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Integrated methods for marine ecosystem physical accounting

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ABSTRACT

Studies assessing marine physical stock accounts compliant with the globally adopted System of Environmental-Economic Accounting - Ecosystem Accounting are limited. These studies typically focus on ecosystems easy to map, and the fundamental linkage between ecosystem extent and condition and ecosystem services is rarely considered. Moreover, methodologies assessing marine ecosystem condition are diverse, but there is no standardised framework yet. In this study, we have successfully integrated an ecological valuation approach — the Ecological Value Assessment — with an established habitat classification system — EUNIS — to monitor marine ecosystem extent and condition in three European Atlantic regions. Results obtained in this study reveal that ecosystems' condition spatial patterns vary depending on the context and are driven by specific ecosystem components. Additionally, this study has proven that our methodology has the potential to track ecosystem extent and condition over time, enable the direct connection of both extent and condition to ecosystem services and potentially be standardised to support national and international accounting efforts in oceans. The integrated method proposed in this study can operate with limited data and is easily adaptable to other coastal and marine regions, fostering its reusability.

1. Introduction

Coasts and oceans include some of the world's most valuable and productive ecosystems (Costanza et al., 1997, 2014), providing essential resources (Cooley et al., 2022). Ecosystems such as seagrasses, saltmarshes and mangroves contribute to human well-being with a variety of ecosystem services, including coastal protection (Lopez-Arias et al., 2023; Maza et al., 2021; Spalding et al., 2014), carbon sequestration (Inoue et al., 2019; Nellemann et al., 2009) and food provisioning (Millennium Ecosystem Assessment, 2005; Luisetti et al., 2011). Unfortunately, like most terrestrial and marine ecosystems, their distribution and extent have generally decreased due to land use changes, ecological degradation and climate change impacts (Goldberg et al., 2020; Ondiviela et al., 2014; Orth et al., 2006). For example, a global mangrove decline of 2.1% was estimated between 2000 and 2016, at an average annual rate of 0.13% (Goldberg et al., 2020). Airoldi and Beck (2007) estimated that over 50% of European saltmarsh habitats have been lost by coastal development alone. Waycott et al. (2009) estimated an extinction of 29% of known seagrasses since 1879. Due to these alarming losses of coastal and marine ecosystems, in recent years, emphasis has been placed on protecting and sustainably managing these ecosystems, as well as determining their value. One approach to valuing these ecosystems, both in economic and non-economic terms, is through the concept of natural capital (Costanza et al., 1997).

Natural capital, defined by the European Investment Bank as "the value of everything that comes from nature — soil, air, water and all living creatures" (European Investment Bank, 2023), is a well-known in economics, but it has recently emerged as a useful concept to inform

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policy decisions for better management on a systematic basis (Ruijs et al., 2019) and emphasizes the role of nature in supporting human welfare and its economy (Burdon et al., 2024). Whilst the ecosystem services framework focuses on the flow of benefits that humans gain from natural capital, natural capital focuses on the management of the natural environmental components (i.e., natural assets) that generate the flow of ecosystem services, goods and benefits and related income (Barbier, 2011; Dasgupta, 2021). Identification and quantification of indicators for marine natural capital and ecosystem services allows changes in their delivery to be monitored spatially and temporally (Hattam et al., 2015). Recently, many countries have examined integrating natural capital measures into their economic and physical accounts (Virto et al., 2018). The globally adopted statistical standard on ecosystem accounts is the System of Environmental-Economic Accounting - Ecosystem Accounting (hereinafter SEEA-EA), which constitutes an integrated and comprehensive statistical framework for organizing data about ecosystems and is in use by more than 50 countries worldwide (Vallecillo Rodriguez et al., 2022; United Nations, 2025). The SEEA-EA is a spatially based, integrated statistical framework for organizing bio-physical information on 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, 2024). Ecosystem accounts in SEEA-EA are divided into two subgroups: stock accounts and flow accounts; in turn, these can be either physical or monetary accounts (Fig. 1).

Physical stock accounts are composed of ecosystem extent and ecosystem condition accounts (La Notte et al., 2022). However, the fundamental linkage between the extent and condition of ecosystems and the ecosystem services that they deliver (physical flow accounts) has received relatively little attention (United Nations, 2021), with limited studies integrating both components, such as Vallecillo et al. (2020), who successfully linked ecosystem condition with riverine flood control ecosystem service, and Polce et al. (2013), who did so with pollination services. Thus, physical flow accounts have generally been assessed based on ecosystem extent rather than a combination of the ecosystem's extent and its condition. Nevertheless, ecosystem services not only depend on extent, but ecosystem condition also plays a crucial role in supporting ecosystem service provision (Rapport et al., 1998). Declines in ecosystem condition have been frequently related to reduced ecosystem service delivery [e.g. Hou et al., 2023; La Notte et al., 2022]. Therefore, the integration of the interdependencies of ecosystem services with both ecosystem extent and condition is critical for a correct monitoring of ecosystem state changes due to ecosystem loss and degradation.

In relation to studies assessing physical stock accounts in the oceanic realm, the literature estimating Ocean Physical Stock Accounts (hereinafter OPSA) remains limited, and there is no standardized framework for assessing both the ocean's ecosystem extent and condition (Cummins et al., 2023), although the Global Ocean Accounting Partnership Technical Guidance provides some initial indications (Global Ocean Accounting Partnership, 2021). Furthermore, OPSA studies are geographically and methodologically diverse, and the adopted frameworks vary widely. Certain studies have assessed both ecosystem extent and condition (Australian Bureau of Statistics, 2017; Department of Environment et al., 2016; Institute for Development of Environmental-Economic Accounting Group and Synthesis report, 2020; Indian Ocean Commision, 2014; Office for National Statistics, 2021; Schenau et al., 2019), whilst others have focused solely on assessing ecosystem extent (Carnell et al., 2019). Not surprisingly, 95% of the estimates related to open sea and oceans from the Ecosystem Service Valuation Database (Brander et al., 2025) lack ecosystem condition assessment (Herná et al., 2022).

Ecosystem extent assessments are commonly based on international standard classification systems, including IUCN GET (Institute for Development of Environmental-Economic Accounting Group and Synthesis report, 2020; United Nations, 2017) or European Nature Information System (EUNIS) (Office for National Statistics, 2021), as suggested by the SEEA-EA (United Nations, 2024). Additionally, within marine and ocean contexts, assessments typically focus on ecosystems that are easy to map, such as nearshore or intertidal areas, but OPSA literature that includes pelagic and deep-sea ecosystems remains limited mainly due to the limited availability of deep-sea ecosystem data (Cummins et al., 2023). This data limitation also entails that changes in both the extent and condition of marine ecosystems are poorly implemented. However, when looking at the assessment of ecosystem condition, it is observed that a variety of studies have used different approaches to assess condition in oceans (Bruzó et al., 2023; Madden

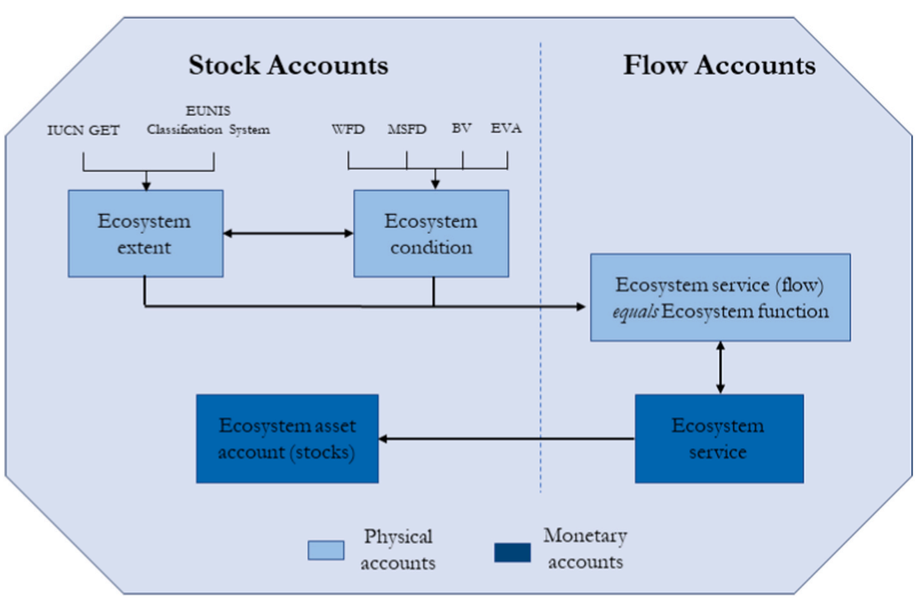

Fig. 1. SEEA-EA accounting framework. Subgroups of ecosystem accounts and the interconnections between SEEA-EA, classification systems, and condition indicators. (adapted from SEEA). (SEEA-EA: System of Environmental-Economic Accounting - Ecosystem Accounting; IUCN GET: IUCN Global Ecosystem Typology; EUNIS: European Nature Information System; WFD: Water Framework Directive; MSFD: Marine Strategy Framework Directive; BV: Biological Valuation; EVA: Ecological Value Assessment).

et al., 2009; Watson et al., 2022). Whilst some studies have often used study-specific condition indicators (Institute for Development of Environmental-Economic Accounting Group and Synthesis report, 2020; Madden et al., 2009; Dvarskas, 2019), others have performed more standard methodological approaches based on supranational, national or subnational rules (Office for National Statistics, 2021; Watson et al., 2022; Weatherdon et al., 2018). Both these approaches are still restricted to those contexts and their specific characteristics, making them difficult to replicate or adapt to other contexts. Nevertheless, a variety of supranational rules and methodologies related to ecosystem condition such as the Water Framework Directive (WFD) (European Union, 2000) and the Marine Strategy Framework Directive (MSFD) (European Union, 2008), the Biological Valuation (Derous et al., 2007a, 2007b) and the Ecological Value Assessment (hereinafter EVA) (Franco et al., unpublished results) emerged as promising tools to overcome these challenges.

The WFD establishes a framework for the protection of all water bodies (including marine waters up to one nautical mile from shore) by 2015. The Directive's aim is for all water bodies in EU member states to achieve "good ecological status", whilst the MSFD aims to achieve or maintain 'good environmental status' in European marine regions. Both directives are founded on an initial assessment of the current ecological/environmental status of their waters and a socio-economic analysis of human activities in these waters. To perform this assessment, baseline conditions (i.e., reference values) need to be determined by member states and these have also been considered for determining how WFD "ecological status" and/or MSFD "environmental status", and other indicators of ecosystem condition (state or quality), can be coupled with habitat extent information to deliver a more precise locally-tailored natural capital approach (Watson et al., 2022).

The Biological Valuation (Derous et al., 2007a, 2007b) has also shown to be a promising tool for assessing ecosystem properties in marine areas, including both biodiversity and structural aspects of ecosystems with a flexible method. Biological Valuation has been applied in several contexts to determine the areas with the highest intrinsic biological value. It has been successfully applied in the North Sea (Derous, 2007), the Polish Exclusive Economic Zone (Węsł et al., 2009), the Basque continental shelf (Pascual et al., 2011), the Belgian coastline (Vanden Eede et al., 2014) and the Portuguese continental shelf (Gomes et al., 2018). Biological Valuation has also been demonstrated to be applicable for other purposes such as guidance for conservation and management plans (Węsławski et al., 2009; Vanden Eede et al., 2014; Gomes et al., 2018), ecological indicators (Derous, 2007; Pascual et al., 2011) and determining the status of marine waters in the context of the MSFD (Mamede et al., 2023; MAMAOT, 2012).

A recent adaptation of this Biological Valuation methodology is the EVA (Franco et al., unpublished results). Like Biological Valuation, EVA facilitates the spatial evaluation of ecosystem properties and allows for the comparative analysis of ecological value (EV) among subzones within the study area and can be applied from intertidal to deep-sea areas. Although the terms EV and biological value are equivalent in the literature, EV and EVA account for the broader applicability of the approach to ecosystem components encompassing not only biological ones (e.g., species groups; as considered in Biological Valuation) but also habitats. Habitats are central components of ecosystems supporting numerous ecosystem services. Consequently, the EVA methodology aligns better than Biological Valuation with accounting frameworks.

This study proposes an integrated methodological approach (Section 2.1) for coupling marine ecosystem extent and condition towards physical accounting assessments. The methodology integrates the EUNIS ocean habitat classification system with the EVA approach to effectively link ecosystem extent with ecosystem condition and enable tracking of ocean physical stock account changes in oceanic contexts. The proposed integrated methodological approach has been tested at three European Atlantic case study areas (Section 2.2), each with different ecological and biological contexts.

2. Methodology

2.1. Integrated methodological approach

In this section, the integrated methodological approach is explained. We combine the EUNIS marine habitat classification, as suggested by the SEEA-EA, with the EVA approach to assess OPSA. The approach comprises two steps: Step 1) Ecological Value Assessment (EVA); Step 2) Ocean Physical Stock Accounts (OPSA) compilation (Fig. 2).

Step 1: Ecological Value Assessment (EVA)

We tested the approach at the three European Atlantic case study areas, with minor adaptations depending on the case study. Step 1 varied in each case study due to: the diverse use of data sources depending on the type of evidence (e.g., field data, modelling products) and the relevance of an ecosystem component in each case study (e.g., angiosperms might not be a relevant ecosystem component for a case study if they do not occur in the area or the region). We performed the Ecological Value Assessment at the case study areas based on Franco et al. (unpublished results).

Ecosystem components are functional groups that cluster species and/or habitats that share taxonomic groups, ecosystem functions, and/or spatial characteristics. The EV of the ecosystem components is estimated as the combination of a set of metrics (calculated from available data using defined algorithms to answer multiple assessment questions, AQs) that estimate the presence of ecological processes and biodiversity structure in an area. A total of 15 AQs are identified in EVA (Table 1), expanding the previous Biological Valuation methodology proposed by Derous et al. (2007b). The EVA approach then stipulates that the resulting values estimated for each AQ are aggregated into a single EV for each ecosystem component.

We considered the EV as an indicator of the ecosystem components' inherent condition as it reflects the state or quality of the component, as manifested by its biodiversity, at a certain spatial location and timeframe. As the overall idea of the proposed integrated methodological approach is to test the use of the EVA approach as an inherent condition indicator for physical accounting, we used the term condition instead of EV to homogenize the assessment terms with those of the accounting frameworks. To compile physical accounts, condition indicators must be linked to extent metrics. EUNIS is the adopted marine habitat classification system for the EU, so it is sensible to use EUNIS for extent measurement in our methodology. Since EVA is a broad-scale approach to ecosystem assessment, it includes components linked to marine habitats, such as benthic species, as well as components that are not directly linked to habitats in the classical sense, such as seabirds and mammals. For our methodology, we applied the EVA only to those ecosystem components which can be directly linked to EUNIS habitats (benthos, macroalgae, etc.) (Table 2). The methodological procedures applied in our study differ from the EVA methodology by restricting the assessment to ecosystem components that can be directly linked to EUNIS habitats. Step 1 contains two sub-steps:

Sub-step 1.1: We defined the assessment system including determination of a) the case study spatial area, b) the assessment subzones within the case study area, c) the ecosystem components to be analysed (Table 2), d) the AQs (and associated algorithms) relevant for each ecosystem component (Table 2), and e) the construction of a database with the available data (Table 2).

Sub-step 1.2: We evaluated the ecosystem component condition by calculating and aggregating the AQs into a condition for each ecosystem component. We assessed the associated confidence of each of these ecosystem component condition values as stipulated by the EVA approach. Confidence scores within EVA account for multiple sources of uncertainty, including the type of evidence used (such as expert judgement, modelling products and direct monitoring data) and the data availability (e.g., amount of data informing the assessment).

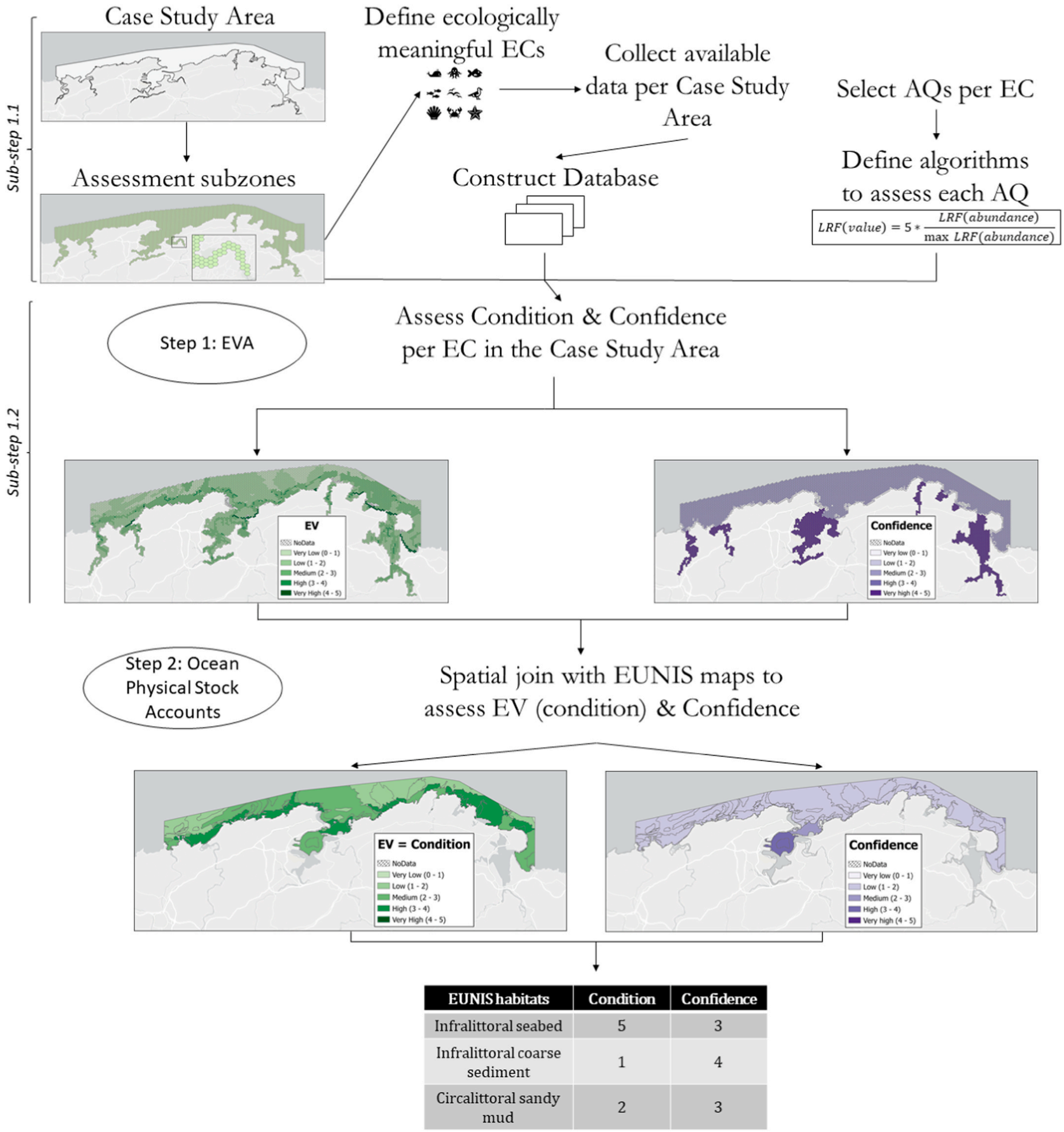

Fig. 2. Proposed integrated methodological approach. The figure represents the workflow of the proposed integrated methodological approach as detailed under Section 2.1 (modified from (Pascual et al., 2011)). (AQ: Assessment Question; EC: Ecosystems Component; EV: Ecological Value; EVA: Ecological Value Assessment; LRF: Locally Rare Features; EUNIS: European Nature Information System).

To determine the assessment subzones within the case study areas (part b under sub-step 1.1), we split the case study spatial area into regular grids (hereinafter subzones) that enable standardisation and comparison between subzones using *ArcGIS Pro* and *QGIS* software. We used variable subzone sizes (Table 2) to ensure ecologically meaningful areas for each ecosystem component (Węsł et al., 2009) based on mobility, habitat patch size and data source spatial resolution criteria. We assigned 3000-m subzones to highly mobile ecosystem components (e.g., demersal fish) that occur over broad spatial ranges. We also applied 3000-m subzones to non-mobile components when the spatial resolution of the source data was too coarse for smaller subzones to provide additional information. In contrast, we used 250-m subzones for immobile or limited mobility ecosystem components supported by high-resolution data (e.g., angiosperms, subtidal macroalgae). We acknowledge that subzone size may influence condition scores through spatial smoothing. Therefore, we selected grid sizes that minimize this bias by

Table 1
AQs used to compute the condition of each subzone and EC in a CS area. AQ: assessment question; EC: ecosystem component; CS: case study; LRF: locally rare features; RRF: regionally rare features; NRF: nationally rare features; FN: feature number; ROF: regularly occurring features; ESF: ecologically significant features; HFS/BH: habitat-forming species/biogenic habitats; MSS: mutualist or symbiotic species (see Supplementary Material B for detailed feature information).

AQ code	Assessment Question (type)	Algorithm	Clarifications
AQ1	Is the subzone characterized by the presence of many LRF? (qualitative)	1) Select LRF for the EC of interest. 2) Give 0 value where LRF is absent and 5 where it is present. 3) Repeat steps 1 and 2 separately for each LRF in the CS area, then average the values across LRFs in every subzone.	LRF are species or habitats occurring in <5% of the subzones in the CS area. In case ECs are naturally restricted to specific depths (e.g., saltmarshes to intertidal areas), the % should be calculated considering only the subzones relevant to the EC (e.g., all subzones covering intertidal areas).
AQ2	Is the subzone characterized by high abundance of many LRF? (quantitative)	1) Select LRF for the EC of interest. 2) Extract the abundance value for the LRF in each subzone (*LRF-abundance*). 3) Assign an *LRF-value* to each subzone by rescaling *LRF-abundance* to values between 0 and 5 using the following formula: *LRF-value = 5 * LRF-abundance/maximum LRF-abundance** 4) Repeat steps 1 to 3 separately for each LRF in the CS area, then average the values across LRFs in every subzone.	* *Maximum LRF-abundance* is measured from the range of abundance values obtained across subzones in the CS area
AQ3	Is the subzone characterized by the presence of many RRF? (qualitative)	Same as AQ1 algorithm but applied to RRF instead of LRF.	RRF are species occurring in <2% of 50 × 50 km squares of each of littoral, sublittoral, or abyssal zones in the region or habitats occurring in <2% of 50 × 50 km squares in the region where the CS area occurs. Alternatively, RRF may be identified as species or habitat identified as regionally rare from accredited national or international lists, or based on expert judgement.
AQ4	Is the subzone characterized by high abundance of many RRF? (quantitative)	Same as AQ2 algorithm but applied to RRF instead of LRF.	
AQ5	Is the subzone characterized by the presence of many NRF? (qualitative)	Same as AQ1 algorithm but applied to NRF instead of LRF.	NRF are species occurring in <0.5% of 10 × 10 km squares within national waters/habitat type restricted to a limited number of locations in territorial waters in which the CS area occurs. Alternatively, NRF may be identified as species or habitat identified as nationally rare from accredited national/international lists, based on expert judgement.
AQ6	Is the subzone characterized by high abundance of many NRF? (quantitative)	Same as AQ2 algorithm but applied to NRF instead of LRF.	
AQ7	Is the FN in the subzone high? (qualitative)	1) Consider all species or habitats for the selected EC in the CS area. 2) Give 0 value where species or habitat is absent and 5 where is present. 3) Average the values across the species/habitats within an EC for the subzone.	
AQ8	Is the subzone characterized by high counts of ROF? (quantitative)	Same as AQ2 algorithm but applied to ROF instead of LRF.	Species/habitats occurring in >5% of the subzones in the CS area.
AQ9	Is the abundance of ROF very high in the subzone? (quantitative)	1) Select ROF for the EC of interest. 2) Determine the mean abundance of the individual ROF for the whole CS area (=X). 3) Calculate the mean abundance of the ROF in every subzone (=Xi). 4) Calculate the ratio Xi/X for the ROF in each subzone. 5) Determine the 5% subzones with the highest ratio. Calculate the percentage of the abundance of the ROF that occurs in these 5% most important subzones (=Y) 6) Determine in how many subzones the ROF occurs (=Z, i.e. the number of subzones where Xi > 0). 7) Calculate the ratio Y/Z which is the aggregation coefficient for the ROF. 8) Multiply the ratio Y/Z with the ratio Xi/X for the ROF (*ROF-unscaled*). 9) Assign a *ROF-value* to each subzone by rescaling *ROF-unscaled* to values between 0 and 5 using the following formula: *ROF-value = 5 * ROF-unscaled/maximum ROF-unscaled**, 10) Repeat steps 1 to 9 separately for each ROF in the CS area, then average the values across ROFs in every subzone.	**Maximum ROF-unscaled* is measured from the range of RSF unscaled values obtained across subzones in the CS area. Leave blank (no ROF-value) where the ROF-value cannot be ascertained.

(*continued on next page*)

Table 1 (*continued*)

AQ code	Assessment Question (type)	Algorithm	Clarifications
AQ10	Is the presence of ESF high in the subzone? (qualitative)	Same as AQ1 algorithm but applied to ESF instead of LRF.	Species or habitats which have a significant ecological role in supporting the ecosystem structure and functioning. E.g. keystone species; species which constitute important food sources of certain seabirds, etc.
AQ11	Is the abundance of ESF high in the subzone? (quantitative)	Same as AQ2 algorithm but applied to ESF instead of LRF.	
AQ12	Is the presence of HFS/BH high in the subzone? (qualitative)	Same as AQ1 algorithm but applied to HFS/BH instead of LRF.	
AQ13	Is the abundance of HFS/BH high in the subzone? (quantitative)	Same as AQ2 algorithm but applied to HFS/BH instead of LRF.	
AQ14	Is the presence of MSS high in the subzone? (qualitative)	Same as AQ1 algorithm but applied to MSS instead of LRF.	
AQ15	Is the abundance of MSS high in the subzone? (quantitative)	Same as AQ2 algorithm but applied to MSS instead of LRF.	

aligning the subzone resolution with the ecological scale of each component, the habitat patch sizes and the spatial resolution of its associated data sources (Table 2).

We translated the AQs into mathematical algorithms to allow for their value calculation per ecosystem component per subzone (Sub-step 1.2). We capped the condition scores obtained from the AQs between 0 and 5 to enable their discretization into five classes — 0-1 (very low), 1-2 (low), 2-3 (medium), 3-4 (high), and 4-5 (very high) — to facilitate the interpretation of the results. Decisions and clarifications for calculating each of the AQs are detailed in Table 1. We calculated final subzone condition scores per ecosystem component using the highest of the AQ scores obtained for that ecosystem component within each subzone. We used the highest AQ score as condition indicator because it allows to reflect the most ecologically relevant attribute expressed in each location.

We then assigned to each subzone a confidence value corresponding to the AQ score used. AQ confidence scores in EVA range between 1 and 5, depending on the data supporting its assessment (Table 3). We calculated AQs' confidence scores as the arithmetic mean of all species or habitats' confidence scores included in the AQs for each ecosystem component. The confidence scores in the EVA do not represent statistical uncertainty of the condition scores. Instead, they should be interpreted as a proxy for evidence strength, reflecting the type and data availability of the supporting evidence for informed decision-making. Consequently, decision-makers should apply the precautionary principle when interpreting low-confidence condition scores to avoid over-reliance on extrapolated data and refrain from using confidence scores as weights in economic valuation.

Step 2: Ocean Physical Stock Accounts (OPSA)

Ocean Physical Stock Accounts comprise the compilation of ocean ecosystem extent and ecosystem condition accounts (La Notte et al., 2022). Step 2 integrates EVA outcomes into EUNIS marine habitat classification (Vasquez et al., 2023) maps (hereinafter EUNIS habitats) to assess ecosystem extent and condition. We used the EUNIS marine habitat classification (version 2012) to map habitat types as polygons in the case study areas. Where an EUNIS habitat class was not assigned, we

Table 2
Summary table of ecosystem components and AQs used in the EVA per case study, including grid size used in the assessment and data sources. (AQ: assessment question; EVA: ecological value assessment; EUNIS: European Nature Information System; ICES: International Council for the Exploration of the Sea; ILVO: Flanders Research Institute for Agriculture, Fisheries and Food).

Case Study	Ecosystem component	Grid size	Data Sources	AQs
Cantabria	Angiosperms	250 m	Field data (2020 - 2023)	AQ3 - AQ13
	Benthic macroinvertebrates		Field data (2019 - 2023) extrapolation using soft-bottom EUNIS habitats (level 3)	AQ7, AQ10, AQ11
	Intertidal macroalgae		Field data (2011) extrapolation using coastal typologies from Ramos et al. (2017)	AQ7 - AQ13
	Subtidal macroalgae		Field data (2017) extrapolation using Maxent model (version 3.3.3k)	AQ7 - AQ13
	Benthic habitats		Estuarine benthic habitats: field maps of habitats of Community Interest (sensu Habitat Directive 92/43/CEE) (2020 - 2023). Marine benthic habitats: modelled distribution of EUNIS benthic habitats (2023).	AQ1 - AQ13
North Sea	Benthic habitats	3000 m	Modelled distribution of EUNIS benthic habitats (2023)	AQ1 - AQ9
	Macrozoobenthos		Field occurrence data (1969 - 2023) from Cooper et al. (2017), ICES Environmental Database (n. d.-a), ILVO and OneBenthic (2020).	AQ1, AQ2, AQ7 - AQ13
Irish Sea	Benthic habitats	250 m	Modelled distribution of EUNIS benthic habitats (2023)	AQ1 - A13
	Macrozoobenthos		Field data (2012 - 2022) from OneBenthic (2020) and Marine Institute (2002).	AQ1 - AQ15
	Demersal fish	3000 m	Field data (2012 - 2021) from ICES Database on Trawl Surveys (n. d.)	AQ1 - AQ11

Table 3
Ecological Value Assessment confidence scoring criteria. (Range: range of the number of samples or observations available to answer each assessment question for each ecosystem component per subzone across the case study; Min.: minimum number of observations; Max.: maximum number of observations; X: (Max. – Min.)/3).

Data support	Data availability	Score
Expert judgement	-	1
Indirect data (e.g. modelling products, extrapolated data)		2
Direct field data	Low: Range (Min., Min. + X)	3
	Medium: Range (Min. + X, Min. +2X)	4
	High: Range (Min. +2X, Max.)	5

used energy, biozone and substrate characteristics to derive habitat classes equivalent to EUNIS (see Supplementary Material A for detailed information on EUNIS habitat classes and codes, as well as those polygons classified using energy, biozone and substrate characteristics).

Integrating EVA with the EUNIS habitats incorporates the oceanic realms linked to EUNIS habitats into the accounting framework, enabling OPSA assessments in European waters. We conducted the spatial integration using *ArcGIS Pro* (version 3.1.0) software following 4 sub-steps.

Sub-step 2.1: To avoid including subzones with data absence in the OPSA, we removed those subzones with no condition data in each ecosystem component.

Sub-step 2.2: We spatially overlaid EUNIS habitat polygons and ecosystem component condition and confidence results obtained from EVA to calculate condition-confidence pairs for each ecosystem component within each EUNIS habitat polygon. To this end, we averaged the scores of all subzones intersecting with each EUNIS habitat polygon. We used the arithmetic mean to ensure that condition scores represent the overall condition of the ecosystem component in the underlying EUNIS habitat polygon, and avoid unbalanced weightings introduced by specific subzones.

Sub-step 2.3: We estimated the total condition and associated confidence of each EUNIS habitat polygon by averaging all condition and confidence scores across ecosystem components. We used the arithmetic mean to aggregate the condition and confidence scores across ecosystem components in the absence of ecological evidence supporting differential weighting. Thus, we considered all ecosystem components to contribute equally to habitat condition. This aggregation logic targets the practicality for accounting and the precautionary principles, ensuring that data-rich ecosystem components do not disproportionately dominate the habitat condition scores. However, alternative aggregation rules, such as 'one-out, all-out' or a weighted arithmetic mean, could change habitat condition results, leading to more pessimistic results or skewed towards data-rich components, respectively.

Sub-step 2.4: Finally, we calculated ecosystem extent and condition by EUNIS habitat type (OPSA compilation) across the entire study area. For this purpose, we computed ecosystem extent as the total area of habitat polygons for each EUNIS habitat type present. We obtained condition and confidence as area-weighted averages of the polygon scores — first for each habitat type and then across all habitat types.

2.2. Case study areas

The proposed methodological approach was tested at three European Atlantic case study areas (Fig. 3). These three areas were selected because they differ markedly in size, represent distinct coastal typologies, ranging from open ocean to semi and quasi-enclosed systems, and are dominated by different habitat types and ecological groups. Data from these study areas were collected to represent key ecosystem components (Table 2), which were assessed using EVA.

The first case study (CS1) is in Cantabria, Spain, in the southern Gulf of Biscay. CS1 covers areas from intertidal to 50-m depth and has an extent of 237 km^2, where EUNIS marine habitat classes (Vasquez et al., 2023) cover 78% of the total extent (185.5 km^2). CS1 is characterized by

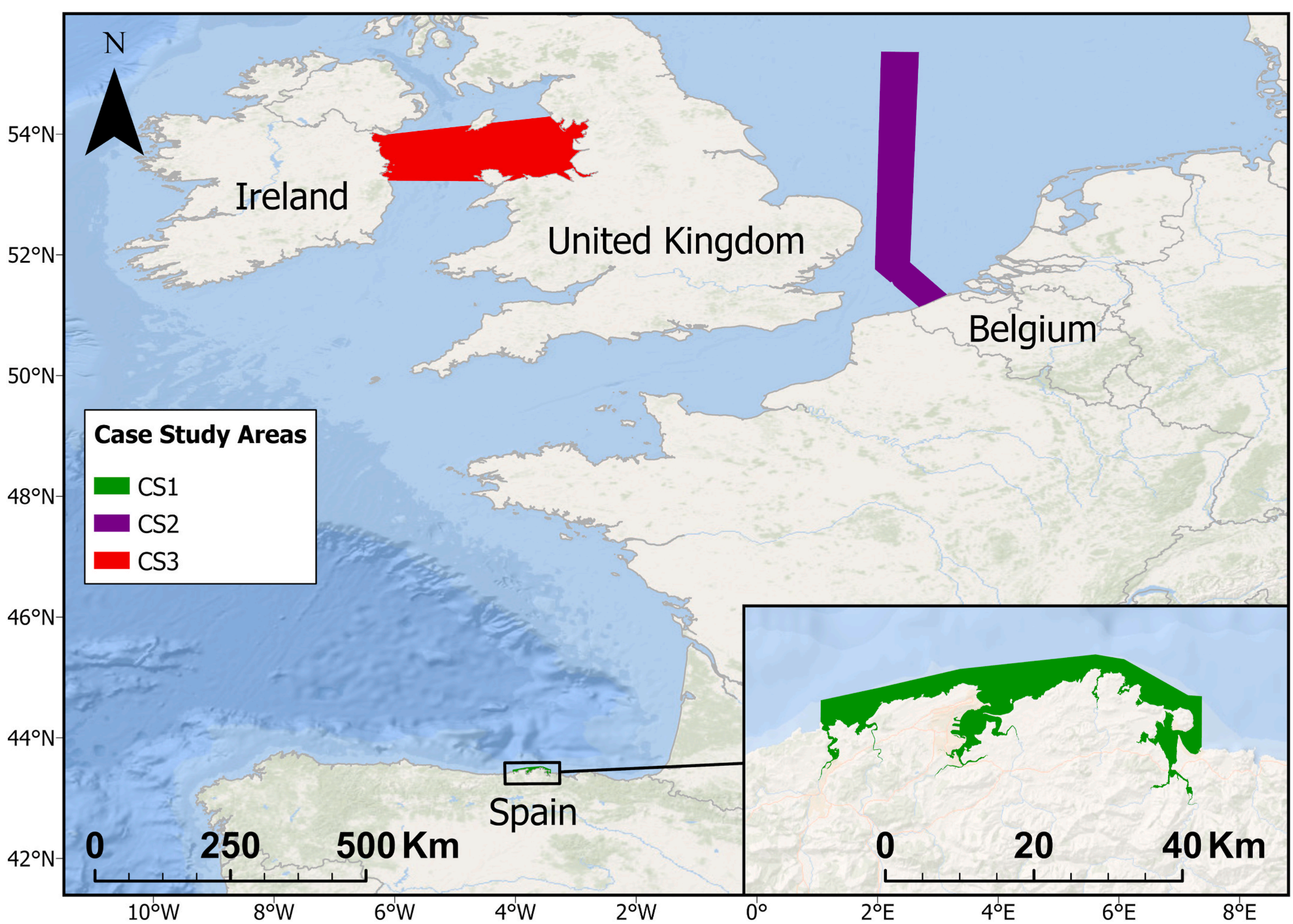

Fig. 3. Case Study Area locations. Map representing the location of the three Case Study Areas: CS1 - Cantabria case study; CS2 - North Sea case study; CS3 - Irish Sea case study.

the presence of coastal hard-bottom and soft-bottom habitats, as well as estuarine ecosystems. Coastal areas are exposed to high-energy waves impacting on extensive cliffs (Seoane et al., 2012), with pocket beaches and small inlets isolated between rocky headlands (Ramos et al., 2016). The estuaries of CS1 are dominated by large intertidal areas (Galvá et al., 2010), which are colonized by seagrasses (*Zostera noltii* and *Zostera marina*) and saltmarshes. Estuaries also harbour rich benthic macroinvertebrate communities (Puente et al., 2008), as well as important commercial fish such as *Anguilla anguilla*, *Solea solea* and *Pegusa lascaris* (Ministerio de Agricultura, Pesca y Alimentación, 2022). Seabird communities include ecologically important species such as *Arenaria interpres*, *Larus sp*. and *Sterna sandvicensis* (SEO/BirdLife, 2020) and rocky substrates in the coast are dominated by different species of macroalgae depending on site characteristics (Ramos et al., 2016; Borja et al., 2004) (see Supplementary Material B & Table 2 for more detailed information on each of the data sources and methodological adaptation decisions taken under CS1).

The second case study (CS2) covers the southwestern part of the North Sea within Belgian and UK waters, with an extension of 19,017 km^2 and EUNIS marine habitat classes (Vasquez et al., 2023) covering almost 100% of the total extension. CS2 is a semi-enclosed sea (Ducrotoy et al., 2000) dominated by shallow waters (<50-m depth) (Cotterill et al., 2017; Verwaest et al., 2022), strong tidal currents and subtidal sandbanks (Verwaest et al., 2022). Dominant benthic ecosystems in CS2 include biogenic reefs constructed by *Lanice conchilega* (Van Hoey et al., 2008), geogenic reefs and sandbanks dominated by species such as *Abra alba*, *Asterias rubens* and *Nephtys hombergii,* among others. CS2 hosts 140 fish species, including important commercial fish species such as *Solea solea* and *Gadus morhua* (Deconinck et al., 2020). CS2 is also an important migratory route for 60 species of seabirds (Vanermen et al., 2020) and a place for marine mammals such as *Halichoerus grypus* and *Phocoena phocoena* (Haelters and Kerckhof, 2024) (see Supplementary Material B & Table 2 for more detailed information on each of the data sources and methodological adaptation decisions taken under CS2).

The third case study (CS3) is placed in the Irish Sea, covering the waters of the UK and Ireland. CS3 has an extension of 20,385 km^2, EUNIS marine habitat classes (Vasquez et al., 2023) covering 98% of the total extent (20,011.06 km^2), and covers areas from intertidal to 100-m depth. CS3 is a quasi-enclosed sea characterized by the presence of the summer gyre, where warm waters rotating around a dome of cold water in the deeper basin retain nutrients and provide a food source for marine wildlife in the western Irish Sea (Hill et al., 1997). Coastal areas of CS3 are dominated by extensive salt marshes, mudflats and dunes as well as rocky shores and seagrass beds (Irish Sea Network, 2022). Ecologically important species present in CS3 include a wide variety of marine mammals (e.g. seals and dolphins), seabirds (e.g. *Sterna dougallii* and *Cepphus grylle*) and fish (e.g. *Engraulis encrasicolus* and *Scomber scombrus*) (see Supplementary Material B & Table 2 for more detailed information on each of the data sources and methodological adaptation decisions taken under CS3).

3. Results

By applying the proposed integrated methodological approach (Section 2.1), we obtained two types of results: 3.1) Ecological Value Assessment (EVA), and 3.2) results from the Ocean Physical Stock Accounts (OPSA) compilation.

3.1. Results from the ecological value assessment (EVA)

During this process, we obtained spatial-explicit condition and confidence scores by ecosystem component, comparable within a case study area.

The condition and confidence scores showed marked variability across ecosystem components and case study areas. In the Cantabria case study, angiosperms scored high condition (3.64 ± 1.64) and very high confidence (4.23 ± 0.00), benthic macroinvertebrates scored medium condition (2.58 ± 0.69) and very low confidence (0.31 ± 0.19), intertidal macroalgae scored very high condition (4.35 ± 0.35) and low confidence (1.42 ± 0.12), subtidal macroalgae scored medium condition (2.40 ± 1.36) and low confidence (1.40 ± 0.11) and benthic habitats scored low condition (1.71 ± 0.91) and high confidence (3.42 ± 1.03). Conversely, in the North Sea, benthic habitats scored high condition (3.21 ± 1.29) and low confidence (1.92 ± 0.55), and benthic macroinvertebrates scored high condition (3.33 ± 1.41) and very low confidence (0.89 ± 0.21). Finally, in the Irish Sea, benthic habitats scored very low condition (0.84 ± 0.19) and medium confidence (2.98 ± 0.10), benthic macroinvertebrates scored medium condition (2.20 ± 1.59) and low confidence (1.96 ± 1.21) and demersal fish scored high condition (3.13 ± 1.17) and medium confidence (2.53 ± 1.20) (see Supplementary Material C for comparable ecosystem component spatial-explicit condition and confidence of the three case study areas).

3.2. Results from the ocean physical stock accounts compilation

During this process, we estimated overall EUNIS habitat extent, condition, and confidence, individual EUNIS habitat condition and confidence maps, as well as ecosystem component condition by habitat class.

3.2.1. Overall condition and confidence values

In terms of condition, Cantabria case study scored overall with a medium value (2.48). More concretely, 50.91% of the extent scored medium condition values, 29.29% low condition values, and 19.70% high condition values (Table 4). North Sea case study scored overall with a high value (3.10). More concretely, 80.59% of the extent scored high condition values and 19.41% medium condition values (Table 5). Irish Sea case study scored overall with a low value (1.86). More concretely, 47.66% of the extent scored medium and 40.76% scored low condition values (Table 6).

Assessment confidence was overall low in the Cantabria and North Sea case study (1.60 and 1.44 values for 99.77% and 89.32% of the extent, respectively) (Table 4, Table 5), but of medium value at the Irish Sea case study (2.67) (Table 6).

These results reveal clear spatial variability within each case study (Fig. 4 & Supplementary Material D). For example, Cantabria case study results show that condition declines as we move offshore — with the

Table 4
Cantabria case study EUNIS habitats extent, condition and confidence, ordered from highest to lowest condition. (EUNIS: European Nature Information System; EV: ecological value; HEISb: *high energy infralittoral seabed*; A3.1: *Atlantic and Mediterranean high energy infralittoral rock*; A5.23 or A5.24: *Infralittoral fine sand or Infralittoral muddy sand*; A5.13: *Infralittoral coarse sediment*; A5.25 or A5.26: *Circalittoral fine sand or Circalittoral muddy sand*; A5.27: *Deep circalittoral sand*; A4.1: *Atlantic and Mediterranean high energy circalittoral rock*; A4.12: *Sponge communities on deep circalittoral rock*; A3: *Infralittoral rock and other hard substrata*; ISb: *Infralittoral seabed*; A5.14: *Circalittoral coarse sediment*; A5.35: *Circalittoral sandy mud*).

EUNIS habitat codes	Extent (km^2) (%)	Condition = EV (std)	Confidence (std)
HEISb	0.57 (0.31)	3.25 (±0.56)	1.74 (±0.70)
A3.1	35.96 (19.39)	3.12 (±0.56)	1.46 (±0.35)
A5.23 or A5.24	38.47 (20.74)	2.95 (±0.56)	1.84 (±0.43)
A5.13	1.46 (0.79)	2.70	1.49
A5.25 or A5.26	54.49 (29.38)	2.30 (±0.68)	1.58 (±0.36)
A5.27	<0.00 (<0.00)	2.14 (±0.00)	1.53 (±0.00)
A4.1	53.07 (28.61)	1.90 (±0.62)	1.54 (±0.19)
A4.12	1.01 (0.55)	1.88	1.53
A3	0.02 (0.01)	1.46	2.80
ISb	<0.00 (<0.00)	1.26	2.80
A5.14	0.23 (0.12)	1.24	2.20
A5.35	0.21 (0.11)	0.82	3.00

Table 5
North Sea EUNIS habitats extent, condition and confidence, ordered from highest to lowest condition. (EUNIS: European Nature Information System; EV: ecological value; A5.611: *Sabellaria spinulosa on stable circalittoral mixed sediment*; A5.44: *Circalittoral mixed sediments*; A5.33: *Infralittoral sandy mud*; A5.45: *Deep circalittoral mixed sediments*; A5.6: *Sublittoral biogenic reefs*; A5.37: *Deep circalittoral mud*; A5.15: *Deep circalittoral coarse sediment*; A5.13: *Infralittoral coarse sediment*; A5.14: *Circalittoral coarse sediment*; A5.23 or A5.24: *Infralittoral fine sand or Infralittoral muddy sand*; A5.27: *Deep circalittoral sand*; A5.35: *Circalittoral sandy mud*; A5.25 or A5.26: *Circalittoral fine sand or Circalittoral muddy sand*; ISb: *Infralittoral seabed*; CSb: *Circalittoral seabed*).

EUNIS habitat codes	Extent (km^2) (%)	Condition = EV (std)	Confidence (std)
A5.611	1.15 (0.01)	4.99 (±0.20)	1.54 (±0.25)
A5.44	0.58 (<0.00)	4.28 (±0.76)	1.72 (±0.35)
A5.33	0.63 (<0.00)	3.96 (±1.01)	2.08 (±0.00)
A5.45	97.11 (0.51)	3.88 (±0.76)	1.38 (±0.36)
A5.6	133.57 (0.70)	3.66 (±0.90)	1.59 (±0.34)
A5.37	720.03 (3.79)	3.62 (±0.74)	1.46 (±0.41)
A5.15	3590.16 (18.88)	3.39 (±0.86)	1.46 (±0.39)
A5.13	375.90 (1.98)	3.35 (±0.59)	1.4 (±0.35)
A5.14	1040.47 (5.47)	3.21 (±0.88)	1.51 (±0.41)
A5.23 or A5.24	1290.71 (6.79)	3.14 (±1.08)	1.43 (±0.3)
A5.27	8073.95 (42.47)	3.00 (±0.93)	1.42 (±0.38)
A5.35	228.21 (1.20)	2.95 (±0.82)	1.46 (±0.34)
A5.25 or A5.26	3459 (18.2)	2.84 (±0.87)	1.46 (±0.36)
ISb	0.02 (<0.00)	1.80 (±0.75)	2.08 (±0.00)
CSb	0.01 (<0.00)	1.43 (±0.00)	2.08 (±0.00)

highest scores along the coastline and lower scores in open-ocean habitats — and infralittoral types' condition outperforms circalittoral ones (Table 4, Fig. S11). On the other hand, the Irish Sea case study shows a gradient of improving condition with distance from the coast (Fig. 4). The North Sea case study exhibits two distinct zones: the UK Exclusive Economic Zone performing with high scores, and the Belgian Exclusive Economic Zone performing with medium scores (Fig. S12).

3.2.2. Ecosystem component condition and confidence values per EUNIS habitat type

If one looks into the results per ecosystem component condition across different habitats within each case study, we observe a significant variability of results. At the Cantabria case study, intertidal macroalgae and angiosperms ecosystem components obtained very high condition scores, whilst subtidal macroalgae ranged from low scores for *Deep circalittoral sand* (A5.27) habitats to high scores in *High energy infralittoral seabed* (HEISb) habitats. Benthic macroinvertebrates ecosystem component mostly obtained medium scores (i.e. *Infralittoral coarse sediment* (A5.13), *Sponge communities on deep circalittoral rock* (A4.12)) and benthic habitats ecosystem component scores ranged from very low in *Circalittoral sandy mud* (A5.35) to high in *High energy infralittitoral seabed* (Table 7; Supplementary Material E).

In the North Sea case study, the differences in ecosystem component condition values are evident between habitats. Macrozoobenthos ecosystem component presents varying condition scores, from low scores at *Circalittoral Seabed* (CSb) to very high scores in *Sabellaria spinulosa on stable circalittoral mixed sediment* (A5.611) (Table 7). In addition, benthic habitats mostly obtained high condition scores (i.e. *Deep circalittoral sand*, *Circalittoral mixed sediments* (A5.44), *Circalittoral coarse sediment* (A5.14)), with *Sabellaria spinulosa on stable circalittoral mixed sediment* scoring very high condition scores.

The Irish Sea case study results show that benthic habitats all score very low or low condition values (Table 7). Nevertheless, macrozoobenthos obtained scores which ranged from very low in *Atlantic and mediterranean high energy infralittoral rock* (A3.1) to high in *Deep circalittoral seabed* (DCSb) (Table 7). Demersal fish ecosystem component consistently obtained high and very high scores in all EUNIS habitat types in the Irish Sea case study (Table 7).

Table 6
Irish Sea EUNIS habitats extent, condition and confidence, ordered from highest to lowest condition. (EUNIS: European Nature Information System; EV: ecological value; A5.15: *Deep circalittoral coarse sediment*; A5.27: *Deep circalittoral sand*; A5.45: *Deep circalittoral mixed sediments*; A5.37: *Deep circalittoral mud*; A5.6: *Sublittoral biogenic reefs*; LECSb: ; A5.61: *Sublittoral polychaete worm reefs on sediment*; A5.33: *Infralittoral sandy mud*; HECSb: *High energy circalittoral seabed*; A5.611: *Sabellaria spinulosa on stable circalittoral mixed sediment*; LEDCSb: *Low energy deep circalittoral seabed*; A5.625: *Mytilus edulis beds on sublittoral sediment*; CSb: *Circallitoral seabed*; A5.23 or A5.24: *Infralittoral fine sand or Infralittoral muddy sand*; A5.44: *Circalittoral mixed sediments*; A5.25 or A5.26: *Circalittoral fine sand or Circalittoral muddy sand*; A4.12 or A4.27 or A4.33: *Sponge communities on deep circalittoral rock or Faunal communities on deep moderate energy circalittoral rock or Faunal communities on deep low energy circalittoral rock*; A5.36: *Circalittoral fine mud*; ISb: *Infrallitoral seabed*; A5.14: *Circalittoral coarse sediment*; A5.35: *Circalittoral sandy mud*; A3.3: *Atlantic and Mediterranean low energy infralittoral rock*; MECSb: *Moderate energy circalittoral seabed*; MEISb: *Moderate energy infralittoral seabed*; A5.34: *Infralittoral fine mud*; HEISb: *High energy infralittoral seabed*; LEISb: *Low energy infralittoral seabed*; A4: *Circalittoral rock and other hard substrata*; A3.2: *Atlantic and Mediterranean moderate energy infralittoral rock*; MEDCSb: *Moderate energy deep circalittoral seabed*; A4.33: *Faunal communities on deep low energy circalittoral rock*; A3.1: *Atlantic and Mediterranean high energy infralittoral rock*; ISd: *Infrallitoral sediment*; A4.1: *Atlantic and Mediterranean high energy circalittoral rock*; DCSb: *Deep circalittoral seabed*; A3: *Infralittoral rock and other hard substrata*; A4.3: *Atlantic and Mediterranean low energy circalittoral rock*; A4.2: *Atlantic and Mediterranean moderate energy circalittoral rock*; A4.27: *Faunal communities on deep moderate energy circalittoral rock*; A5.13: *Infralittoral coarse sediment*; HEDCSb: *High energy deep circalittoral seabed*; A5.43: *Infralittoral mixed sediments*; A4.12: *Sponge communities on deep circalittoral rock*; DCSd: *Deep circalittoral sediment*; SCSd: *Shallow circalittoral sediment*).

EUNIS habitat codes	Extent (km^2) (%)	Condition = EV (std)	Confidence (std)
A5.15	5892.82 (29.45)	2.43 (±0.37)	2.87 (±0.35)
A5.27	3644.71 (18.21)	2.27 (±0.55)	2.78 (±0.22)
A5.45	1046.68 (5.23)	1.99 (±0.37)	3.05 (±0.18)
A5.37	4789.00 (23.93)	1.66 (±0.52)	2.56 (±0.23)
A5.6	122.29 (0.61)	1.65 (±0.47)	2.81 (±0.88)
LECSb	14.93 (0.07)	1.61 (±0.41)	1.80 (±0.65)
A5.61	0.14 (<0.00)	1.39 (±0.12)	3.00 (±0.00)
A5.33	99.18 (0.50)	1.33 (±0.52)	2.45 (±0.62)
HECSb	169.29 (0.85)	1.31 (±0.53)	1.83 (±0.67)
A5.611	0.15 (<0.00)	1.24 (±0.17)	3.00 (±0.00)
LEDCSb	1.48 (0.01)	1.21 (±0.49)	1.56 (±0.44)
A5.625	12.30 (0.06)	1.21 (±0.39)	2.26 (±0.74)
CSb	31.27 (0.16)	1.17 (±0.68)	1.83 (±0.78)
A5.23 or A5.24	352.78 (1.76)	1.13 (±0.48)	2.65 (±0.72)
A5.44	68.55 (0.34)	1.11 (±0.37)	2.41 (±0.41)
A5.25 or A5.26	959.12 (4.79)	1.08 (±0.37)	2.53 (±0.40)
A4.12 or A4.27 or A4.33	0.24 (<0.00)	1.06 (±0.37)	2.70 (±0.82)
A5.36	11.86 (0.06)	1.05 (±0.33)	3.00 (±0.00)
ISb	143.97 (0.72)	1.03 (±0.49)	1.75 (±0.76)
A5.14	335.11 (1.67)	1.03 (±0.41)	2.64 (±0.38)
A5.35	875.24 (4.37)	0.98 (±0.57)	2.27 (±0.28)
A3.3	0.01 (<0.00)	0.97 (±0.50)	2.55 (±0.49)
MECSb	187.51 (0.94)	0.96 (±0.87)	1.75 (±0.60)
MEISb	157.41 (0.79)	0.96 (±0.71)	1.46 (±0.62)
A5.34	16.76 (0.08)	0.95 (±0.40)	2.86 (±0.71)
HEISb	301.24 (1.51)	0.90 (±0.42)	1.71 (±0.69)
LEISb	20.03 (0.10)	0.88 (±0.41)	1.62 (±0.64)
A4	0.91 (<0.00)	0.88 (±0.37)	2.80 (±0.68)
A3.2	2.01 (0.01)	0.86 (±0.37)	2.81 (±0.79)
MEDCSb	6.41 (0.03)	0.78 (±1.26)	1.76 (±0.52)
A4.33	16.87 (0.08)	0.76 (±0.76)	2.98 (±0.32)
A3.1	13.24 (0.07)	0.76 (±0.33)	2.66 (±0.76)
ISd	2.15 (0.01)	0.73 (±0.42)	1.80 (±0.81)
A4.1	38.42 (0.19)	0.73 (±0.24)	2.90 (±0.62)
DCSb	5.40 (0.03)	0.72 (±1.04)	1.53 (±0.78)
A3	6.69 (0.03)	0.69 (±0.40)	2.84 (±0.78)
A4.3	1.81 (0.01)	0.69 (±0.13)	3.00 (±0.00)
A4.2	30.64 (0.15)	0.68 (±0.38)	2.94 (±0.34)
A4.27	326.05 (1.63)	0.67 (±0.24)	2.74 (±0.62)
A5.13	85.60 (0.43)	0.65 (±0.35)	2.84 (±0.43)
HEDCSb	16.70 (0.08)	0.63 (±0.51)	1.79 (±0.60)

(*continued on next page*)

Table 6 (*continued*)

EUNIS habitat codes	Extent (km^2) (%)	Condition = EV (std)	Confidence (std)
A5.43	4.61 (0.02)	0.60 (±0.18)	2.64 (±0.31)
A4.12	55.06 (0.28)	0.59 (±0.37)	2.86 (±0.76)
DCSd	131.19 (0.66)	0.51 (±0.21)	2.15 (±0.35)
SCSd	13.24 (0.07)	0.48 (±0.33)	1.93 (±0.74)

3.2.3. Ocean physical stock accounts (OPSA) compilation results

Merging the condition results with those from extent, the results show a great variation within most EUNIS habitat types, their extents, conditions and confidence scores.

The Cantabria case study comprises 12 habitat types, where four cover more than 98% of the total extent (Table 4). *Circalittoral fine sand or Circalittoral muddy sand* (A5.25 or A5.26) and *Atlantic and mediterranean high energy circalittoral rock* (A4.1) cover 29.38% and 28.61% (Table 4). Other habitats such as *Infralittoral fine sand or Infralittoral muddy sand* (A5.23 or A5.24) and *Atlantic and mediterranean high energy infralittoral rock* (A3.1), also represent a great portion with 20.74% and 19.39% (Table 4). Additionally, habitats scoring best conditions in Cantabria are *High energy infralittoral seabed*, *Atlantic and mediterranean high energy infralittoral rock* and *Infralittoral fine sand or Infralittoral muddy sand*. On the other hand, habitats scoring lowest conditions are *Circalittoral sandy mud*, *Circalittoral coarse sediment* and *Infralittoral seabed* (ISb) (Table 4). In Cantabria, the predominant EUNIS habitats ranked 5th, 7th, 3rd, and 2nd in terms of condition. Confidence scores for EUNIS habitat types ranged from low to high, although eight habitats, covering 99.77% of the total extent, received low confidence scores (Table 4, Fig. S11).

In the North Sea case study, three out of 15 habitat types cover 79.55% of the total extent (Table 5). Predominant habitats in the North Sea are *Deep circalittoral sand*, *Deep circalittoral coarse sediment* (A5.15) and *Circalittoral fine sand or Circalittoral muddy sand*, covering 42.47%, 18.88% and 18.20%, respectively (Table 5). Other habitats such as *Infralittoral fine sand or Infralittoral muddy sand*, *Circalittoral coarse sediment* and *Deep circalittoral mud* (A5.37) also represent a significant portion of the North Sea with 6.79%, 5.47% and 3.79% of the extent (Table 5). Moreover, habitats with the best condition in the North Sea are *Sabellaria spinulosa on stable circalittoral mixed sediment*, *Circalittoral mixed sediments* and *Infralittoral sandy mud* (A5.33), while the worst condition has been obtained by *Circalittoral Seabed*, *Infralittoral seabed* and *Circalittoral fine sand or Circalittoral muddy sand* (Table 5). Surprisingly, habitats with the best condition represent a negligible portion of the total extent and *Circalittoral fine sand or Circalittoral muddy sand* cover 18.2% of the total extent. Thus, predominant habitats in the North Sea ranked 11th, 7th and 13th, respectively. Confidence scores in the North Sea comprise low and medium scores, and almost 100% in the North Sea obtained low scores (Table 5, Fig. S12).

The Irish Sea case study comprises 45 habitat types, whereas the three dominant types cover 71.59% of the total extent (Table 6). *Deep circalittoral coarse sediment*, *Deep circalittoral mud* and *Deep circalittoral sand* represent 29.45%, 23.93% and 18.21% of the area, respectively (Table 6). However, *Deep circalittoral mixed sediments* (A5.45),

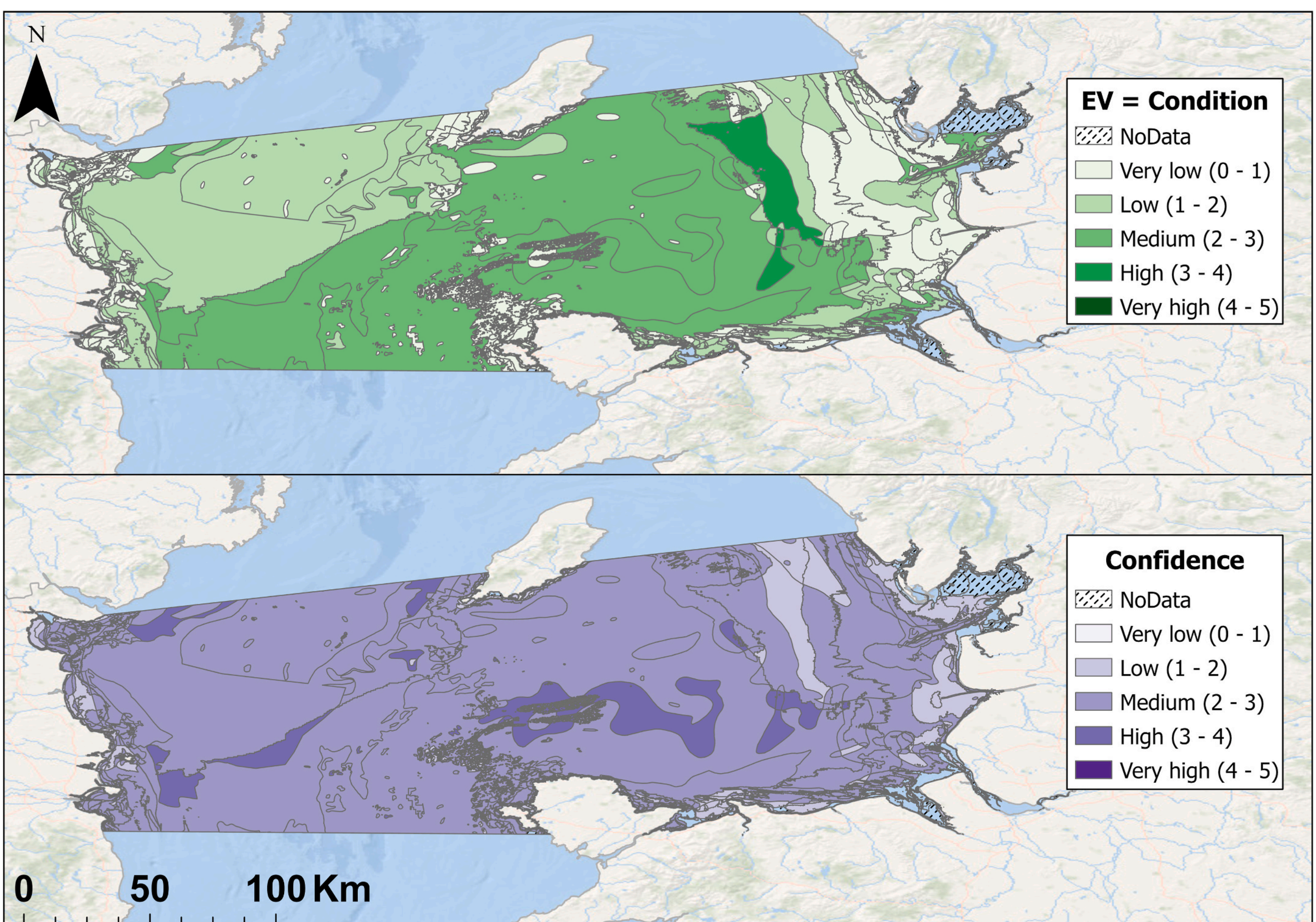

Fig. 4. Irish Sea CS condition and confidence. Map representing EUNIS habitat condition (top) and assessment confidence (bottom). (CS: case study; EUNIS: European Nature Information System; EV: Ecological Value).

Table 7
Ecosystem component condition by EUNIS habitat type in the three case study areas. (EUNIS: European Nature Information System; Con.: ecosystem component condition; Hab.: EUNIS habitats codes; A5.15: *Deep circalittoral coarse sediment*; A5.27: *Deep circalittoral sand*; A5.45: *Deep circalittoral mixed sediments*; A5.37: *Deep circalittoral mud*; A5.6: *Sublittoral biogenic reefs*; LECSb: ; A5.61: *Sublittoral polychaete worm reefs on sediment*; A5.33: *Infralittoral sandy mud*; HECSb: *High energy circalittoral seabed*; A5.611: *Sabellaria spinulosa on stable circalittoral mixed sediment*; LEDCSb: *Low energy deep circalittoral seabed*; A5.625: *Mytilus edulis beds on sublittoral sediment*; CSb: *Circalittoral seabed*; A5.23 or A5.24: *Infralittoral fine sand or Infralittoral muddy sand*; A5.44: *Circalittoral mixed sediments*; A5.25 or A5.26: *Circalittoral fine sand or Circalittoral muddy sand*; A4.12 or A4.27 or A4.33: *Sponge communities on deep circalittoral rock or Faunal communities on deep moderate energy circalittoral rock or Faunal communities on deep low energy circalittoral rock*; A5.36: *Circalittoral fine mud*; ISb: *Infralittoral seabed*; A5.14: *Circalittoral coarse sediment*; A5.35: *Circalittoral sandy mud*; A3.3: *Atlantic and Mediterranean low energy infralittoral rock*; MECSb: *Moderate energy circalittoral seabed*; MEISb: *Moderate energy infralittoral seabed*; A5.34: *Infralittoral fine mud*; HEISb: *High energy infralittoral seabed*; LEISb: *Low energy infralittoral seabed*; A4: *Circalittoral rock and other hard substrata*; A3.2: *Atlantic and Mediterranean moderate energy infralittoral rock*; MEDCSb: *Moderate energy deep circalittoral seabed*; A4.33: *Faunal communities on deep low energy circalittoral rock*; A3.1: *Atlantic and Mediterranean high energy infralittoral rock*; ISd: *Infralittoral sediment*; A4.1: *Atlantic and Mediterranean high energy circalittoral rock*; DCSb: *Deep circalittoral seabed*; A3: *Infralittoral rock and other hard substrata*; A4.3: *Atlantic and Mediterranean low energy circalittoral rock*; A4.2: *Atlantic and Mediterranean moderate energy circalittoral rock*; A4.27: *Faunal communities on deep moderate energy circalittoral rock*; A5.13: *Infralittoral coarse sediment*; HEDCSb: *High energy deep circalittoral seabed*; A5.43: *Infralittoral mixed sediments*; A4.12: *Sponge communities on deep circalittoral rock*; DCSd: *Deep circalittoral sediment*; SCSd: *Shallow circalittoral sediment*).

Cantabria		
Ecosystem component	Con.	Hab.
Angiosperms	Very high	HEISb, A5.23 or A5.24, A5.25 or A5.26
Benthic macroinvertebrates	Low	A3, ISb
	Medium	HEISb, A5.23 or A5.24, A5.25 or A5.26, A5.13, A3.1, A4.1, A4.12, A5.27
Intertidal macroalgae	Very high	HEISb, A5.23 or A5.24, A5.25 or A5.26, A5.13, A3.1, A4.1
Subtidal macroalgae	Low	A5.25 or A5.26, A5.14, A4.1, A4.12, A5.27
	Medium	A5.23 or A5.24, A5.13, A3.1
	High	HEISb
Benthic habitats	Very low	A5.35
	Low	A5.23 or A5.24, A5.25 or A5.26, A5.13, A5.14, A4.1, A3, A4.12, ISb
	Medium	A3.1, A5.27
	High	HEISb
North Sea		
Ecosystem component	**Con.**	**Hab.**
Benthic habitats	Medium	A5.35, A5.13
	High	A5.15, A5.27, A5.23 or A5.24, A5.25 or A5.26, A5.45, A5.44, A5.6, A5.14, A5.37
	Very high	A5.611
Macrozoobenthos	Low	ISb, CSb
	Medium	A5.27, A5.25 or A5.26
	High	A5.15, A5.35, A5.33, A5.23 or A5.24, A5.6, A5.14, A5.37, A5.13
	Very high	A5.45, A5.44, A5.611
Irish Sea		
Ecosystem component	**Con.**	**Hab.**
Benthic habitats	Very low	A5.15, A5.27, HEISb, HECSb, A5.14, DCSb, LEISb, MEISb, MECSb, A5.13, A5.45, A5.43, A5.44, A3, A3.1, A4.27, A4.2, A3.2, A4.1, A3.3, A4, A4.3, A4.33, A4.12, HEDCSb, MEDCSb, A5.34, A5.37, A5.35, A5.6, ISd, SCSd, DCSd
	Low	A5.25 or A5.26, ISb, CSb, LECSb, A5.23 or A5.24, A4.12 or A4.27 or A4.33, LEDCSb, A5.36, A5.33, A5.611, A5.61, A5.625
Macrozoobenthos	Very low	A3.1, A5.37, A5.35
	Low	A5.25 or A5.26, ISb, HEISb, MECSb, MEISb, A5.23 or A5.24, A5.13, A5.44, A4.1, MEDCSb, A5.33, SCSd, DCSd
	Medium	A5.14, A4.27, A5.625
	High	A5.15, A5.27, HECSb, CSb, DCSb, A5.45
Demersal fish	High	A5.15, A5.27, A4.27, A5.37, A5.6
	Very high	A5.45, A4.33, DCSd

Circalittoral fine sand or Circalittoral muddy sand and *Circalittoral sandy mud* habitats also represent a considerable portion of the Irish Sea with 5.23%, 4.79% and 4.37% (Table 6). With regards to the habitats with the best condition, *Deep circalittoral coarse sediment*, *Deep circalittoral sand* and *Deep circalittoral mixed sediments* ranked best while *Shallow circalittoral sediment* (SCSd), *Deep circalittoral sediment* (DCSd) and *Sponge communities on deep circalittoral rock* (A4.12) obtained the worst condition scores (Table 6). In the Irish Sea, predominant habitats ranked 1st, 4th and 2nd, respectively (Table 6). Confidence scores in the Irish Sea ranged from low to high confidence scores and habitats with medium, low and high confidence scores covered 89.32%, 5.37% and 5.3%, respectively (Table 6 and Fig. 4).

4. Discussion

This paper presents a first attempt to integrate the EVA approach with a standardised habitat classification system (e.g., EUNIS) to evaluate ocean physical stock accounts within a natural capital accounting framework. The methodology incorporates multiple ecological levels into the accounting framework by linking different ecosystem components and their biodiversity with underlying habitats.

The paper demonstrates that.

1. Spatial patterns of ecosystem condition vary depending on the context. In the Cantabria case study (CS1) and the Belgian Exclusive Economic Zone of the North Sea case study (CS2), the most ecologically valuable areas are located along the coastal zone, agreeing with previous findings from studies of (Derous et al., 2007a; Pascual et al., 2011; Vanden Eede, 2007). By contrast, spatial patterns obtained in the Irish Sea (CS3) can be largely explained by the offshore concentration of higher condition scores in the ecosystem components macrozoobenthos and demersal fish (Figs. S9 and S10). A similar offshore pattern for macrozoobenthos is observed in the other case study areas, reflecting the tendency of macrozoobenthos and benthic macroinvertebrates higher condition to be associated with circalittoral habitats (e.g. *Sponge communities on deep circalittoral rock* in Cantabria, *Sabellaria spinulosa on stable circalittoral mixed sediment* in the North Sea and *Deep circalittoral seabed* in the Irish Sea) (Table 7). Additionally, differences in condition between the Belgian and UK Exclusive Economic Zones in the North Sea could be explained by the EVA criterion for Nationally Rare Features (Table 1), which assesses rarity within national waters. Because this definition (<0.5% occurrence in 10 × 10 km squares) is applied separately for each country, the classification of a species as nationally rare depends partly on the spatial extent of its national Exclusive Economic Zone (e.g. *Abra alba* may be classified as nationally rare within the UK Exclusive Economic Zone but not within the Belgian Exclusive Economic Zone).

2. Specific ecosystem components and data sources drive habitat condition variability. Cantabria case study (CS1) habitat condition is influenced by the presence of intertidal macroalgae and angiosperms, which scored very high and drive high condition scores in particular areas. Intertidal macroalgae and angiosperms are characterised by habitat-forming species inhabiting particular regions that meet specific environmental conditions. For angiosperms, these typically include low-energy intertidal or sheltered shallow waters (Allen and Pye, 1992; Unsworth et al., 2019), whereas intertidal macroalgae also have habitat restrictions but can inhabit areas more exposed to wave action. This geographically restricted nature highlights the uniqueness of these habitats, leading to higher condition scores in AQs related to rarity and ecologically significant feature presence. However, the relatively good conditions in the Irish Sea (CS3) predominant habitats can be explained by the presence of demersal fish and macrozoobenthos ecosystem components, which systematically obtained better scores compared to benthic habitats. Moreover, demersal fish and macrozoobenthos EVA were carried out directly using monitoring data, leading to a positive contribution of these ecosystem components to the total condition only in areas/habitats covered by the respective monitoring programs. Modelling and extrapolation techniques, such as species distribution models and regression algorithms in data-scarce contexts like oceans, are a suitable option to reduce such spatial gaps and biases arising from the use of monitoring data to inform the EV/condition assessment. On one hand, species distribution models could be used to identify areas appropriate for specific species and habitats where no monitoring was carried out (Costa et al., 2022; Moore et al., 2009; Valle et al., 2014). On the other hand, regression techniques can estimate abundances of ecologically significant features, nationally rare features and habitat-forming species/biogenic habitats to assess quantitative AQs where data is not available (Brosse et al., 1999).

Apart from these two main findings, we should highlight that this study also shows that the averaged area-weighted condition scores do not imply comparability among case studies, as condition scores are highly dependent on data availability and how EVA has been applied in each case study. Such variations are inherent to initial applications of the method and highlight the importance of developing standardised criteria to ensure ecological relevance and comparability across case studies. Area-weighted condition scores of EUNIS habitats are not comparable due to a) the diverse application of field data in the ecosystem component condition assessments (e.g. extrapolated field data in the Cantabria case study and direct use of field data in the Irish

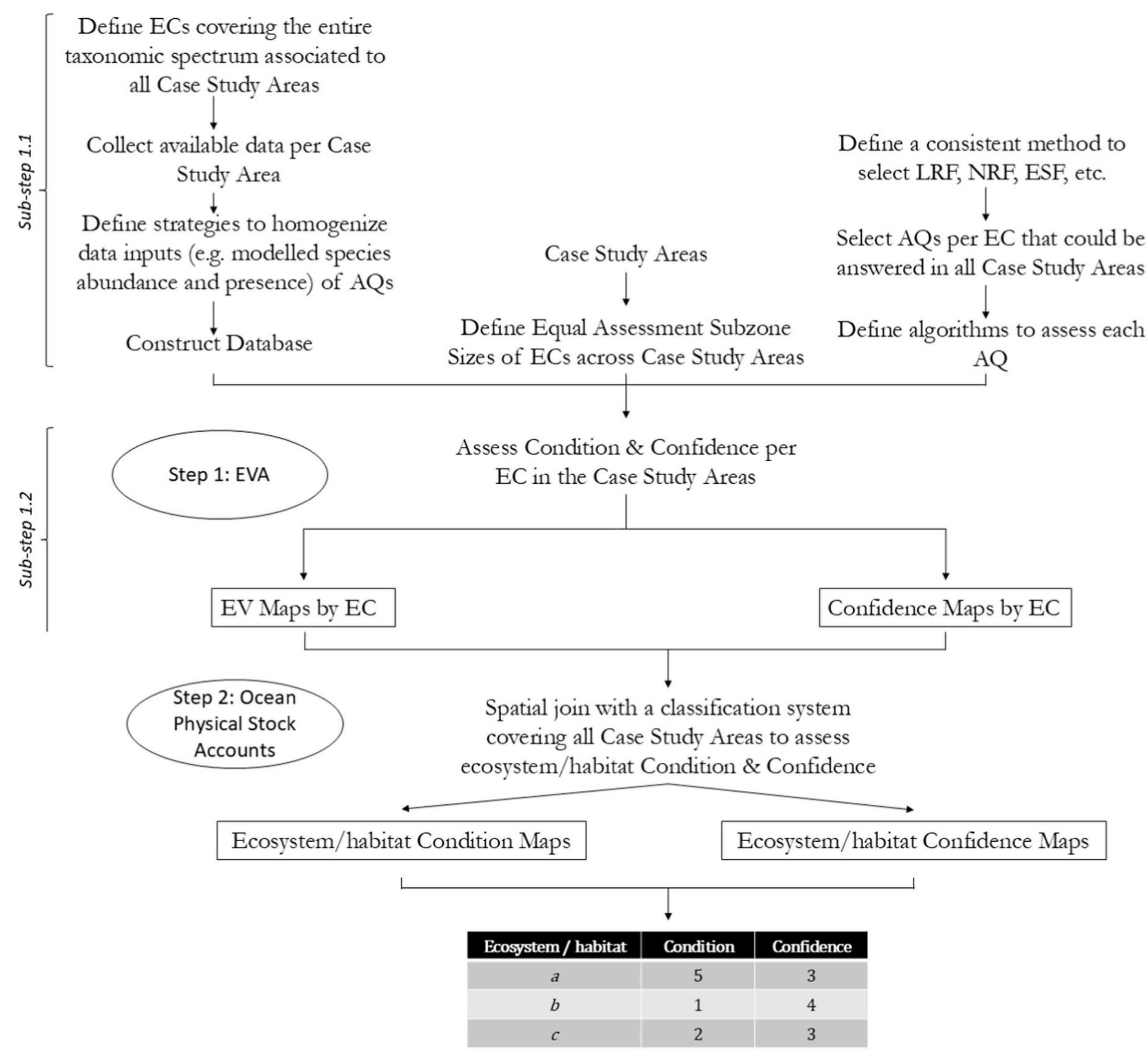

Fig. 5. Potential process for methodological standardization. The figure represents the process to follow for methodological standardization when comparability among case study areas is needed. (EC: ecosystem component; AQ: assessment question; LRF: locally rare features; NRF: nationally rare features; ESF: ecologically significant features; EVA: ecological value assessment; EV: ecological value).

Sea case study), b) the mismatches of AQs assessed for the same ecosystem component derived by data availability differences (e.g. AQ1 - AQ13 for benthic habitats in Cantabria case study and AQ1 – AQ9 in the North Sea case study) and c) the mismatches of grid sizes for the same ecosystem component (e.g. 3000-meter in the North Sea case study and 250-m in the Irish Sea case study). Therefore, we suggest following the process described in Fig. 5 to apply the methodology in future studies where comparability across case studies is needed. However, methodological adaptations that may be needed for comparability could lead to an oversimplification of the condition assessments, as a reduced number of AQs may be answered to homogenize AQs across case studies due to limited data availability in many oceanic contexts. Thus, while data limitations in oceanic contexts persist, the proposed methodology aligns better with frameworks with a focus on national-scale implementation, such as the SEEA-EA. Nevertheless, this limitation is inherent to the lack of data in oceanic contexts, rather than a methodological limitation.

On the other hand, as previously mentioned, most existing studies have estimated physical flow accounts solely with ecosystem extent, forgetting about the fundamental linkage between ecosystem condition and ecosystem service delivery (United Nations, 2021). This gap limits the ability for informed and robust decision making, as extent alone does not capture the actual capacity of ecosystems to sustain the flow of goods and services (Maes et al., 2020). Thus, when considering the subsequent necessary step of linking the OPSA with the ocean physical flow accounts (OPFA) (see Fig. 1), the EVA approach emerges as a promising method. This is supported by our results, which show that EVA results can effectively translate a standardisable, spatially explicit assessment (comparable across subzones) into an integrated estimate of the underlying ecosystem condition, thereby providing a robust basis for linking OPSA to OPFA requirements (Fig. 6). As such, EVA remains a valuable method for generating OPSA, as OPSA can be easily linked to OPFA, and data inputs, ecosystem components and subzone sizes can be harmonized for future studies' standardisation. Such standardisation would not only enable robust cross-site comparisons but also facilitate the integration of condition metrics into national natural capital accounts, which require a consistent application of the methodologies across both space and time.

Simultaneously, the condition results of this study can be used to compare the condition within a case study and identify hotspots with the highest condition, as well as those areas with low condition where either policy or management intervention is needed. It is important to note that condition scores should only be used as an inherent condition of biodiversity for those habitats where a direct link between the ecosystem component and habitat can be made. At the same time, the proposed integrated methodology shows its potential to become a method capable of tracking ocean ecosystem extent and condition changes. This would be essential to monitor changes or progress of some of these ecosystems throughout time or as responses to human impacts or managerial actions.

Furthermore, the methodology requires limited data and is highly adaptable, making it suitable for use with various types of data. It can also be easily adapted to other contexts with different characteristics and classification systems (e.g. IUCN GET), thus showcasing significant scalability. In addition, this flexible methodology can be readily adapted to other contexts and may enable linking ecosystem inherent condition with physical flow accounts, when that information is also available. This flexibility, combined with the potential for methodological harmonization across case studies, makes a promising methodology for supporting national and international ocean natural capital accounting efforts.

Finally, we acknowledge that, as a novel methodology for OPSA and a potential link to OPFA in oceanic realms, the assumptions and limitations of the methodology should be highlighted for future testing and improvement. We highlight two groups of assumptions and limitations.

1) Assumptions and limitations inherent to the EVA:
   a. Smoothing of the spatial variability of ecological processes and biodiversity structure represented by AQs in case the resolution of the data sources is higher than the subzone size.
   b. AQs fully cover the ecosystem components' condition characteristics.
   c. The most ecologically relevant attribute expresses the condition of the ecosystem components in each subzone.
   d. Although the aggregation steps follow an ecological rationale, a formal sensitivity or robustness analysis remains a key area for future methodological refinement.
   e. Confidence scores are a proxy of evidence strength rather than a measure of uncertainty.
2) Assumptions and limitations inherent to the OPSA assessment:
   a. The most relevant ecosystem components defined in each case study area completely represent the underlying EUNIS habitat condition.
   b. Equal importance of all subzones in the EUNIS habitat polygon condition assessment.
   c. Equal importance of all ecosystem components in the EUNIS habitat condition assessment.
   d. Pelagic ecosystem components do not reflect EUNIS habitat condition.

5. Conclusion

Our study illustrates that integrating, within an Ocean Accounting framework, existing ecological valuation methods (such as EVA) with established habitat classification systems (EUNIS) offers a novel approach to monitoring changes in marine ecosystem extent and condition. While the focus has been on the development of Ocean Physical Stock Accounts (OPSA), the methodology supports future OPFA development, which has typically linked the Ocean Physical Flow Accounts (OPFA) only to habitat extent data. By providing a spatially explicit and ecologically grounded assessment of condition, our methodology establishes the enabling conditions to connect both extent and condition to

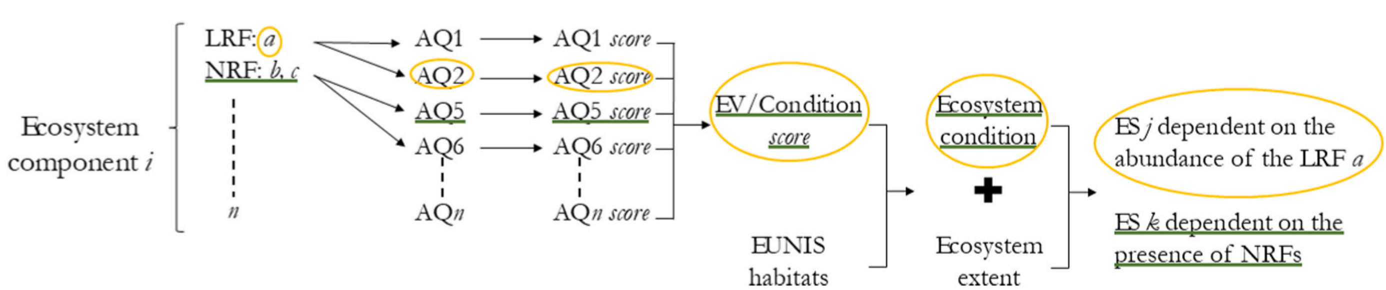

Fig. 6. Example of how EVA-based condition links to Ocean Physical Flow Accounts. Schematic representation of how an EVA-based condition could be linked to Ocean Physical Flow Accounts. (LRF: locally rare features; NRF: nationally rare features; AQ: assessment question; EV: ecological value; EUNIS: European Nature Information System; ES: ecosystem service).

OPFA, thereby strengthening the basis for ocean ecosystem service flow accounting and providing a necessary precondition for subsequent asset valuation.

The proposed methodology requires limited data and is highly adaptable, making it suitable for use with various types of data, adaptable to other contexts and scalable to other regions. It makes a promising methodology for supporting national and international ocean natural capital accounting efforts.

CRediT authorship contribution statement

Beñat Egidazu-de la Parte: Writing – review & editing, Writing – original draft, Visualization, Software, Resources, Methodology, Investigation, Formal analysis, Data curation. **Stefano Balbi:** Writing – review & editing, Funding acquisition. **Ferdinando Villa:** Funding acquisition. **Anita Franco:** Writing – review & editing, Project administration, Methodology, Funding acquisition, Conceptualization. **Tiziana Luisetti:** Writing – review & editing, Methodology, Funding acquisition, Conceptualization. **Daryl Burdon:** Writing – review & editing, Methodology, Funding acquisition, Conceptualization. **Bárbara Ondiviela:** Writing – review & editing, Software, Resources, Funding acquisition, Data curation. **Cristina Galván:** Writing – review & editing, Software, Resources, Funding acquisition, Data curation. **Dorota Kolbuk:** Writing – review & editing, Software, Resources, Data curation. **Julie Bremner:** Writing – review & editing, Supervision, Project administration, Funding acquisition. **Willem Boone:** Writing – review & editing, Software, Resources, Data curation. **Hanneloor Heynderickx:** Software, Resources, Data curation. **Klaas Deneudt:** Software, Resources, Funding acquisition, Data curation. **Marta Pascual:** Writing – review & editing, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.indic.2026.101163.

Data availability

Data will be made available on request.

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