems to reflect the human impacts, at least in large spatial scales. Using the large-scale biodiversity status of the Baltic Sea sub-basins and the mean cumulative impacts in the same scale, Andersen et al. (2015) were able to show a clear dependency. In the pan-European scale, the Marine Messages II showed a similar dependency of the integrated ecosystem health on cumulative impacts (Reker et al., 2019) and this was also shown for the coastal waters between the ecological status and CEAs (Korpinen et al., 2021; HELCOM, 2023). Research is going on to establish more sophisticated models for cumulative impacts to predict risks for the ecosystem state than the pressure data alone.

C. Tool requirements: The tool requires numerical and spatial datasets, expert knowledge and computational and analytics skills. As such it is a data-rich tool with specialised skills needs.

D. Key example References or Resources: Halpern et al., 2008, 2015; Stock 2016; Stock and Micheli, 2018; Korpinen and Andersen, 2016; Reker et al., 2019; Korpinen et al., 2021; HELCOM, 2023

S3.07. Impact risk assessment through linkage-chain frameworks (e.g., ODEMM)

A. Description of the methodological approach: The EU ODEMM and AQUACROSS projects provide an assessment methodology tracing sector–pressure–ecosystem component pressure pathways (also known as 'linkage chains') and scoring them through expert judgement and data where available. The methodology has been adapted and evolved since in several ways, including for use in ICES Ecosystem Overviews, for use in Integrated Ecosystem Assessment in the Mission Atlantic project, linking to management objectives (such as MSFD descriptors and criteria) and to better account for cumulative impacts (ICES WGCEAM, ICES 2019a). There is much documentation on how to use the tool in the project websites and papers (resources below), but in general the approach consists of two main steps, identifying where linkages exist (mapping in a 'linkage matrix') and then scoring each linkage that does occur for a number of attributes (e.g., spatial overlap, temporal overlap, degree of impact, resilience or resistance, although there are variations on these). The templates are free, most analysis can be carried out in excel, but the Mission Atlantic project has developed an R script with a standardised list of outputs which is freely available: https://github.com/missionatlantic/MissionAtlantic-RISK-Analysis

B. Application of the tool: The benefits of such an approach lie in its comprehensiveness; all relevant sectors and pressures are considered, and the ecological components can be specified at a resolution that is relevant for your purposes. It is very useful for providing context (avoiding tunnel-vision and focusing only on where there is data available), and its key output is a prioritisation of the top risks for the region. The approach is limited in that there are currently few ways to incorporate quantitative knowledge where it does exist, it relies on expert and/or stakeholder judgement (which is fallible and dependent on who is present), and it is often limited to the three components detailed above, limiting the connection to the social system. As such, it is best used in tandem with a few other tools detailed here (e.g., mental models and BBNs) and used to identify key areas of focus for further investigation (as in the case of scoping in IEAs). Recently, Piet et al. (2023) introduced SCAIRM (Spatial Cumulative Impact Risk Assessment for Management) that builds on the ODEMM/AQUACROSS approach (Knights et al., 2015; Piet et al., 2015; Piet et al., 2017; Borgwardt et al., 2019; Culhane et al., 2019a; Piet et al., 2019). The method is modified to address several shortcomings of those approaches. SCAIRM can be used in data-poor situations while allowing the use of available quantitative information in more data-rich situations resulting in an improved capacity to inform policy or guide EBM. Human activities are included at their basic sectoral level, allowing a straightforward link to the socio-economic system and its relevant data, scenarios or stakeholders. In addition, the linkage framework that is at the basis of SCAIRM has been extended by incorporating Ecosystem Services (Piet et al., 2024).

C. Tool requirements: The method can be carried out with nothing but expert/stakeholder knowledge and refined as data is identified/becomes available. However, building the linkage tables can be a time consuming and difficult process. There are several existing assessments (e.g., for four European regional seas: Knights et al., 2015; for Irish sea: Pedreschi et al., 2019) that can be used as a strawman, which often makes progress much faster. Current efforts often result in an initial expert assessment by the assessment team (reaching out to specialists where knowledge is lacking) which is then ground-truthed with stakeholders. This greatly cuts down on the time requirements. It can be carried out with very basic data and analysis skills – however strong facilitation skills and knowledge of the scoring rules are essential to ensure consistency in application.

D. Key example References or Resources: ODEMM: https://odemm.com/ (and references therein), AQUACROSS: https://aquacross.eu/ (and references therein), ICES 2021a (WKTRANSPARENT), WGCEAM Working Group on Cumulative Effects Assessment Approaches in Management: https://www.ices.dk/community/groups/Pages/wgceam.aspx, Mission Atlantic Analysis Script: https://github.com/missionatlantic/MissionAtlantic-RISK-Analysis, Pedreschi et al., 2019, 2023; Knights et al., 2015 ; Piet et al., 2023.

S3.08. Single-species models (e.g., life cycle, stock assessment)

A. Description of the methodological approach: Single-species models are mathematical representations used to study and understand the dynamics of a particular species within an ecosystem. The models focus on the population size, growth, and interactions of a single species, while often considering the species' interactions with its environment and other factors that influence its population dynamics. These models can incorporate limited ecosystem or multispecies information. Examples of types of single-species models are dynamic energy budget (DEB) models, metapopulation models, dynamic population models and individual-based models (IBM).

DEB models are mechanistic models that integrate the dynamics of individual metabolism, simulating energy allocation, growth, reproduction, and other physiological processes based on the principles of mass and energy conservation. These models are extensively used as basic components of complex models of marine ecosystems. Metapopulation models investigate the dynamics of a network of interconnected subpopulations within a larger population and therefore explicitly consider the spatial structure of populations. Dynamic population models describe the changes in size and structure of a single population through time. These models incorporate various factors that influence population dynamics, such as new individuals, mortality, migrations and interactions with the environment. The results of these models are also often used as input for more general models, such as ecosystem end-to-end models. IBMs, or agent-based models, are a population and community modelling approach that accounts for a high degree of complexity or interactions between individuals. IBMs simulate populations or systems of populations as being composed of discrete individual organisms and explore the mechanisms through which population and ecosystem ecology are affected by the way individuals interact with each other and their environment.

A wide range of software packages are available to apply the different types of single-species models. For example, NicheMapR is a package to implement DEB using R (Kearney and Porter, 2020). The Fisheries Library in R is a collection of tools that include Dynamic population models also developed for R (Kell et al, 2007). RAMAS Metapop RAMAS-METAPOP (Applied Biomathematics, Setauket, NY, USA) is a software package specifically designed for studying metapopulation dynamics. Ichthyop is a free Java tool that applies an IBM designed to study the effects of physical and biological factors on ichthyoplankton dynamics (Lett et al., 2008).

B. Application of the tool: Single-species models are useful to understand factors influencing the abundance, distribution, and sustainability of marine organisms, by for example elucidating causal processes underpinning declines of individual species. The models can provide useful information for policy and management, especially for threatened species, keystone species and invasive species (Lindenmayer et al, 2007). Single-species models have also been traditionally used in fisheries management for estimating stock status and fluctuations over time and providing management advice (Burgess et al., 2017).

DEB models have been widely used to analyse and predict the dynamics of marine organisms, and to understand their life history strategies and responses to environmental changes. DEB models can be valuable tools in assessing the potential impacts of environmental change, habitat degradation, or fishing on population dynamics (e.g., Jager et al., 2016; Mangano et al., 2019). Metapopulation models are relevant for species that exhibit spatially structured populations, such as coral reef fish or bird populations that use multiple habitats or breeding sites. By modeling the dynamics of subpopulations in fragmented landscapes, these models can estimate the impacts of habitat loss, fragmentation, or degradation on the overall metapopulation viability (e.g., Guizien et al., 2014; Mari et al., 2021). Dynamic population models are mainly applied to commercially exploited fish species for stock assessment and management purposes (e.g., Szuwalski, 2022). By integrating the available data, projecting future population dynamics, and considering various management scenarios, these models provide insights into the sustainability and productivity of fisheries resources.

IBM are a useful tool for understanding the dynamics of marine ecosystems, studying species interactions, assessing human impacts, exploring management strategies, predicting species responses to environmental change, and integrating diverse data sources (e.g., Marzloff et al., 2009, Xuareb et al., 2021)

C. Tool requirements: Single-species models encompass a large variety of models that differ in the level of complexity and the amount of data required. Among the single species models indicated above IBM have highest computational demands. DEB models need parameterizations based on literature, but there is large database collecting data for hundreds of species (Marques et al., 2018).

D. Key example References or Resources: Burgess et al., 2017; Guizien et al., 2014; Jager et al., 2016; Kearney et al., 2020; Kell et al., 2007; Lett et al., 2008; Lindenmayer et al., 2007; Mangano et al., 2019; Mari et al., 2021; Marques et al., 2018; Marzloff et al., 2009; Stelfox et al., 2020; Szuwalski, 2022; Talbot et al., 2019; Xuareb et al., 2021 ; Dynamic Energy Budget Model in the Sea: http://bioforecasts.science.unimelb.edu.au/app_direct/deb_sea/; Icthyop: https://ichthyop.org/

S3.09. Biogeochemical models

A. Description of the methodological approach: Biogeochemical models capture two-way interactions between the biology and chemistry of ecosystems. They are used to simulate how abiotic and biotic variables interact through time and across space and provide a means to explore management scenarios in relation to climate change and change in the flow of nutrients from land into the ocean. Typically, biogeochemical models are used to study nutrient cycling (nitrogen, phosphorus, oxygen, silicon, and iron) and impacts on planktonic communities due to events such as eutrophication and oxygen depletion.

B. Application of the tool: A range of tools exist, a few key examples are:
The European Regional Seas Ecosystem Model (ERSEM) is a plankton functional type model, developed from a NPZD model, which describes the biogeochemical and plankton cycles (Baretta et al., 1995). BFM is a numerical model designed to study stoichiometric relationships in the biogeochemistry of marine ecosystems by describing the dynamics of major marine biogeochemical processes (Vichi et al., 2007). The model extends and advances the original philosophy of ERSEM. Petihakis et al. (2007) used a high resolution coupled Princeton Ocean Model (providing the physics) and ERSEM (providing the biogeochemical processes) ecosystem model for scenario testing of fisheries management strategies. ECOSMO ('ECOSystem MOdel') is a coupled physical-biogeochemical model (Schrum et al., 2006a, 2006b), with the hydrodynamics based on the HAMSOM ('HAMburg Shelf Ocean Model'; Schrum and Backhaus, 1999), a free-surface 3D baroclinic coupled sea-ice model. The prognostic variables of HAMSOM include temperature, salinity, relative sea surface elevation, 3D-transports, vertical exchange coefficients and turbulent air–sea exchange. BALTSEM, the 'Baltic Sea Long-Term Large-Scale Eutrophication Model', which divides the Baltic Sea into multiple interconnected marine basins each of which is assumed to be horizontally homogeneous but with high vertical resolution (Savchuk et al. 2012).

C. Tool requirements: Biochemical models are data rich tools with various data requirements for biotic and abiotic parameters. Specialised skills and computer power are also required.

D. Key example References or Resources: Baretta et al., 1995; Vichi et al., 2007; Savchuk et al., 2012; Schrum and Backhaus, 1999; Schrum et al., 2006a, b; Butenschön et al., 2016.

S3.10. Food web models (e.g., multispecies models EwE)

A. Description of the methodological approach: Marine Ecosystem Models (MEMs) are of different types and include a variety of assumptions (Tittensor et al., 2018), including as size based, food-web based and individual based processes. Ecosystem models frequently describe the interactions between at least two ecosystem components (e.g., populations, species, functional groups), whereby the interactions are real ecological processes (e.g., predator–prey interactions, mediation, size relationships) and are driven by ecological dynamics, including movement, and perturbations (both natural and anthropogenic). Some of the most used MEMs are food web models, which are often visualized as networks, where nodes denote interacting ecological components, and the causal relationships between them are shown by edges (Geary et al., 2020). Food-web models are a particular type of ecosystem models. They simulate the structure and flow of energy and nutrients between ecosystem components and are commonly used for quantifying food web interactions in a whole-ecosystem context. Generally, for modelling, multiple species are aggregated according to a certain criterion and represent a single species, groups of species or orthogenic phases of a species (juveniles and adults) which are different compartment in the directed network. Food web models aim to the understand the population dynamics among predators and prey, the stability of communities and the implications of change for ecosystem structure, functioning and stability, as well as the flow of matter/energy among the nodes as an ecosystem or ecological network. Food web dynamic models can be used to forecast/hindcast scenarios, and often to describe and understand the current ecosystem. They are often able to account for temporal dynamics and sometimes to incorporate multiple spatial scales, and regularly represent ecosystem components at the level of aggregates/groups of taxa and ecosystem condition/state, but often also at the level of population (size, stage) structure and individual species. They model ecosystem processes such as species interactions as well as various perturbations (Geary et al., 2020).

Examples of such type of models are Ecopath with Ecosim (EwE) and Ecological Network Analysis (ENA). EwE is a whole ecosystem model that quantifies food-web and fishery interactions and can include the impact of environmental change and human activities, including the effect of spatial management options such as the placement of marine protected areas and conservation measures (Christensen and Walters 2004; Heymans et al., 2016). EwE has been widely applied in several marine ecosystem types and regions (Colleter et al., 2015; Stock et al., 2023). The EwE modelling approach and software consists of a suite of three main sub-models (or routines): (i) Ecopath, a static model, representing a mass-balanced snapshot of the food web; (ii) Ecosim, a temporal dynamic model, producing time-dynamic simulations for exploring alternative scenarios (e.g. past and future impacts of fishing and environmental disturbances); and (iii) Ecospace, a spatially-explicit time dynamic 2D model. The last version of Ecospace has been substantially improved with the Habitat Capacity Model, which bring in the capacity to incorporate a niche model within the food web model dynamics (Christensen et al., 2014). This capability has allowed to incorporate a diversity of climate and human activities changing in time and space (de Mutsert et al., 2023).

ENA combines modelling and analysis used to investigate the structure, function, and evolution of ecosystems and other complex systems. ENA is applied to network models that follow the flow of energy or nutrients through the ecosystem by tracing the movement of thermodynamically conserved energy or matter through the system, so that the networks function like resource-distribution maps. Network nodes represent species, functional groups, or non-living resource pools, and the directed edges indicate the transfer of the resources between nodes (e.g., eating, excretion, death). Multiple methods exist to build this type of model, including for example a phenomenological energy or nutrient budget approach, linear inverse modelling methods, dynamic simulation models, and bioenergetics modelling as implemented in EwE. ENA indices can quantify emerging properties of ecosystems and monitor their evolution and offers a quantitative assessment of marine ecosystem functioning.

Food web models may be used to quantify changes in ecosystem indicators, including GES indicators relevant to MSFD Descriptor 3 (D3: Commercially exploited species) and Descriptor 4 (D4: Food webs) (Lynam et al. 2016; Piroddi et al. 2021; Korpinen et al., 2022), and other ecological indicators (Coll and Steenbeek 2017).

B. Application of the tool: The Ecopath mass-balance model can be calibrated using the time dynamic version (Ecosim) by comparing model predictions with data of historical changes in populations and fisheries. Inputs from other models (such as stock assessments) and data sources (e.g., from scientific surveys) are used for the fitting procedure, and outputs from EwE include sea estimates of biomass, production and other ecological indicators for functional groups (ranging from mammals, seabirds, reptiles, fish, invertebrates, seaweeds and primary producers, from both and benthic and pelagic habitats) and ecosystem indicators. The 'EcoBase' model repository, which was developed to gather EwE models published worldwide, includes hundreds of unique models. EwE models have been used to analyse, among others, the ecosystem functioning and the impacts of fisheries, ; trophic functioning in marine systems; the effects of pollution, aquaculture and Marine Protected Areas on a wide variety of ecosystems (including polar regions and terrestrial systems), and the impacts of climate change or cumulative impacts (Colleter et al., 2015; Stock et al., 2023).

Within the scope of GES4SEAS, other specific examples include evaluating the trade-offs among alternative fishing strategies (e.g., discard policy, North Sea, Mediterranean Sea); evaluating relative impacts of fisheries and climate effects (North Sea, Irish Sea, Celtic Sea, Mediterranean Sea); evaluation of closed area management (North Sea, Mediterranean Sea); evaluation of impact of aggregate extraction (Eastern Channel); dynamics of the gadoid and demersal fisheries (West Scotland); feasibility for ecosystem-based management (Clyde Sea). EwE has also been used as part of overarching ensemble models, as developed for example within the Marine Ecosystems Research Programme (MERP) to provide a more holistic and representative assessment of marine ecosystems than could be achieved through the application of a single model or as part of FishMIP the Fisheries and Marine Ecosystem Model Intercomparison Project (Fish-MIP) (Tittensor et al., 2018). In MERP it was used as a West coast of Scotland model, an area covering 110,000 km2, and a Celtic Sea model. The joint research centre of the EU lists EwE as an important tool in its modelling suite.

ENA techniques have been applied to characterize food web organization, assess ecosystem maturity or status, trace biogeochemical cycling in ecosystems, and characterize the sustainability of urban metabolisms and other socio-ecological systems. ENA indices have been proved useful to evaluate the impacts of human pressures on ecosystem functioning and to simulate likely impacts given climatic change scenarios (Lynam et al., 2016). Due to the ability of ENA to characterize the whole ecosystem, and its suitability to answer the need for ecosystem-based management, the use of ENA has been suggested to guide ecosystem assessment and management, including the use of ENA system metrics as GES indicators in the MSFD.

C. Tool requirements: Food web models are data rich tools. They require data from biological surveys and assessments to specify the abundance, productivity of groups and who eats (or catches) whom, and how much. Fishing fleets can be described at any level of detail including landings, discards and economics. Specialised skills and computer power are also required.

D. Key example References or Resources: Borrett et al., 2018; Christensen and Walters, 2004; Colléter et al., 2015; D'Alelio et al., 2016; Fath et al., 2019; Funes et al., 2022; Geary et al., 2020; Korpinen et al., 2022; Lynam et al., 2016; MERP 2019; Nogues et al., 2022; Piroddi et al., 2021; Heymans et al., 2016; Tittensor et al., 2018; Christensen et al., 2014; Stock et al; 2023; De Mutsert et al., 2023; Coll and Steenbeek 2017; https://www.masts.ac.uk/media/4706/ecopath_with_ecosim.pdf ; http://ecopath.org/

S3.11. Ecosystem models (e.g., E2E)

A. Description of the methodological approach: End-to-end (E2E) models are one type of ecosystem models. They are a mathematical representation of an entire ecosystem, a single modelling framework that integrates physico-chemical oceanographic descriptors and organisms ranging from microbes to higher-trophic-level organisms (including humans) and that links to the marine socio-economic aspects. This approach was developed from the need for quantitative tools for Ecosystem-Based Management, particularly models that can deal with bottom-up and top-down controls that operate simultaneously and vary in time and space and that are capable of handling multiple impacts. E2E models are used to describe and understand the current ecosystem and forecast/hindcast scenarios, and often also to make decisions on management actions. They are able to incorporate multiple spatial scales and account for temporally dynamics, and regularly represent ecosystem components at the level of aggregates/groups of taxa and ecosystem condition/state, but often also at the level of population (size, stage) structure and individual species. They model ecosystem processes such as species interactions as well as perturbations (to a single node or to the whole ecosystem) and may also account for dispersal processes. Owing to the complexity of such models, they are generally inappropriate for use as tools for setting tactical management measures such as quotas, as absolute values are far less reliable than the patterns and relative distributions produced (Geary et al., 2020).

Examples of E2E models are Atlantis and STRATH E2E. Atlantis is an E2E ecosystem model that considers all parts of marine ecosystems including the biophysical, economic and social systems (Fulton 2010; Fulton et al., 2011). Originally focused on the biophysical world, Atlantis was further developed to investigate fisheries and more recently for multiple use and climate questions. A summary of Atlantis is provided by Pinnegar (2019) and Geary et al. (2020): Atlantis is a deterministic biogeochemical whole-of-ecosystem model. At the core of Atlantis is a biophysical sub-model that tracks nutrient flows through the biological groups. The primary ecological processes modelled are consumption, production, migration, predation, recruitment, habitat dependency, and mortality. The trophic resolution is typically at the functional group level and the physical environment is spatially represented through a series of irregular polygons. Atlantis includes a detailed exploitation model that is able to examine the impact of pollution, coastal development and broad-scale environmental change. The dynamics of multiple fishing fleets can also be examined each with its own characteristics of gear selectivity, habitat association, targeting, effort allocation, and management structures. Atlantis can be used to assess economic consequences, the result of compliance decisions, exploratory fishing and fishery management instruments including gear restrictions, days at sea, quotas, spatial and temporal zoning, discarding restrictions, size limits, bycatch mitigation, and biomass reference points. The complexity of Atlantis models does make them potentially unwieldy, and they are often used for strategic (what) but not tactical (how) decisions (Peck et al., 2018; https://www.masts.ac.uk/media/4695/atlantis.pdf).

STRATH E2E is geared towards marine ecosystem-based management. The model couples an ecological model with either a fishing fleet model or a fishers' behaviour model, so creating feedback between ecological state and properties of the fishing fleet. The model is designed for application in the North Sea, West of Scotland, Celtic Sea and English Channel. It works on two vertical layers in the water column and two horizontal components (inshore and offshore waters); it uses three inshore and three offshore habitats based on sediment properties. STRATH E2E is ready for use for carnivorous and omnivorous zooplankton and fish, while there is growing confidence in outputs for nutrients, phytoplankton, benthos and seabirds and mammals. Fish are represented by three types, including the external input from fish that migrate in at a set time of year, feeding, growing, being eaten or dying. STRATH E2E includes horizontal transport, vertical mixing, and

sediment resuspension by waves and currents as 'physical' inputs. Suspended and sediment detritus and bacteria are combined as detritus in STRATH E2E, sediment detritus is fixed in each seabed habitat from observational data. The outputs can be processed into other variables such as landings and discards for an individual fleet. Results of a model run can be converted into a range of graphs, spreadsheets, maps etc. At the current time STRATH E2E is best suited to annual projections. This model is also part of a suite of models integrated within an overarching ensemble model as developed within the Marine Ecosystems Research Programme (MERP) to provide a more holistic and representative assessment of marine ecosystems than could be achieved through the application of a single model (MERP, 2019).
E2E modelling which represents large parts of the marine ecosystem by including the most relevant processes in the system, from physics to chemistry, and plankton to fish has been developed with explicit links to MSFD Descriptor 3 (D3: Commercially exploited species) and Descriptor 4 (D4: Food webs) (JRC, 2021; Piroddi et al., 2021). Most often, E2E models are used for strategic insight into system function and the consequences and potential trade-offs associated with different combinations of management strategies, providing information for strategic planning and decision support towards achieving GES (e.g., Bossier et al., 2018).

B. Application of the tool: The Atlantis modelling framework is very flexible and can be applied to any situation world-wide using the many alternative model formulations. Atlantis has been used in 30+ systems round the world, from small estuaries to large ocean regions. Atlantis models have been developed for the North Sea and the English Channel (Pinnegar, 2019). The North Sea model has been used to explore interactions between fisheries, windfarms, and MPAs (calibration of the model is still ongoing). The English Channel model has been coupled to a fishing fleet behaviour model and used to explore interactions between fisheries (French and English) targeting sole and plaice, as well as the influence of riverine inputs. A Baltic implementation of the spatially-explicit E2E Atlantis ecosystem model has also been developed to explore the different pressures (e.g., eutrophication, climate change and fishing pressure) on the marine ecosystem on the medium to long term, and to support strategic management evaluations (Bossier et al., 2018). This links the Baltic Atlantis model to two external models: the HBM-ERGOM, which initialises the Atlantis model with high-resolution physical-chemical-biological and hydrodynamic information while the FISHRENT model analyses the fisheries economics of the output of commercial fish biomass for the Atlantis terminal projection year. The Baltic Atlantis model composes 29 subareas, 9 vertical layers and 30 biological functional groups. The balanced calibration provides realistic levels of biomass for, among others, known stock sizes of top predators and of key fish species. Furthermore, it gives realistic levels of phytoplankton biomass and shows reasonable diet compositions and geographical distribution patterns for the functional groups. Several scenarios of nutrient load reductions on the ecosystem were simulated and the model sensitivity to different fishing pressures tested, allowing to evaluate the impacts on different trophic levels, fish stocks, and fisheries associated with changed benthic oxygen conditions.

Another example is shown by Geary et al. (2020) for the application of Atlantis for a whole-of-ecosystem management strategy evaluation (MSE) in support of a strategic restructuring of southeast Australian federal fisheries (model Atlantis-SE44). Ecosystem-based management solutions were developed and tested for a complex of multispecies and multi-gear fisheries to predict ecosystem-scale responses to the consequences of alternative management scenarios. Strategies focused on different types and combinations of management including alternative quota management, spatial management and gear controls. Different future management scenarios were quantitatively compared, and outputs showed that trade-offs are required for the successful management of such large and complex fisheries, so that various input, output and technical management options may be balanced. MSE outputs were used as a decision-support for understanding potential futures of the ecosystem given different scenarios, rather than providing prescriptive management advice. The model is one of the most complex dynamic ecosystem models for fisheries ever developed, with uncertainty a crucial consideration (including structural uncertainty and human behavioural uncertainty).

STRATH E2E is designed for application in the North Sea, West of Scotland, Celtic Sea and English Channel. Examples of its application include: simulation of fishery yields and maximum sustainable yield (MSY) in relation to the combination of pelagic and demersal harvesting rates (Heath, 2012); simulation of trophic cascades and the sensitivity to top-down and bottom-up drivers (fishing and river nutrient inputs (Heath et al., 2014a); sensitivity of fishery yields to environmental drivers and biological parameters (Morris et al., 2014); cascading trophic effects of scenarios for implementing a discard ban (Heath et al., 2014b).

C. Tool requirements: Ecosystem models are parametrized using field-collected, experimental and/or expert-elicited data to make inferences about specific components (for example, individual species), the entire ecosystem, or even a large part of the coupled socio-ecological system. As such they are data rich tools. Parameterisation of ATLANTIS models requires significant effort including many iterations of validation, calibration and data manipulation. Larger E2E models allow to account for feedbacks as well as multiple ecosystem states and scales but are computationally expensive. They need considerable effort, data and computing power to be implemented. Highly specialised skills and training are also required for users.

D. Key example References or Resources: Bossier et al., 2018; Fulton 2010; Fulton et al., 2011; Geary et al., 2020; Heath, 2012; Heath and Steele, 2009; Heath et al., 2014a, b; JRC, 2021; MERP, 2019; Morris et al., 2014; Peck et al., 2018; Pinnegar, 2019; Piroddi et al., 2021; http://www.mathstat.strath.ac.uk/outreach/e2e; graphical representation of a generic Atlantis model with specifications of examples of models for UK shelf areas (from https://www.masts.ac.uk/media/4695/atlantis.pdf)

S3.12. Habitat suitability models/species distribution models

A. Description of the methodological approach: Habitat suitability models (HSM) are predictive models used to predict the spatial distribution of species based on their observed relationship with environmental conditions. These are also referred in the literature as species distribution models (SDM) or predictive habitat distribution models (Guisan and Zimmermann, 2000). On calibration of the model, the species-environment relationship is established based on data on the species distribution and the environmental conditions where it occurs. It is used to identify the key environmental predictors which determine the suitability of a location for the species. On prediction, the environmental conditions are used as predictors of the likelihood of presence or density of the species throughout a study area. As such, this modelling approach can be seen as an operational application of the potential ecological niche. Different techniques can be applied for the modelling (e.g., generalized linear or additive regression models, classification and regression trees, Random Forest, maximum entropy algorithm), which may rely solely on presence/absence, presence or on density data from surveys. Several of these techniques may be applied using the open-source programming language R, but other statistical software may be available for the individual techniques. A Geographic Information System (GIS) interface is very useful to extract environmental variables from relevant data layers for the model calibration, and to spatialise the resulting model predictions.

B. Application of the tool: HSM are commonly applied to individual species, to identify areas with environmental characteristics that may support the species as a whole, or sensitive stages or processes of the species' life cycle (e.g., nursery habitats for juveniles, spawning grounds). Examples of this application are the mapping of Essential Fish Habitats for fish and shellfish species in the marine environment. Habitat suitability models have also been used to identify geographical regions suitable for different cetacean species, seabirds, and elasmobranchs. These models may be applied to identify potential important marine areas where to prioritise conservation, restoration or to support spatial planning and project level assessment (e.g., by highlighting possible sensitive areas during the environmental impact assessment of marine developments). When applied to multiple species, these models can be used to identify coexisting species and to characterize interaction networks.

C. Tool requirements: The application of HSM is data-rich and therefore places a considerable demand on data availability. To calibrate HSM, monitoring data are required recording either the presence/absence or the abundance (e.g., as density) of a species at survey locations. Where specific life stages are modelled, additional data on body size, age or life stage are needed to differentiate the presence or abundance of the selected life stage from the rest of the population. Environmental variables (e.g., depth temperature, salinity, habitat type, etc.) at the sample locations are also required, either measured during the survey, or derived from correspondent environmental spatial layers. The wider the environmental ranges covered by the survey, the wider is the validity of application of the model hence its spatial coverage. Specialised skills and computer power are also required.

D. Key example References or Resources: Aires et al., 2014; Chavez-Rosales *et al.,* 2019; Fabbrizzi et al., 2020, 2023; Franco et al., 2022; Frederiksen et al., 2013; González-Irusta and Wright, 2016a, b; Guisan and Zimmermann, 2000; Hirzel and Le Lay, 2008; Katara et al., 2021; Lauria et al., 2015; Oppel et al., 2012; Ramírez-León et al., 2021.

S3.13. Natural capital accounting, ecosystem services valuation

A. Description of the methodological approach: The natural capital approach to policy and decision-making considers the value of the natural environment for people and the economy, providing a tool to support the protection and management of the natural environment and to facilitate the engagement of stakeholders within management decisions. The UN System of Environmental-Economic Accounting Ecosystem Accounting (SEEA EA) is the globally adopted statistical standard on ecosystem accounts used to map, assess and achieve good condition of ecosystems (Vallecillo et al., 2022). The reference framework is used to assess the condition of EU ecosystems, so they continue delivering societal benefits, as required by the EU Biodiversity Strategy for 2030. SEEA EA is a spatially based, integrated statistical framework for organizing biophysical information about ecosystems, measuring ecosystem services, tracking changes in ecosystem extent and condition, valuing ecosystem services and assets and linking this information to measures of economic and human activity (United Nations et al., 2021). The SEEA EA is built on five core accounts (see example below) which are compiled using spatially explicit data and information about the functions of ecosystem assets and the ecosystem services they produce.

B. Application of the tool: Natural capital accounts are developed to assess and monitor the contribution of natural resources to economic activity. Physical accounts tables provide basic information on the state of the environment (the stock and the flows of the natural capital, analogous to ecological structure and functioning) in a specific geographical area. When a condition table is also populated, this information can indicate at what level of the ecosystem an impact of economic activities is occurring. However, given the complexity of the marine environment, and the fact that in the SEEA EA only the societal benefits for which a market value exists are reported, for policy and project decision making there might be the need to go beyond SEEA EA accounts. This is currently being addressed in the Horizon Europe MARBEFES project. Examples of applications of natural capital accounting include: the study by Blazquez (2021) which provides preliminary results and first estimates of natural capital accounts for the NE Atlantic area, the work associated with the first marine natural capital accounts for the United Kingdom by the UK Office for National Statistics (ONS, 2021a, b) and Grilli et al. (2021) which reviews the limitations, opportunities and lessons learned from the United Kingdom experience..

C. Tool requirements: The following data are required to populate the five ecosystem accounts: 1. ECOSYSTEM EXTENT accounts record the total area of each ecosystem, classified by type within a specified area (ecosystem accounting area). Ecosystem extent accounts are measured over time in ecosystem accounting areas (e.g., nation, province, river basin, protected area, etc.) by ecosystem type, thus illustrating the changes in extent from one ecosystem type to another over the accounting period; 2. ECOSYSTEM CONDITION accounts record the condition of ecosystem assets in terms of selected characteristics at specific points in time. Over time, they record the changes to their condition and provide valuable information on the health of ecosystems; 3. and 4. ECOSYSTEM SERVICES flow accounts (physical and monetary) record the supply of ecosystem services by ecosystem assets in physical units and the use of those services by economic units, including households, in monetary units; and 5. MONETARY ECOSYSTEM ASSET accounts record information on stocks and changes in stocks (additions and reductions) of ecosystem assets in monetary units. This includes accounting for ecosystem degradation and enhancement. These are data rich approaches requiring specialised skills.

D. Key example References or Resources: Blazquez, 2021; Gacutan et al., 2022a, b, Grilli et al., 2021; ONS, 2021a, b; United Nations et al., 2021; Vallecillo et al., 2022.

S3.14. Bioeconomic models, socioeconomic models, CBA, societal goods and benefits valuation

A. Description of the methodological approach:

Bio-economic modelling**:** Bio-economic models are integrated economic-ecological models. Among the important fields of application for bio-economic modelling are resource management and sustainable resource use. For example, bioeconomic theory and models in fisheries combine the biological and economic aspects of a fishery to explain stock, catch, and effort dynamics under different regimes, and provide insights on the optimal management of the stock (Christensen et al., 2011, Sola et al., 2020).

Valuation of Societal Benefits**:** Since marine ecosystem services have the potential to lead to benefits for society it is appropriate to consider and determine their value (Atkins et al., 2011). Value can be defined in terms of ecological value, economic value, and socio-cultural value (MA, 2003). The concept of 'total social value', which comprises these three definitions, can be used to incorporate value preferences of society associated with natural capital into the decision-making process to inform policy options and management measures (see figure below). Whilst ecological valuation does not directly contribute to total social value, its contribution is indirect in that it provides the basis for both assessments of economic value and socio-cultural value (Burdon et al., 2018).

Cost Benefit Analysis**:** Cost-benefit analysis (CBA) is a core tool of public policy. The systematic process of calculating the benefits and costs, expressed in monetary units, of policy options and projects is now widely regarded as an essential step in the policy process as it helps decision makers to have a clear picture of how society would fare under a range of policy options for achieving goals (OECD, 2018). Environmental CBA is the application of CBA to projects or policies that have the deliberate aim of environmental improvement or actions that somehow affect the natural environment as an indirect consequence. In the UK, the Green Book (HM Treasury, 2022) offers detailed guidance issued by HM Treasury on how to appraise public policies, programmes and projects, including those involving the natural environment.

B. Application of the tool:

Bio-economic data: The data required for bio-economic models is dependent on the focus of the model, but in general requires the input of both ecological and economic data. For example, Bartellings et al. (2015) present the Spatial Integrated bio-economic Model for FISHeries (SIMFISH). In this model fishers' behaviour is simulated based on optimal effort allocation. The added value of this model compared to other existing spatial management tools lies in the presence of (i) short- and long-term fishers' behaviour (ii) spatial explicit stock and fleet dynamics and (iii) relatively low data requirements. As an illustration, SIMFISH is applied in this paper to estimate the impact of area closures in the North Sea. In a second example, Cisse et al. (2013) developed and applied a bio-economic model for the ecosystem-based management of the coastal fishery in French Guiana.

Valuation data can be used to monitor changes in the provision of ecosystem services, and their associated societal benefits, over time. Such assessments can be used to assess natural changes in the system or the impact of existing or proposed management interventions (e.g., the designation of an extended MPA network – Hussain et al., 2010). Valuation data can be incorporated into natural capital accounts (e.g., ONS, 2021a - see Table X15), can be used to support Economic Impacts Assessments (e.g. in the case of the designation of new Marine Conservation Zones – Fletcher et al., 2012) or can feed into Cost Benefit Analyses e.g. Cooper et al. (2013) assessed the costs and benefits associated with seabed restoration following the cessation of dredging activities in the North Sea and Davis et al. (2019) estimated the economic benefits and costs of highly-protected marine protected areas.

C. Tool requirements:

Bio-Economic Models: In the case of the fisheries models, these require data on biological, economic and transversal variables (catch and effort). Biological data requirements include (age structured) assessment outputs and biological parameters (such as natural mortality, maturity and weight at age). Economic data requirements include costs and earnings at the fleet segment level. Transversal variables should match both the biological and economic levels of disaggregation. Currently there is typically a mismatch across the different data sources.

Valuation of Societal Benefits: Regarding economic valuation, for some marine ecosystem benefits market prices may reflect their value (e.g., fish landed for human consumption), but for others a market price either does not exist (e.g., spiritual and cultural well-being) or does not reflect the social value of that benefit. It is not appropriate to value basic marine processes without identifying explicitly the associated ecosystem services and societal benefits which have human welfare implications (Turner et al., 2015). Therefore, valuation focusses on societal benefits only, to avoid double counting of values from natural capital and/or ecosystem services. A suite of economic valuation methods, including market and non-market approaches, are available which can be applied to value the flow and changes in the flow of ecosystem services (see example below). Primary data collection can be costly with respect to time and resources. Therefore, where valuation data are not available for a specific location/region, management decisions may need to be based upon value (or benefit) transfer methods. This approach uses primary valuation research results from one area (a study site) to make secondary predictions about values at a different area (the policy site) (Atkins et al., 2013). There is currently a paucity of valuation data for the broad range of societal benefits associated with the marine environment. Much of the valuation data available within the literature relates to a limited number of societal benefits, largely those which have widely recognised unit prices such as food provision (e.g., Cooper et al., 2013), tourism and nature watching (e.g., Luisetti et al., 2011), and carbon sequestration resulting in a healthy climate (e.g., Luisetti et al., 2019; Watson et al., 2020). Such valuation data are often obtained at the local level through case studies, with few such studies collecting time series data, which is valuable for identifying

changes in benefit delivery over time or collecting data at the national level. The ensuing data gaps make valuing the marine environment challenging.
Cost Benefit Analysis: The approach to the monetary valuation of costs and benefits includes assessment based on opportunity costs (defined by the value which reflects the best alternative use a good or service could be put to), and valuation may include data based on market prices and non-market monetary valuation where market prices are not available. Data on all relevant costs and benefits implies a requirement for data on a range of variable including those associated with natural capital, health and risks to life. Significant benefits and risks that are beyond direct monetisation can be considered by comparing alternative scenarios with and without their inclusion, which can be used to reveal their costs. This informs choice by considering whether these cost differences are worth paying.
All three are data and skills rich tools/approaches.

D. Key example References or Resources: Atkins et al., 2011, 2013; Bartelings et al., 2015; Burdon et al., 2018, 2022; Cissé et al., 2013; Davis et al., 2019; Fletcher et al., 2012; HM Treasury, 2022; Hussain et al., 2010; Luisetti et al., 2011, 2019; OECD, 2018; MA, 2003; Turner et al. 2015; Watson et al., 2020.

S3.15. Spatial planning models (e.g., GIS, VAPEM)
A. Description of the methodological approach: Spatial planning models, adapted to marine realm, are tools used to help planners and policymakers make informed decisions about the use of marine space and resources. The models are designed to provide insights into the potential impacts of different planning scenarios, and to help identify the most effective strategies for achieving specific planning goals. There are several different types of spatial planning models, each of which is suited to different types of planning challenges. Overall, spatial planning models are valuable tools for helping planners and policymakers make informed decisions about how to manage and develop the built environment in a way that promotes social, economic, and environmental sustainability. They are used to assess the impact of different land-use and development scenarios on environmental quality and natural resources. They can help planners identify areas of the city or region that are most vulnerable to pollution, habitat loss, or other environmental hazards, and evaluate the effectiveness of different conservation strategies.

Geographic Information Systems (GIS) are computer-based tools used to store, analyse, and visualize spatial or geographic data. GIS systems allow users to capture, manipulate, analyse, and present geographic data in a variety of ways, including as maps, charts, and 3D models. GIS applications can be used for a wide range of purposes, including environmental monitoring and management, urban planning, natural resource management, emergency response, and market analysis, among others. For example, a GIS system could be used to map the distribution of pollutants in an area, or to analyse the impact of proposed infrastructure projects on the natural environment. GIS systems are powerful tools that allow users to integrate and analyse data from a variety of sources to gain insights into complex geographic phenomena. They have become an essential tool for decision-makers in a wide range of issues.

VAPEM tool (Ecological Assessment and Marine Spatial Planning Tool) is an open access web-based decision support tool (https://aztidata.es/vapem/) which integrates the environmental risk information (through WEC-ERA tool information, Galparsoro et al., 2021) together with the technical and socio-ecological information linked to wave energy projects, into a Bayesian model (Galparsoro et al., 2022b). The environmental dimension of the model is based on the integration of 16 pressure types and 27 ecosystem elements according to the MSFD.

B. Application of the tool: The VAPEM tool permits the user the exploration of predefined scenarios or the generation of scenarios by the user. It provides information on the feasibility of wave energy projects development under different technical, environmental, and socio-economic conditions. The outcome of the assessment is also provided as a spatially explicit feasibility map. VAPEM tool is designed especially for managers and decision makers, but also for the industry or other kind of stakeholders, to inform about different options of development of wave energy projects under MSP framework. More details in Maldonado et al. (2022).

C. Tool requirements: Technical information: seafloor score, wave resource, weather window, distance to the nearest port, and distance to the nearest electric substation. Environmental information: pressures produced by the wave converters, sensitivity of the ecosystem elements to the pressures, the spatial distribution of the ecosystem elements, the impact risk for each ecosystem element and the overall ecological risk. Socioeconomic information: activities that make use of marine space and that may be incompatible or cause limitations to the development of wave energy farms. VAPEM is a rich data tool with many types of data layers needed (as detailed above) requiring a range of skills (e.g., in GIS and analytical skills).

D. Key example References or Resources: VAPEM tool: https://aztidata.es/vapem/; Maldonado et al., 2022.

S3.16. Systematic conservation planning models (e.g., MARXAN, ZONATION)

A. Description of the methodological approach: Conservation planning is "the process of locating, configuring, implementing and maintaining areas that are managed to promote the persistence of biodiversity and other natural values" (Pressey et al., 2007). Systematic conservation planning (SCP) has six distinctive characteristics (Margules and Pressey, 2000): (1) requirement for clear choices about the ecological features that will be used as surrogates for overall biodiversity; (2) it is based on clear goals, translated into operational targets; (3) it recognizes the level of achievement of conservation goals in the existing protected areas; (4) it uses an explicit and transparent methodology to locate and design new reserves complementing the existing network of protected areas; (5) it applies explicit criteria for implementing conservation actions; and (6) it adopts explicit objectives and mechanisms for maintaining the needed conditions in protected areas required to foster the persistence of the protected ecological features, as well as adequate monitoring schemes and adaptive management. SCP can implicitly account for the spatial variability of human uses and the associated cost of excluding human activities for the sake of protection, ideally expressed in monetary values (Naidoo et al., 2006). Hence, SCP approaches can account for trade offs between conservation and socio-economic priorities. The importance of SCP approaches for marine spatial prioritization in Europe has been repeatedly highlighted by experts (e.g., Giakoumi et al., 2012; Metcalfe et al., 2013; Mazor et al., 2014; Katsanevakis et al., 2020; Gimenez et al., 2020).

Decision support tools have been developed to facilitate systematic conservation planning, the most widely used being MARXAN and ZONATION. MARXAN attempts to solve the minimum set problem, i.e., design networks of protected areas that capture a set amount of biodiversity for the least cost (Ball et al., 2009). On the other hand, ZONATION attempts to solve the maximum coverage problem, i.e., design networks of protected areas that capture as much biodiversity as possible for a fixed budget/cost (Moilanen et al., 2009). This means the approaches underpinning MARXAN and Zonation are fundamentally different; nevertheless, comparisons between the two tools have not detected very different results (Delavenne et al., 2012). MARXAN is a suite of spatial prioritization decision support tools that includes free software that can be used to solve conservation planning problems (https://marxansolutions.org/). The suite includes various variants of MARXAN (e.g., MARXAN with Zones, MARXAN with Connectivity) focusing on different applications. MARXAN uses a simulated annealing algorithm to find a suite of good near-optimal systems of priority areas that meet conservation targets while attempting to minimize socio-economic costs. MARXAN with Zones allows any parcel of land or sea to be allocated to a specific zone, not just reserved or unreserved. Then, each zone has discrete management actions, objectives, and constraints, with the flexibility to define the contribution of each zone to achieve targets for pre-specified features (e.g., species or habitats). The objective is to minimize the total cost of implementing the zoning plan while ensuring a set of conservation objectives are achieved. In 2018, through the Biodiversity and Protected Areas Management programme (BIOPAMA), the Joint Research Centre working with The Nature Conservancy managed to prototype a web-based MARXAN platform that improves accessibility to non-experts. In 2020, MARXAN developers partnered with Microsoft to bring the MARXAN platform to the cloud, scaling its infrastructure for global accessibility. MARXAN is currently a key application featured on Microsoft's Planetary Computer.

Zonation (https://zonationteam.github.io/Zonation5/) is a freely available decision support software tool, operating on spatial data about ecological features, costs, and threats, also utilizing information about uncertainty and connectivity. It is versatile, including a broad set of methods and analyses that can be used to address different types of spatial prioritization problems and analytical needs (Arponen et al., 2007; Moilanen et al., 2009). The spatial dynamic module of EwE, Ecospace, includes an optimization routine which incorporates Marxan's principles linking them to the spatial prediction capabilities of Ecospace (Christensen et al., 2009). Ecospace modelling capabilities are also being used to assess the effects of different management options, such as the placement of MPAs and other management interventions (Gomei et al., 2021; de Mutsert et al., 2023).

B. Application of the tool: SCP tools can be used in various ways, such as to design networks of marine protected areas to reach specific spatial targets, complement existing MPA networks with additional sites, create zones within an existing or expanded network of MPAs, prioritize management actions, assess and improve the connectivity of an existing network of MPAs, integrate terrestrial-freshwater-marine conservation planning, assess trade offs between ecological and socioeconomic priorities in maritime spatial planning, assess the cost-efficiency of an existing or proposed network of MPAs, and assess the outcomes of different scenarios affecting goals and operational targets. Such tools are very relevant for the implementation of the spatial targets of the Biodiversity Strategy for 2030 (30% protection – 10% strict protection).

Selected examples:
Giakoumi et al., 2012: MARXAN was used in the Greek Ionian Sea to identify priority areas for conservation of coastal and offshore biodiversity, under different scenarios, taking into account socio-economic factors expressed as a single cost metric, weighting different economic sectors in proportion to their contribution to the GDP of the region. The results were compared with the existing Natura 2000 sites in terms of goal achievement, area requirements, and cost. Existing Natura 2000 sites were found to fail in achieving conservation goals for some EU priority and other important coastal and offshore habitats and species.

Doxa et al., 2022: A framework for 4D conservation planning was developed, where the 3rd dimension is depth, and the 4th dimension is time (climate-driven predicted changes in the distribution of biodiversity). Spatial prioritization of marine areas in the Mediterranean Sea was based on both ecological and climatic criteria, considering the present and future distribution of >2000 benthic and pelagic species, using ZONATION.

Hermoso et al., 2021: An approach was developed to identify priority areas for conservation across three different realms (freshwater–terrestrial, estuary and marine) for multiple species, including species that inhabit or move across the three realms and accounting for different types of connectivity (longitudinal connectivity along rivers or multidimensional connectivity in the estuary and marine realms, guided by currents and depth similarity), using MARXAN.

Evans et al., 2015: MARXAN coupled with a biophysical habitat map was used to investigate representative MPA network scenarios and to assess the efficiency and representativeness of the existing High Seas MPA network in the Northeast Atlantic.

C. Tool requirements: Data needed include:
- Spatial distribution of ecological features (habitats, species, geomorphological features, other surrogates of biodiversity), i.e., GIS layers.
- Spatial distribution of human uses, e.g., fishing effort, shipping, aquaculture, UW cables or pipes, offshore exploration, wind farms, port facilities, coastal infrastructure etc.
- Spatial distribution of existing management measures, sectoral or regional management plans, e.g., Fisheries Restricted Areas, OECMs, management plan for aquaculture, management plan for tourism, etc.
- Valuation of human uses or surrogates to estimate opportunity costs (i.e., loss of revenue by restricting human uses).
- Data on connectivity (e.g., based on oceanographic models, species molecular analyses, tagging etc.).
- Vulnerability data (linking the impact of human activities to ecological components).

Data rich but several approaches can be applied by using surrogates to cover data gaps. Specialised skills needed: GIS, use of SCP tools such as MARXAN or ZONATION, analytical skills.

D. Key example References or Resources: Arponen et al., 2007; Ball et al., 2009; Delavenne et al., 2012; Doxa et al., 2022; Evans et al., 2015; Fabbrizzi et al., 2023; Giakoumi et al., 2012; Hermoso et al., 2021; Katsanevakis et al., 2020; Margules and Pressey, 2000; Mazor et al., 2014; Metcalfe et al., 2013; Moilanen et al., 2009; Naidoo et al., 2006; Pressey et al., 2007.

S3.17. Simple assessment index (e.g., M-AMBI)

A. Description of the methodological approach: Simple assessment indices are methods for classifying marine systems according to human pressures and include those focusing on the primary community structural variables (abundance, species richness, and biomass) and derived community structural variables (such as diversity indices, abundance and biomass ratios, and evenness indices) (Gray and Elliott, 2009). These indices

can also include functional analyses such as those involving feeding guilds (as in the Infaunal Trophic Index (ITI)) and their responses to organic matter inputs or other human pressures (such as the so-called 'biotic indices'). There are dozens, if not hundreds, of biotic indices, covering different biological elements, which include phytoplankton, zooplankton, macroalgae, seagrasses, macroinvertebrates and fishes (Borja et al., 2023).
Among the assessing biotic indices, those using several metrics (e.g., richness, diversity, proportion of opportunistic/sensitive species, etc.) and combined in different ways (e.g., multimetric, multivariate) have been very successful, especially after the Water Framework Directive requirements (Birk et al., 2012). One of the most successful indices, used for macroinvertebrates, is the AZTI's Marine Biotic Index (AMBI) and its derivative Multivariate AMBI (M-AMBI), which have given place to multiple adaptations, changes, new indices and even applications to other biological elements, such as foraminifera or bacteria (Borja et al., 2019). M-AMBI is a multivariate index, which includes richness, diversity and AMBI, and is calculated comparing monitoring data and reference conditions of high and bad status, using a factor analysis (Muxika et al., 2007). The result of the calculation is a value between 0 (worst status) and 1 (best status), being the quality status classification divided into five quality classes, from bad to high. The thresholds between the different quality classes, and especially those between moderate and good, have been intercalibrated and agreed among the European countries, not only for this index, but also for all used in the different regional seas (EC, 2018).

B. Application of the tool: M-AMBI was developed to be used primarily in the ecological status assessment within the Water Framework Directive, where it is intercalibrated with other European indices (EC, 2018). In addition, it has been used in assessing the environmental status within the Marine Strategy Framework Directive, both in Descriptor 5 (eutrophication) and Descriptor 6 (seafloor integrity). Also, it has been used worldwide in assessing the impacts from different human pressures. Recently, it has been adopted by the US Environmental Protection Agency (EPA) as the official method to assess the status of all estuaries and coasts in USA (US EPA, 2021a, b). It has been used to assess the impacts produced by many different human pressures (e.g., 15 in the case of Borja et al., 2015), or in a meta-analysis of 834 references, including 6 human pressures, 11 biogeographical areas (covering from poles to equator), and 26 responsive variables, including nutrients, metals and organic compounds (Borja et al., 2019).

C. Tool requirements: for the index calculation only the abundance or biomass per species and station/replicate is needed (also, it can be calculated at other taxonomical levels, e.g., genus, family). The most important is to set reference conditions for each type to be assessed i.e., community, defined by salinity, depth and sediment grain size. There is a free software tool allowing the calculation (https://ambi.azti.es). It can be calculated with any number of stations, but it is true that a minimum number of 50 sampling stations is recommended, to reduce uncertainty (Borja et al., 2008)

D. Key example References or Resources: Borja et al., 2008, 2015, 2019, 2023; EC, 2018; Muxika et al., 2007.

S3.18. Descriptor or theme specific combination of indices and models (e.g., HEAT, BEAT and CHASE)
A. Description of the methodological approach: The indicator-based multi-metric assessment tools integrate quantitative indicators to inform of the integrated state of the assessment area. Tools have been developed for hazardous substances (CHASE), eutrophication (HEAT), biodiversity (BEAT). The indicators use a threshold value indicating acceptable state from the un-acceptable state and the indicators can be grouped within the tool to best reflect the assessment topic. For example, in the integrated biodiversity assessment, habitat-related indicators are assessed together while species population indicators are assessed in another group of indicators. The integration of the indicators is on two levels: (1) indicators within a group are first integrated by weighted averaging to derive a state of the group and then (2) each of the groups are compared: the group with the worst state is selected to represent the integrated state of the area, following the precautionary principle. As the tools function with relatively simple integration techniques, the major effort is put into using representative indicators, reliable thresholds and ecologically meaningful grouping of indicators. The tools also include a confidence assessment which informs of any shortcomings in meeting adequate standards in data, indicator representativeness or threshold setting.

B. Application of the tool: The hazardous substances (CHASE), eutrophication (HEAT), biodiversity (BEAT) tools were developed first for the Baltic Sea region but have expanded to the NE Atlantic (HEAT and CHASE), Western Mediterranean Sea (HEAT), Black Sea (HEAT/BEAST) and the European Environment Agency's pan-European state assessments. The eutrophication tool has been indirectly accepted by the European

Commission as a tool to be used for the descriptor 5 assessment under the MSFD. The D5 assessment guidance by the EC was developed by taking into account the HEAT requirements. While the Descriptor 8 assessment (contaminants) requires substance-specific results, the CHASE tool can assist in comparing the substances by using the tool's Contaminant Ratios (CR) as well as in presenting overall status of the areas. The BEAT tool is not directly applicable to the MSFD biodiversity assessments (descriptors 1, 3, 4 and 6) but is a good communication tool to illustrate the complicated multi-species/habitat results in a simple manner over large areas.

C. Tool requirements: The HEAT, BEAT and CHASE assessment tools can be applied to any spatial level where data exists. HELCOM HOLAS III applies the HEAT tool into large assessment units (18 Baltic Sea sub-basins) where the underlying indicator data is aggregated over the assessment area whereas EEA is using the same tool in a grid where each grid cell calculates the HEAT result. The use of monitoring data which may restrict the spatial resolution of the assessment, but this has been recently improved by adding earth observation data. In principle, also modelled data could be used, but that should be validated against the indicator and its threshold. EEA has applied HEAT, BEAT and CHASE on a European-wide scale. The HEAT and CHASE tools have been used as analytical tools, e.g., focusing on long time series (Andersen et al., 2017) or on distance-to-target (Fleming-Lehtinen et al., 2015; see example below).

D. Key example References or Resources: Andersen et al., 2011, 2014, 2016 a, b and 2017; Fleming-Lehtinen et al., 2015; Nygård et al. 2018;

S3.19. Overarching assessment tools (e.g., NEAT and OHI)

A. Description of the methodological approach: NEAT stands for "Nested Environmental status Assessment Tool". It is a tool primarily designed for (marine) environmental assessment but can in principle be used for other types of assessment too. The tool by Berg et al. (2018), in the context of DEVOTES, an EU funded project NEAT has a clean implementation of indicator aggregation through a hierarchy that is based on Spatial Assessment Units (SAUs). Due to a process of indicator normalization, NEAT can aggregate various indicators in such manner that all indicators within that SAU equals the same weight, unless indicated the opposite. Thus, NEAT avoids a bias from an uneven number of indicators per certain ecosystem component. The method is available with the free NEAT software, which is a desktop software for Windows and MacOS (current version 1.4). https://www.azti.es/en/productos/neat/

B. Application of the tool: The tool is useful for environmental assessments as it has been designed for exactly that purpose. Usually, it is applied to marine environmental status assessment, and it allows describing the environmental status for the whole assessed area but it can also be visualized for the different spatial scales from the pre-defined SAU hierarchy, MSFD descriptors, ecosystem components (e.g., fish, phytoplankton, etc.), or habitat types. To reach this output, it uses state indicators that are assigned to a SAU, a habitat, a descriptor, and an ecosystem component. Each indicator has a unique value (and a standard error associated with it) for the MRUs for which information is available. In addition, each indicator requires defining the target value as well as the upper and lower thresholds. Prior to the aggregation of indicators within and across the different MRUs, indicators are normalized on a scale from 0 to 1, setting the boundary between no good and good environmental status at 0.6. The assessment outputs are accompanied with confidence assessment values, which are based on Monte Carlo permutations.

It was first applied to 10 case studies across the European Seas (Uusitalo, 2016), ranging from smaller parts of e.g., the Dutch North Sea to larger areas such as the Norwegian Barents Sea. Subsequently, the method was applied to the Caspian Sea (Nemati et al., 2017, 2018) where impacted and non-impacted sites were compared and pressure indicators included into the assessment. Other applications include the Black Sea (Marin et al., 2020), Mediterranean Sea (Pavlidou et al., 2019, application in Greece covering 8 descriptors and 24 indicators), Malta (Borja et al., 2021, application covering seven descriptors), deep areas of the Atlantic (Kazanidis et al., 2020, application in 9 study areas in the North Atlantic focusing on five MSFD descriptors with 24 indicators), the Bay of Biscay (Menchaca et al., 2022; with 7 descriptors and 67 indicators) and in 26 Mediterranean MPAs (Fraschetti et al., 2022, application with an extensive dataset across five Mediterranean ecoregions).

C. Tool requirements: The central principle in the NEAT method is a hierarchical, nested structure of spatial assessment units (called SAU) and habitats. These must be defined to run the assessment. The order of these hierarchies is such that the assessment begins with the hierarchically nested SAUs. For each of these, one or more habitats are assigned. These habitats are themselves nested and hierarchically structured so an indicator can be assigned to one individual habitat or to more than one habitat comprising all habitats from the hierarchical levels below. Each of the resulting SAU/habitat combinations is then used to assign the various indicators to. These indicators are typically designed around specific species or communities or functional groups, which spatially are in their own habitats, within a specific SAU. The method requires to know the value of each of the indicators and their corresponding assessment class boundaries (between high, good, moderate, poor and bad status) per SAU and habitats (these may vary). To run the assessment, the obtained indicator values are entered along with their standard error, so confidence assessments can also be obtained.

D. Key example References or Resources: Berg et al., 2018; Borja et al., 2021; Fraschetti et al., 2022, Kazanidis et al., 2020; Marin et al., 2020; Menchaca et al., 2022; Nemati et al., 2017, 2018; Pavlidou et al., 2019; Uusitalo et al. 2016

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