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REVIEW

Towards a broader perspective on marine biodiversity change

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Abstract
Biodiversity decline jeopardizes the foundation of natural ecosystems and human well-being, a concern that prompted major global agreements aiming to bend the curve towards a net positive biodiversity future. Still, the critical importance of safeguarding the diversity of life is far from receiving the attention it deserves, especially in marine settings. To understand (and overcome) the current limits of biodiversity mainstreaming, we integrate insights from the natural and social sciences to offer guidance on how to navigate the seemingly overwhelming complexity of this issue. We start by comparing biodiversity change to climate change to capture key distinctions in their multifaceted and context-dependent nature. Unlike climate change, the status and trends of biodiversity cannot be reduced to a single metric or target. Instead, effective biodiversity governance must focus on understanding how biodiversity is affected and how habitat extent, population size, or trends in composition capture these changes. The rise of molecular data promises to improve the representativeness of assessments and foster mechanistic understanding of the processes involved. Yet, it does not eliminate the need for effective communication of these issues to invoke meaningful action. Given its links to human well-being, biodiversity has a high chance of being engaging, but practitioners and scientists only marginally capitalize on the social, health, economic, and emotional values of their subject. Thus, we advocate for extending the assessment of biodiversity change and its functional consequences to include human values and emotions as an integral part of biodiversity reporting. Such a holistic framing, accounting for the complex spatial and temporal trajectories of biodiversity, will be vital in fostering more effective and inclusive conservation strategies.

Keywords Assessment · Biodiversity loss · Emotion · Human well-being · Monitoring · Nature's contribution to people

Introduction

Exactly 25 years ago, David Tilman started an essay on "Causes, consequences and ethics in biodiversity" by stating that "[t]he most striking feature of Earth is the existence of

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life, and the most striking feature of life is its diversity. This biological diversity, or biodiversity, has long been a source of wonderment and scientific curiosity, but is increasingly a source of concern" (Tilman 2000). These two sentences brilliantly encapsulate the fascination, singularity, and importance of (dealing with) biodiversity.

The fascination with the diversity of life forms on Earth is featured in the oldest surviving human artworks, some over 50,000 years old (Aubert et al. 2018; Oktaviana et al. 2024). Numerous Holocene paintings all over the world are detailed enough to help reconstruct past species distributions (Gámez-Brunswick and Rojas-Soto 2020). The same fascination is embedded in Sybille Merian's drawings (Merian 1705), the medieval *Carta Marina* (Olaus Magnus 1539, see Sandmo 2020), or Ernst Haeckel's *Kunstformen der Natur* (Haeckel 1899–1904) as well as the reports from early naturalists exploring ecosystems worldwide (von Linné 1753; Humboldt and Bonpland 1805; Darwin and Wallace 1858). Beyond fascination, biodiversity is tightly woven into many Indigenous and local knowledge systems, which correspondingly play a major role in the discourse on biological diversity (IPBES 2019; McElwee et al. 2020).

By contrast, the term *biodiversity* itself was only coined in the mid-1980s (Sarkar 2021) as a summary term encapsulating all of the dimensions of biological variation. According to the Convention on Biological Diversity (CBD 1992), the term encompasses "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems". This definition is broad and open, and it needs to be because part of biodiversity's singularity is the fact that there is a seemingly infinite number of ways in which life can vary. Archaea, bacteria, or hybridizing plants and animals regularly defy even the relatively simple biological species concept (organisms that can reproduce and have fertile offspring). Even the concept of an "individual" needs extension in the biodiversity context to include clonal organisms (ramets and genets) or holobionts incorporating a host and several symbionts. As it exceeds our cognitive and perceptual capacities as a whole, biodiversity becomes a "hyperobject" (sensu Morton 2013), as its entities are vast, not bound to a certain locality, and are temporally dispersed.

The notion of the complex nature of biodiversity, however, turned out to be a major obstacle when the topic moved from fascination to concern. Human domination of Earth's biomes (Vitousek et al. 1997) has reduced the spatial extent of wilderness, altered ecological communities, and endangered species survival. From Rachel Carson's *Silent Spring* (1962) to the global assessment by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES 2019), reports abound on this transformation of biodiversity and its anthropogenic drivers. National biodiversity strategies and international agreements such as the Convention on Biological Diversity (CBD 1992) have brought the biodiversity crisis to the political agenda, culminating in the Kunming-Montreal Global Biodiversity Framework (CBD 2022) and the agreement on marine biological diversity of areas beyond national jurisdiction (BBNJ 2023). This political importance reinforced a surge of hundreds of scientific studies emphasizing how much human well-being depends on biodiversity across all ecosystems (Cardinale et al. 2012; Brauman et al. 2020).

Despite its fundamental importance, singularity and imperilment, biodiversity rarely seems to become the foremost policy priority. Even compared to the intertwined (but not identical) issue of climate change, biodiversity change, management, and improvement trail in public attention and media coverage (Mammides and Campos-Arceiz 2025). In this paper, we explore potential reasons for this disparity and take the comparison between climate change and biodiversity change as an illustrative starting point (section "Learning from the climate change debate"). The following sections then establish four major tenets:

(i) To assess how much biodiversity is changing, we can use a suite of logically connected approaches but need to be aware of their limits (section "Measuring biodiversity change").
(ii) To capture biodiversity change more completely and make functional inferences, we can use molecular approaches but these do not alleviate the intrinsic issues of biodiversity metrics (section "The molecular revolution to the rescue—soon").
(iii) To bring sustainable biodiversity conservation into action, we have to both recognize the existing emotional dimensions of biodiversity and foster more meaningful emotional connections to biodiversity (section "Recognizing and integrating emotions in biodiversity conservation").
(iv) To develop useful biodiversity targets, we have to accept the non-equilibrium dynamics of biodiversity in order to develop an outcome space for biodiversity (section "Guiding principles for biodiversity policies and target setting").

Learning from the climate change debate

It seems unwise to propose learning from a debate that arguably has a huge gap between goals and action itself. In our context, learning means using the climate change debate as a mirror to identify some of the peculiarities of the biodiversity change debate. The commonalities between climate and biodiversity change (and measures for their mitigation) are described elsewhere (Pörtner et al. 2021; Pörtner et al. 2023), so that we can focus here on their distinctions.

The main difference becomes immediately evident from the visualization of humanity's future options, which are superficially similar in emphasizing the urgent need for bending and reversing an adverse trend. Climate policy aims to stay below boundaries of + 1.5 or + 2 °C or to revert to these after a potential overshoot, which provides the identical narrative as the "bending the curve" images of biodiversity loss (Leclère et al. 2020). For the latter, the visualization focuses on the aim to slow down the adverse trend and to recover some of what has been lost by achieving a nature-positive future (Fig. 1).

The difference begins already when focusing on what actually is displayed in these diagrams. In the case of climate futures, projected trajectories of carbon emission and temperature are displayed in absolute units such as tons CO_2 per year or °C (the most recent summary for policymakers IPCC 2021 has examples in Figs. 8 and 10). By contrast, the same diagram for biodiversity features a dimensionless "deviation from indicator value" (Leclère et al. 2020) or avoids specific axis labels altogether (Fig. 1). What might seem like a minor visualization detail actually reveals some of the major fallacies of biodiversity conservation, the multitude of metrics that additionally only exist in relative space (see "Measuring biodiversity change" and "The molecular revolution to the rescue—soon" sections), which leads to a level of abstraction that distances the assessment of diversity from the fascination for and thus (emotional) value of biodiversity (see "Recognizing and integrating emotions in biodiversity conservation" section). Like biodiversity, the climate system is vastly complex given the feedbacks between drivers, processes, and Earth system components across a range of temporal and spatial scales. A potential lesson from climate change research is how to map the complexity of the physical foundations into an outcome space encompassing a few key dimensions most important to humans ("Measuring biodiversity change" and "Guiding principles for biodiversity policies and target setting" sections).

A second major discrepancy is the starting point of the diagram. Based on information from different proxies and the start of temperature observation, we have a pretty well-defined pre-industrial climate baseline and can distinguish current temperature anomalies from natural variations. By contrast, for biodiversity, available ecological and paleoecological time series reveal a major gap for the last centuries with decadal resolution (Smith et al. 2023), which would be most needed for creating pre-industrial baselines. The ecological observation of biodiversity covers only the last few decades, whereas the paleoecological information does not necessarily have the temporal resolution to infer a natural range of rates of change (Yasuhara et al. 2020). While climate science can define targets against measurable baselines (e.g., pre-industrial temperatures), biodiversity research lacks comparable reference points (see "Measuring biodiversity change" section).

A third major discrepancy between climate change and biodiversity change is the question of reversibility. The climate system—nonwithstanding the global importance of biodiversity for the carbon cycle—is a largely physical system. Unless crossing climatic tipping points that prevent recovery, a reduction in carbon emissions has known effects on temperature, which allows calculating when humanity needs to be carbon neutral to keep us below 2 °C warming.

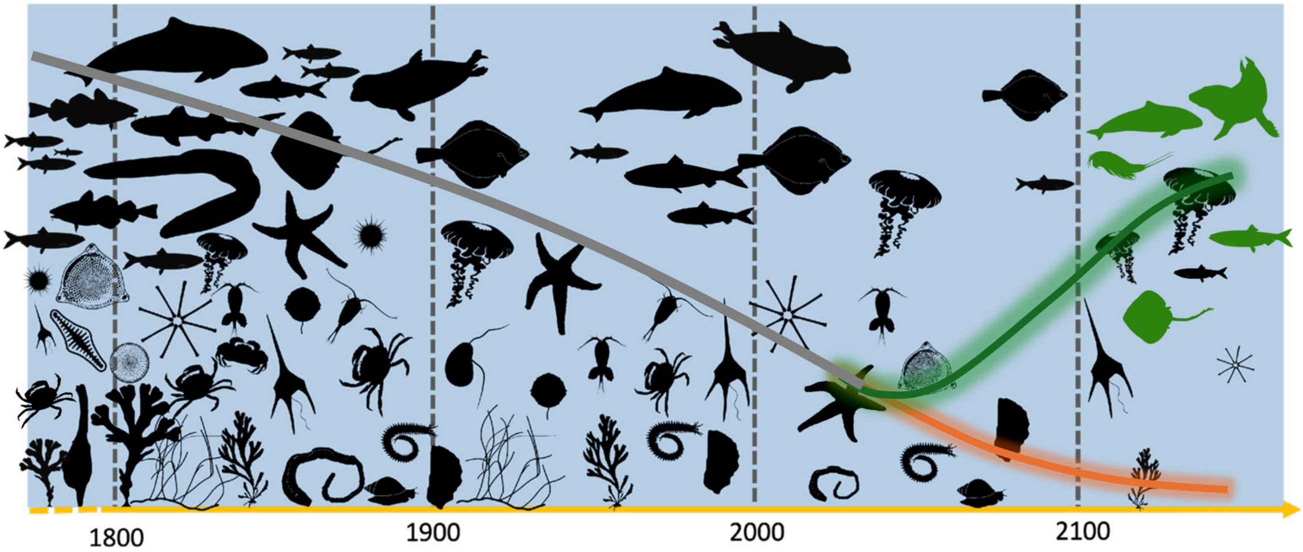


Fig. 1 Marine-focused visualization of the "bending the curve" narrative based on Leclère et al. (2020) using organism vignettes from PhyloPic, https://www.phylopic.org/

Biodiversity, however, does not work like a machine. Bringing back the same abiotic conditions will not necessarily bring back the same biodiversity ("Guiding principles for biodiversity policies and target setting" section). At the same time, the very strong narrative of tipping points is much harder to apply in a biodiversity context (Hillebrand et al. 2023). On the one hand, every species lost is a non-reversible tipping point for that species, but on the other hand, the adaptive capacity of life itself often prevents the tipping of emergent properties of ecosystems by reorganizing biodiversity.

Measuring biodiversity change

No single biodiversity metric can reflect the multiple dimensions of biological variation, which encompass genes, species, their evolution, functions, and ecosystems (Pereira et al. 2013; Antonelli 2022). Biological diversity has traditionally been measured in taxonomic units, e.g., the number of different species (or taxa if identification to the species level is not possible) in an ecological community, although it comprises many other dimensions. For example, the diversity of phenotypes and genets between individuals within and across species, the diversity of functional properties of individuals or species, the phylogenetic distance between species, the diverse ways species can interact in a community, or the diversity of community types across a land- or seascape (Antonelli 2022). These different dimensions do rarely inform each other, and even the best resolution of the taxonomic dimension will not necessarily reflect the change in other dimensions such as intraspecific, functional, phylogenetic, and interaction diversities (Naeem et al. 2016; Heydari et al. 2020; Gaüzère et al. 2022). Each of these dimensions provides unique and complementary insights into ecosystem structure, function, and resilience (Naeem et al. 2016).

While monitoring programs are often well equipped to address the identity and abundance of taxa (allowing for taxonomic diversity calculations), the other aspects require additional measurements (e.g., traits for functional biodiversity) or information (e.g., evolutionary history of species). The added effort pays off in revealing not just the presence of species (taxonomic diversity) but also their roles (functional diversity) or their evolutionary relationships (phylogenetic diversity). Interaction diversity builds upon these previous facets by examining how species interact within the ecosystem (Gaüzère et al. 2022), which again requires additional effort by measuring or inferring interaction types and strengths. By combining taxonomic, functional, phylogenetic, and interaction diversities, we may gain a more comprehensive understanding of ecosystem complexity and resilience (Ceron et al. 2023). At the same time, the complexity may result in an unsurmountable pile of biodiversity information that rather impedes than motivates action.

As most of these metrics are ultimately based on sampling a community and identifying taxa and genets (and their traits and interactions), there is the fundamental issue of effort dependency of biodiversity assessment. Most biodiversity researchers and practitioners are fully aware of this (see Box 1 in the "Appendix"), but the difference from almost all other measurements we do in the environment is so fundamental that it warrants attention. In most cases, different measurements of the same physical quantity, e.g., temperature, by different devices or even different methods, show some variation around a true mean. By contrast, the "true" biodiversity of an assemblage remains largely unknown (as we do not sample and identify all individuals in a habitat); but even more upsetting, changing effort and method give us completely different approximations of this truth (Box 1 in the "Appendix").

Aiming at a comprehensive monitoring and assessment of biodiversity change, we recommend considering how natural and anthropogenic drivers can alter biodiversity in the first place. Thus, it is wise to derive frameworks that cover all three major dimensions of change: the extent of an ecosystem, the abundance (or biomass) of organisms in the ecosystem, and the diversity of organisms comprised by this abundance and biomass (Fig. 2). Each of these dimensions warrants specific attention as their dynamics can be independent from each other—but also highly correlated. More mechanistic insights can be gained by integrating all three dimensions in the assessment of biodiversity.

Ecosystem extent

The first dimension of change is the extent of an ecosystem or natural habitat type, e.g., the area covered by reefs or the size of seagrass meadows (Fig. 2). This extent can decrease by land- or sea-use change (e.g., bottom trawling, nutrient intrusions, construction of offshore windfarms), which reduces the area of intact habitat and fragments the smaller remnants with varying degrees of connectivity. Extent can also increase when natural habitats are restored or missing foundation species recover or are transplanted. The appropriate observation measure is sensing the dimensions of the habitat, by direct observation, ship-based instruments, or remotely with satellites and planes. Recording the area or volume, which laudably is on absolute scales of area or volume, does not convey information on the quality of the habitat and the diversity of organisms within the system. Thus, while the extent of a habitat may change, the next two dimensions of change (abundance and biodiversity) can potentially be stable in the remaining habitat area.

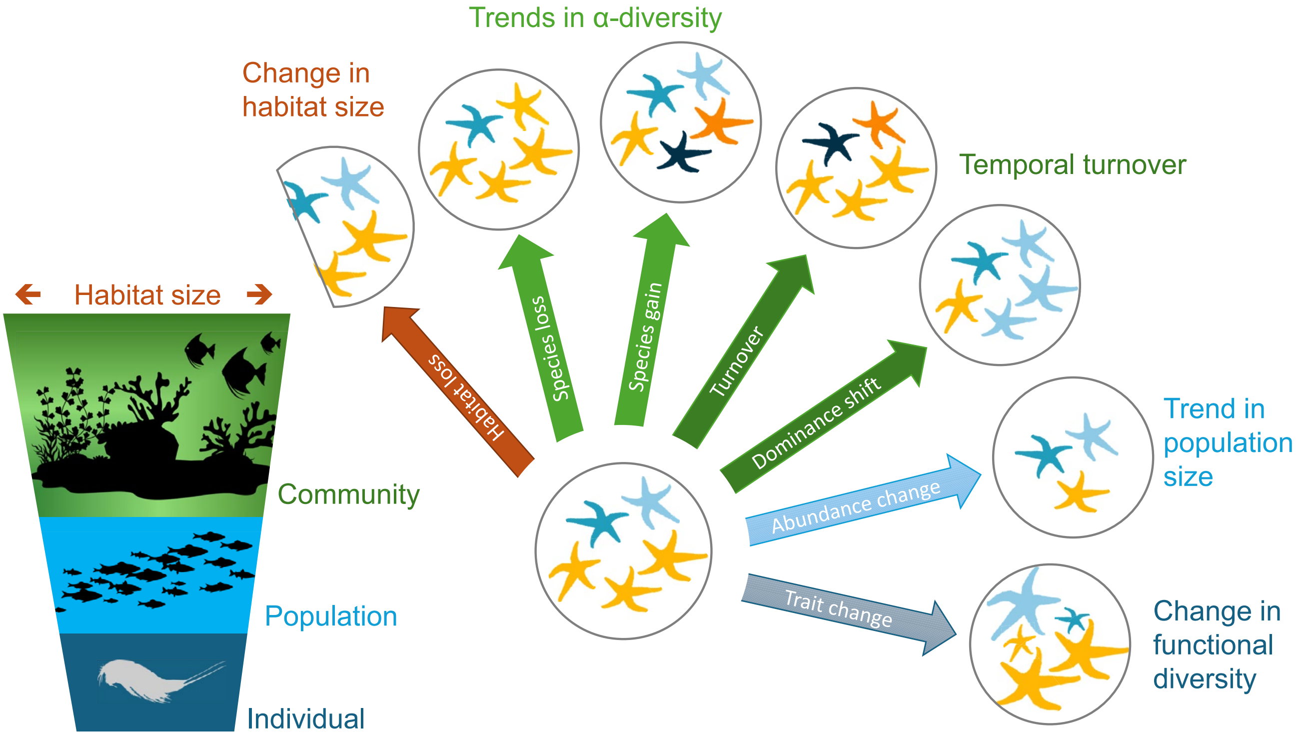


Fig. 2 Dimensions of biodiversity change and their association to organizational scales of biodiversity. Individuals can change in their traits and thereby alter the functional diversity; populations can increase or decrease in their abundance or biomass. Communities can lose or gain taxa or shift the composition of their component taxa. Additionally, the extent of a habitat (as area or volume) can change

By definition, extent is often relevant only for ecosystems defined by sessile organisms.

Abundance

The second dimension is the size of populations of single species or entire communities (Fig. 2). While biodiversity metrics ask how many different taxa there are (and how different they are), this dimension focuses on the much simpler question of how many individual organisms there are. These data often come from repeated point monitoring with a pre-defined approach (method, scale of time and space) to deliver estimates of population density (individuals or genets per m^2 or per m^3) or biomass (g per m^2 or m^3). Again, a change in the absolute number or biomass of individuals does not necessarily mean that the biodiversity of the community has changed. Population size can decrease or increase for numerous reasons, such that it is not always the decrease that is of concern. The decline in bird populations described in Silent Spring (Carson 1962) or the reduction in insect biomass over time observed even in nature protection areas (Hallmann et al. 2017) has been a hallmark of the scientific and public debate on biodiversity loss. The IUCN Red List approach provides important information on the decline or recovery of target species. However, increases in abundance or biomass are not always a sign of healthy biodiversity. For example, the increasing phytoplankton biomass or the presence of harmful algal blooms in coastal areas can be the consequence of eutrophication and thus a sign of declining environmental conditions (van Beusekom et al. 2019). The interpretation of population sizes is thus highly context dependent, and when limited to vulnerable or threatened species, leaves parts of the biodiversity out of focus.

Compositional turnover

The change of community composition can be analyzed using three major approaches (Fig. 4 in the “Appendix”), each of which can be done with different emphasis on dominant species. The first approach is to use simple temporal regressions of α-metrics, which have been performed widely (Vellend et al. 2013; Dornelas et al. 2014; Elahi et al. 2015), disputed (Gonzalez et al. 2016; Vellend et al. 2017; Cardinale et al. 2018), and consolidated (Hillebrand et al. 2018; Blowes et al. 2019; Chase et al. 2019). A major takeaway from this scientific discussion is that temporal trends of α-diversity offer only a limited view on the amount of change, as they focus only on the net balance of colonization and extirpation. For example, if four coral species are replaced by four algal species, no net change in numerical richness is observed, but a qualitatively completely different ecosystem emerges. Additionally, the interpretation of the trends is much less straightforward than often assumed, as positive temporal slopes might reflect actual increases in α-diversity, but they might also arise if colonization is faster than extinction, a transient imbalance that can last for decades (Kuczynski et al. 2023). It is thus highly recommended to amend such analyses always by horizontal or hierarchical turnover analyses (Box 1 in the "Appendix"). Horizontal metrics allow detecting immediate changes (i.e., an unusual change in composition above the mean turnover between years) and the reduction in turnover when a community equilibrates. Hierarchical metrics allow quantifying the overall amount of change and are especially suited for capturing the accumulated history of change. The choice between addressing horizontal or hierarchical turnover is rather a question of the purpose of the monitoring than of metrics (Box 1 in the "Appendix").

Summarizing this section, we recommend planning biodiversity assessments by mapping the potential impacts (do natural or anthropogenic drivers affect the quantity or the quality of a certain environment type) with the potential corresponding responses (extent, population size, composition).

We highlight two caveats. First, point monitoring as a basis of population and composition estimates requires careful consideration of how representative these points are. Most monitoring started and continued for a reason. Sampling points are not regularly or randomly dispersed across the land- or seascape, but often cluster around places of interest such as nature protection areas or high impact areas. The former provides a positive selection bias for high quality, biodiverse habitats and might overestimate the amount of negative trends by statistical convergence to the mean (Mentges et al. 2020). On the other end of the bias spectrum, monitoring ignores the most negative biodiversity trends when it stops after converting areas, e.g., when turning a park into a parking lot.

The second caveat, which follows from the entire section, is that there is no value of α-, β-, or γ-diversity that per se is an indicator of healthiness or good environmental status. The effort dependency of all these metrics boils down to the statement that they can only be interpreted in their own temporal context, i.e., monitoring can provide trends (and interpret these as positive or negative), but cannot provide absolute thresholds for good or bad biodiversity.

The molecular revolution to the rescue—soon

Most of our understanding of biodiversity and its change stems from decades of morphology-based taxonomic work on samples, organisms, and voucher specimens. In addition to the issues outlined above, the taxonomic resolution constraints of visual identification exclude much of the tree of life, as many organisms are either too small or lack distinguishing morphological features for reliable identification. This limitation affects not only archaea and bacteria but also unicellular eukaryotes and metazoans such as meiofauna, all of which remain highly underrepresented in monitoring efforts. Moreover, a large fraction of biodiversity may consist of "sibling" or "cryptic" taxa, where organisms that appear to belong to a single species based on morphology are in fact genetically distinct, reproductively isolated entities (Hebert et al. 2004; Bickford et al. 2007). For bacteria and archaea, the morphological and physiological description requires culturing, but their culturable fraction is minor compared to what is found in the environment (Staley and Konopka 1985).

Thus, even when using all the approaches detailed in the "Measuring biodiversity change" section, we systematically underestimate biodiversity. The advancement of molecular methods such as DNA barcoding of single species, metabarcoding of whole communities, or the analysis of environmental DNA (eDNA, i.e., genetic traces of living organisms from their environment) promises to facilitate the assessment of this "unseen" biodiversity (Taberlet et al. 2018; Compson et al. 2020; Zhang et al. 2023). Moreover, not only can we analyze nucleic acids, but also proteins and metabolites, which give biodiversity science the potential to transcend different scales of biological variation, from genes to populations to communities and higher levels of organization (Eren and Banfield 2024). Rapid technological advances and decreasing costs have enabled the broader and more standardized use of genetic tools, while major improvements in bioinformatic algorithms now allow the efficient processing of these large datasets to inform biodiversity science and assessment. Moreover, molecular methods provide unprecedented insights into functionality by enabling the reconstruction of genomes (either from single organisms (or even cells) or as metagenome-assembled genomes from bulk samples) as well as the analysis of transcriptomes and metabolomes. The papers cited in this paragraph provide details on methods, their promises, and their limits. In the context of our paper, we focus on the question of whether the molecular revolution has alleviated some of the issues described in the previous sections. The same question applies to other automatic or semi-automatic methods for capturing biodiversity, such as acoustic monitoring using microphones and hydrophones or optical monitoring employing camera traps and flow cytometers.

Foremost, molecular methods capture more of the "rare and invisible" fraction of the biodiversity that is easily missed by traditional sampling, while also increasing temporal coverage of biodiversity in hard-to-reach areas such as polar regions, the deep sea, and deep subsurface habitats. Molecular genetic approaches for the identification of communities can be directly applied to whole samples (so-called bulk samples), including net catches (e.g., traps, nets, dredges), water, or sediment. However, the outputs of such analyses are not traditional taxonomic units, but rather amplicon sequence variants (ASVs) or molecular operational taxonomic units (MOTUs). This molecular information can be assigned to species, but only when comprehensive sequence reference databases are available for the corresponding ecosystem. Consequently, the success of species identification through methods such as eDNA depends critically on both the quality of the sequence databases and the quantity of water, sediment, or air filtered, factors that can have an enormous influence on the biodiversity estimates (Ohnesorge et al. 2023).

Given the rapid advancement of methods and resolution, the comparability of historical to molecular datasets remains a central challenge—particularly since biodiversity data require temporal context to be interpretable. Compared to morphology-based taxonomy, molecular methods often yield much higher estimates of richness, potentially discerning "unrecognized species" through genetic delineations of morphologically identical specimens (Hebert et al. 2004; Witt et al. 2006; Bickford et al. 2007). However, these differences may also reflect a non-negligible proportion of misclassifications, including false positives and false negatives (Doi et al. 2019; Burian et al. 2021).

Using DNA collected directly from the environment, particularly from water or air, provides a recent snapshot of biodiversity from a larger region through the genetic remains of the species occurring in or passing through the respective ecosystem. These data thus combine an estimate of the regional diversity (γ) with a local (α) diversity sample, thereby minimizing the limits imposed by small sampling units. However, molecular data cannot easily inform us about the extent of the habitat and the absolute size of a population. This challenge is especially pronounced for multicellular organisms, where biases introduced during sequencing or sample preparation are difficult to quantify (van der Loos and Nijland 2021).

Dominance-weighted approaches (e.g., ENS, ENC; see Box 1 in the "Appendix") are also possible with molecular data, using the number of sequence reads generated from a community sample as an abundance proxy, as has been demonstrated for zooplankton (Ershova et al. 2023). However, such read counts do not represent the actual number of individuals given the vast differences in gene copy numbers among taxa. Yet, it is important to acknowledge that traditional estimates of relative abundance are also not entirely "objective" since counts of individuals favor the smaller species, biomass favors the larger species with heavy skeletons, and carbon favors species with substantial structural tissues.

The molecular revolution holds great promise and already transforms our understanding of biodiversity, but it is not a simple cure for the complexity syndrome of biodiversity assessments. Taxonomic accuracy depends on primer choice and the quality of the available sequence databases, the detection of rare taxa depends on sample size and sequencing depth, and misclassifications remain possible just as in morphology-based taxonomic monitoring. For the near future, biodiversity monitoring will therefore require mixed-method approaches, and it is advisable to apply different methods in parallel.

Recognizing and integrating emotions in biodiversity conservation

Despite numerous local, international, and global efforts for biodiversity conservation, there remains a stark gap between actions taken (on paper) and the level of implementation on the ground (Grorud-Colvert et al. 2021; Relano and Pauly 2023). Two major narratives have been proposed to explain this disconnect. The first claims that we (as decision supporters and makers) "do not know enough" for taking action, referring to a lack of biodiversity understanding and the inaccessibility of existing knowledge to practitioners (Roche et al. 2022). While this may be true for specific cases involving complex drivers and ecosystems, this narrative does not hold at the broader levels at which action is urgently needed (see Box 1 in the "Appendix"). For decades, research has clearly shown how biodiversity responds to major classes of anthropogenic pressures (e.g. pollution or overfishing), and how those impacts could be mitigated, with new findings largely reinforcing this established understanding (see the consistent conclusions over 20 years in Vitousek et al. 1997; Sala et al. 2000; IPBES 2019; Jaureguiberry et al. 2022). In addition, some scepticism is in place as to whether filling knowledge gaps would increase the willingness to act, the more so as adding facts rarely changes minds (Toomey 2023).

The second narrative proposes that we (as scientists and practitioners) "do not communicate (well) enough" on biodiversity's importance, a challenge often referred to as "mainstreaming biodiversity" (Runhaar et al. 2024). The problem, according to this argument, is less about lacking knowledge and more about not making that knowledge visible, relevant, reliable, and actionable. Despite growing awareness, biodiversity consequently struggles to gain traction in broader policy and societal debates. The impacts of climate change on humans are so ubiquitous that heat waves, drought, extreme storms, floods, or rainfall can be perceived, independent of where we live. Surely, biodiversity loss is viscerally and affectively felt not only by research scientists, policy makers, and charities but also by citizens (Levi and

Peters 2024). In contrast to climate impacts, however, perceptions of biodiversity loss are strongly shaped by whether people live in urban or rural areas, near the coast or in mountainous regions. This is especially true for marine ecosystems, notwithstanding the fact that many human livelihoods depend on marine biodiversity (fisheries, tourism, shipping). As most people do not engage with a marine environment every day, biodiversity loss is often "out of sight and out of mind" (Franke et al. 2023).

The communication of the consequences of biodiversity change often follows either of two routes, hinging upon tangible, emotionally loaded key species of concern or trying to create an overall impression of biodiversity's importance, e.g. by the Nature Contribution to People (NCP) framework (IPBES 2019, 2022). Yet, both concepts have so far been ineffective. A focus on charismatic species does not only distort ecological realities (Skibins and Powell 2013) but even leads to counterproductive human behavior, e.g. when wildlife images on social media do not inspire wonder but encourage risky behavior, both for the wildlife and humans, in pursuit of ever more dramatic content (Davis et al. 2024). When proposing abstract frameworks such as NCP, biodiversity again appears as a hyperobject (Morton 2013) that is too complex and fragmented to be fully grasped. It may require simple visualizations as in Fig. 1 in order to provide an imagination of the problem at hand.

We propose a third narrative connecting the previous two, that the emotional value of the diversity of life is not acknowledged enough, especially in Western societies, science and policy making. It is, for example, only in recent years that Western academia has experienced the 'oceanic turn' and begun to recognize concepts such as 'more-than-human' and 'rights of nature' that many communities have advocated for and lived with for thousands of years (Todd 2016). Indeed, emotions are increasingly recognized as critical facilitators of learning and behavior (Li and Monroe 2017; Sánchez-Jiménez et al. 2021), but their central role in biodiversity conservation has been systematically downplayed and delegitimized (Buijs and Lawrence 2013), both within Western science and in the science-society dialogue.

The oceans have enormous emotional significance that extends to how they are experienced, ascribed meaning, and managed (Kearns and Collins 2012). Taking emotions seriously in marine governance processes is essential for developing policies and interventions that are not only environmentally sustainable but also socially just (Kearns and Collins 2012; Pafi 2021; McKinley et al. 2023). McKinley et al. (2023) introduced the concept of "emoceans" as "emotional connections to the ocean," which are "about how a person feels and emotionally responds when they think about, are near/within, or consider issues relating to the ocean, coasts and seas". Emotional relationships to the oceans are often at the core of citizen science and restoration projects, the more so as these often are started and maintained by individuals and their drive or urge to improve the status or trends of biodiversity (Lotze 2020). The rise of marine citizen science projects (Garcia-Soto et al. 2017) and projects aiming at knowledge co-production beyond Western sciences (Caldeira et al. 2025) provide a new quality of engagement with marine biodiversity. While these projects are often aware of differences in cultural and social positioning, they rarely explicitly address emotions as a part of the motivation and relation of different project partners.

One approach to capture "emoceans" such as attachment, hope, and grief in ocean policy and conservation efforts is to leverage advancements in environmental psychology. Metrics such as the Connectedness to Nature Scale (Mayer and Frantz 2004), NR-6 (Nisbet and Zelenski 2013), or the extended Inclusion of Nature in Self scale (Martin and Czellar 2016) capture the relations between humans and nature and have been used in conservation and sustainability research. These tools have the advantage that their outcomes can be followed over time, and they can be compared to biodiversity metrics in similar quantitative dimensions.

However, knowing how humans relate to nature as a combination of social, cultural, emotional, and psychological factors is not in itself sufficient to embrace the emotional and subjective dimensions of the oceans and their biodiversity. Major hindrances include (i) the singular emphasis on scientific, 'evidence-based' policy, (ii) mismatches in social and scientific policy objectives, (iii) challenges in measuring and integrating 'non-scientific' knowledge, and (iv) the tendency to perceive such knowledge as 'feminine' and hence less legitimate and credible than traditionally masculine bureaucratic approaches (Gissi et al. 2018; Peters 2020).

Scientists, decision makers, but also the general public must recognize that subjective emotion and scientific rigor are not mutually exclusive. Indeed, being aware of (rather than ignoring, dismissing, or denying the existence of) our own subjectivities further strengthens the robustness of the science we do (Levi 2025). Scientists are typically perceived (and largely define themselves) as objective observers of natural or societal phenomena (Rietig 2014), but already the choice of studied organismal group is often led by emotions and valuation of aesthetic or functional aspects of these organisms. Acknowledging positionality and using it as a reflexive practice to understand how our worldviews and privileges as scientists influence the research process can help mitigate oppressive practices and ensure that research honors diverse knowledge systems (Murray et al. 2025). Indeed, since emotions are never absent from the research process, reflexivity further allows us to recognize how our emotions as researchers (and as people) influence the decisions we make, the science we do, and the recommendations we give (Levi

2025). Combining our emotional and rational motivation for research can transform ocean research in meaningful ways (Lotze 2020).

At the same time, research and the science-policy dialogue need to avoid the naïve narrative that biodiversity inherently is always positive for humans, which neither reflects the complexity of biodiversity change nor the complexity of emotions towards nature. Cultural perceptions, beliefs, and historical experiences shape how humans relate to nature (Fritts et al. 2003; Jacobs and McConnell 2022). Wildlife conservationists continuously cope with the fact that the same large predator species may evoke fear, anger, or disgust in some but happiness or admiration in others (Castillo-Huitrón et al. 2020). Positive attitudes towards biodiversity may also decline when conservation measures conflict with socioeconomic development (Kellert 1985; Metrick and Weitzman 1998). At the same time, fascination for biodiversity can also result in emotions such as grief, loss, and existential unease in the face of environmental degradation. Albrecht (2019) coined the term "solastalgia" for the distress felt when a once-familiar environment no longer provides solace and differentiated it from nostalgia (longing for an 'unspoiled' past).

Similar emotional complexity arises around non-native (non-indigenous, alien) species introduced outside their natural, historical range either deliberately or accidentally by human activity. The fear of significant ecological, economic, or health-related harm leads to negative emotions that often justify extreme and costly management measures. Even within the biodiversity research community, a deep divide persists. Some argue that non-native species contribute to overall biodiversity, particularly in urban or altered ecosystems, by filling vacant ecological roles (Bartomeus et al. 2008; Kowarik 2011). Others caution that including non-native species in biodiversity metrics may mask declines in native species, mislead conservation priorities, and misrepresent ecosystem health (Simberloff et al. 2024).

We argue here that the recognition and inclusion of biodiversity's emotional value is essential for successful and sustainable conservation efforts as well as natural resource governance. Only by acknowledging the inseparability of humans and nature, and by fostering more nuanced and meaningful emotional connections to biodiversity, can the motivation for action for biodiversity live up to the vast, complex, and only partially perceptible nature of biodiversity as a hyperobject (Morton 2013). Hence, it is crucial to shift towards modes of environmental governance that reflect and are guided by the emotional complexities of human–nature relations (Levi and Peters 2024). As Levi and Peters (2024) assert, "[r]ecognizing emotion thus becomes not simply a matter of ticking a box but a matter of politics and of justice."

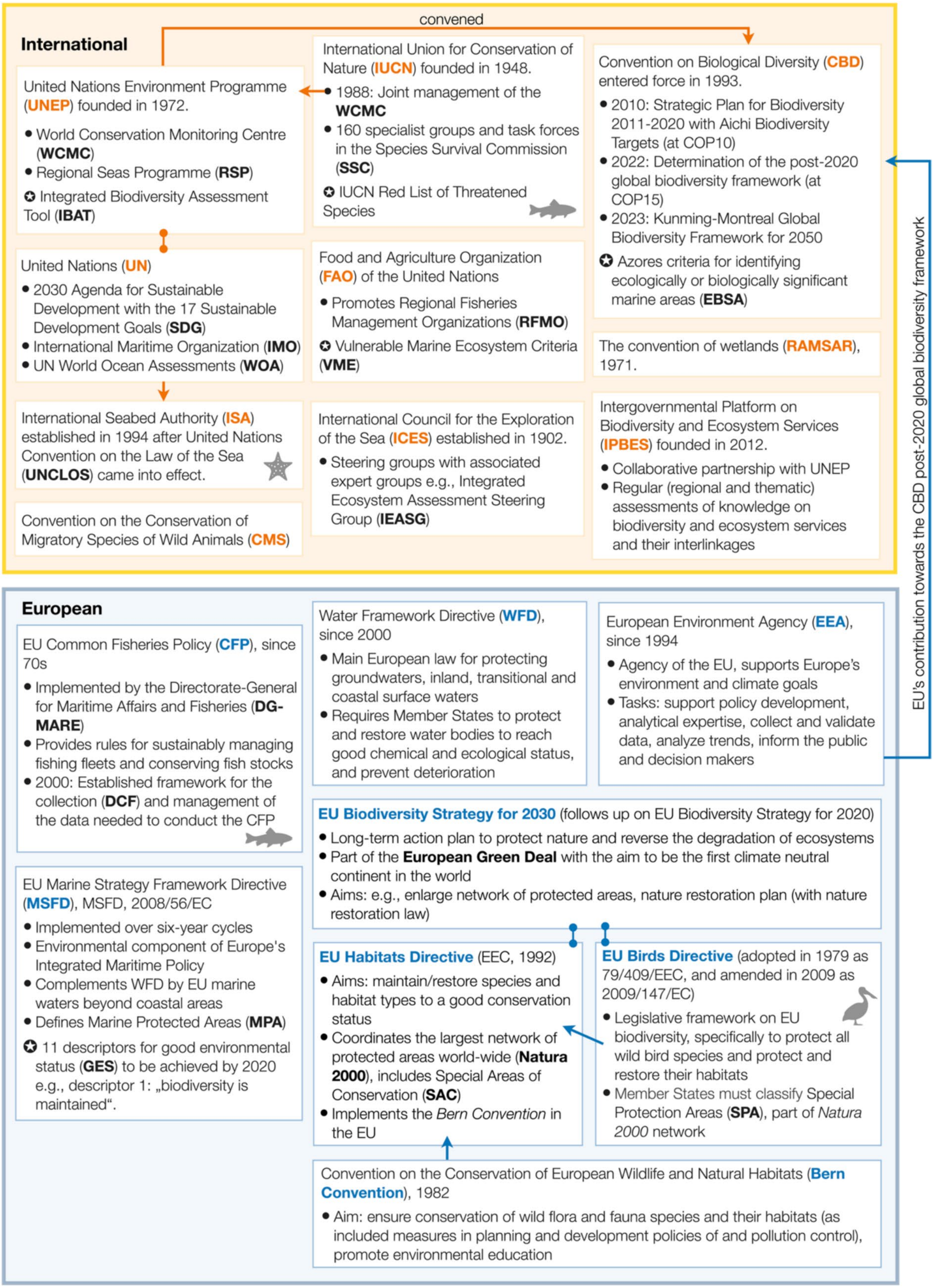
Guiding principles for biodiversity policies and target setting

To effectively address biodiversity loss, policies and targets must be grounded in a nuanced understanding of biodiversity change and its drivers. Biodiversity policies therefore need to match the multilayered complexity of their subject, as illustrated by the European policy landscape governing marine biodiversity assessment and management (Fig. 3). Policies and targets have been agreed upon in a joint and binding Marine Strategy Framework Directive (EU 2017), which is exemplary in terms of international collaboration. However, complexity arises because the different implementation spans spatial, organizational, and legal scales at the regional level (Fig. 3, lower panel). At the same time, the European framework needs to be part of even broader, global contexts (Fig. 3, upper panel).

In this seemingly irreversible entanglement of policy layers, the major question arises whether the actual implementation of the framework requires indicators that need the data categories mentioned earlier ("Measuring biodiversity change" section and Box 1 in the "Appendix"). We analyzed this question with a systematic literature review (Box 2 in the "Appendix"). While there are certainly gaps, we find a broad coverage of organism groups and ecosystem types in the literature (Box 2 in the "Appendix"). Rather than the lack of data, a major impediment seems to be the lack of transfer into operational indicators (Hillebrand et al. 2025).

This tenet seems to hold even at the global scale. Dajka et al. (2025) recently analyzed the targets formulated in the Global Biodiversity Framework (CBD 2022) and found strong alignment of these targets to the scientific knowledge base and the variety of facets of biodiversity (Pereira et al. 2013). Their analysis and "Box 2" in the "Appendix" indicate that the lack of knowledge is not the prime issue for the inaction on biodiversity. However, notable blind spots persist, particularly with regard to 'invisible biodiversity' such as unicellular organisms, meiofauna, and genetic diversity.

Dajka et al. (2025) additionally identified a noticeable shift in target formulation, away from primary biodiversity targets towards secondary facets such as ecosystem function and structure. While this trend can be more widely observed in environmental debates, it risks losing the connection to what is at stake in terms of fascination, singularity, and importance. More broadly, the conceptualization of biodiversity policies, their targets, indicators, and assessment would benefit from systematically accounting for the dimensions of biodiversity change (section "Measuring biodiversity change") and from leveraging emerging molecular approaches while maintaining continuity with



convened

International

United Nations Environment Programme (**UNEP**) founded in 1972.
- World Conservation Monitoring Centre (**WCMC**)
- Regional Seas Programme (**RSP**)

✪ Integrated Biodiversity Assessment Tool (**IBAT**)

International Union for Conservation of Nature (**IUCN**) founded in 1948.
- 1988: Joint management of the **WCMC**
- 160 specialist groups and task forces in the Species Survival Commission (**SSC**)

✪ IUCN Red List of Threatened Species

Convention on Biological Diversity (**CBD**) entered force in 1993.
- 2010: Strategic Plan for Biodiversity 2011-2020 with Aichi Biodiversity Targets (at COP10)
- 2022: Determination of the post-2020 global biodiversity framework (at COP15)
- 2023: Kunming-Montreal Global Biodiversity Framework for 2050

✪ Azores criteria for identifying ecologically or biologically significant marine areas (**EBSA**)

United Nations (**UN**)
- 2030 Agenda for Sustainable Development with the 17 Sustainable Development Goals (**SDG**)
- International Maritime Organization (**IMO**)
- UN World Ocean Assessments (**WOA**)

Food and Agriculture Organization (**FAO**) of the United Nations
- Promotes Regional Fisheries Management Organizations (**RFMO**)

✪ Vulnerable Marine Ecosystem Criteria (**VME**)

The convention of wetlands (**RAMSAR**), 1971.

International Seabed Authority (**ISA**) established in 1994 after United Nations Convention on the Law of the Sea (**UNCLOS**) came into effect.

International Council for the Exploration of the Sea (**ICES**) established in 1902.
- Steering groups with associated expert groups e.g., Integrated Ecosystem Assessment Steering Group (**IEASG**)

Intergovernmental Platform on Biodiversity and Ecosystem Services (**IPBES**) founded in 2012.
- Collaborative partnership with UNEP
- Regular (regional and thematic) assessments of knowledge on biodiversity and ecosystem services and their interlinkages

Convention on the Conservation of Migratory Species of Wild Animals (**CMS**)

EU's contribution towards the CBD post-2020 global biodiversity framework

European

EU Common Fisheries Policy (**CFP**), since 70s
- Implemented by the Directorate-General for Maritime Affairs and Fisheries (**DG-MARE**)
- Provides rules for sustainably managing fishing fleets and conserving fish stocks
- 2000: Established framework for the collection (**DCF**) and management of the data needed to conduct the CFP

Water Framework Directive (**WFD**), since 2000
- Main European law for protecting groundwaters, inland, transitional and coastal surface waters
- Requires Member States to protect and restore water bodies to reach good chemical and ecological status, and prevent deterioration

European Environment Agency (**EEA**), since 1994
- Agency of the EU, supports Europe's environment and climate goals
- Tasks: support policy development, analytical expertise, collect and validate data, analyze trends, inform the public and decision makers

EU Biodiversity Strategy for 2030 (follows up on EU Biodiversity Strategy for 2020)
- Long-term action plan to protect nature and reverse the degradation of ecosystems
- Part of the **European Green Deal** with the aim to be the first climate neutral continent in the world
- Aims: e.g., enlarge network of protected areas, nature restoration plan (with nature restoration law)

EU Marine Strategy Framework Directive (**MSFD**), MSFD, 2008/56/EC
- Implemented over six-year cycles
- Environmental component of Europe's Integrated Maritime Policy
- Complements WFD by EU marine waters beyond coastal areas
- Defines Marine Protected Areas (**MPA**)

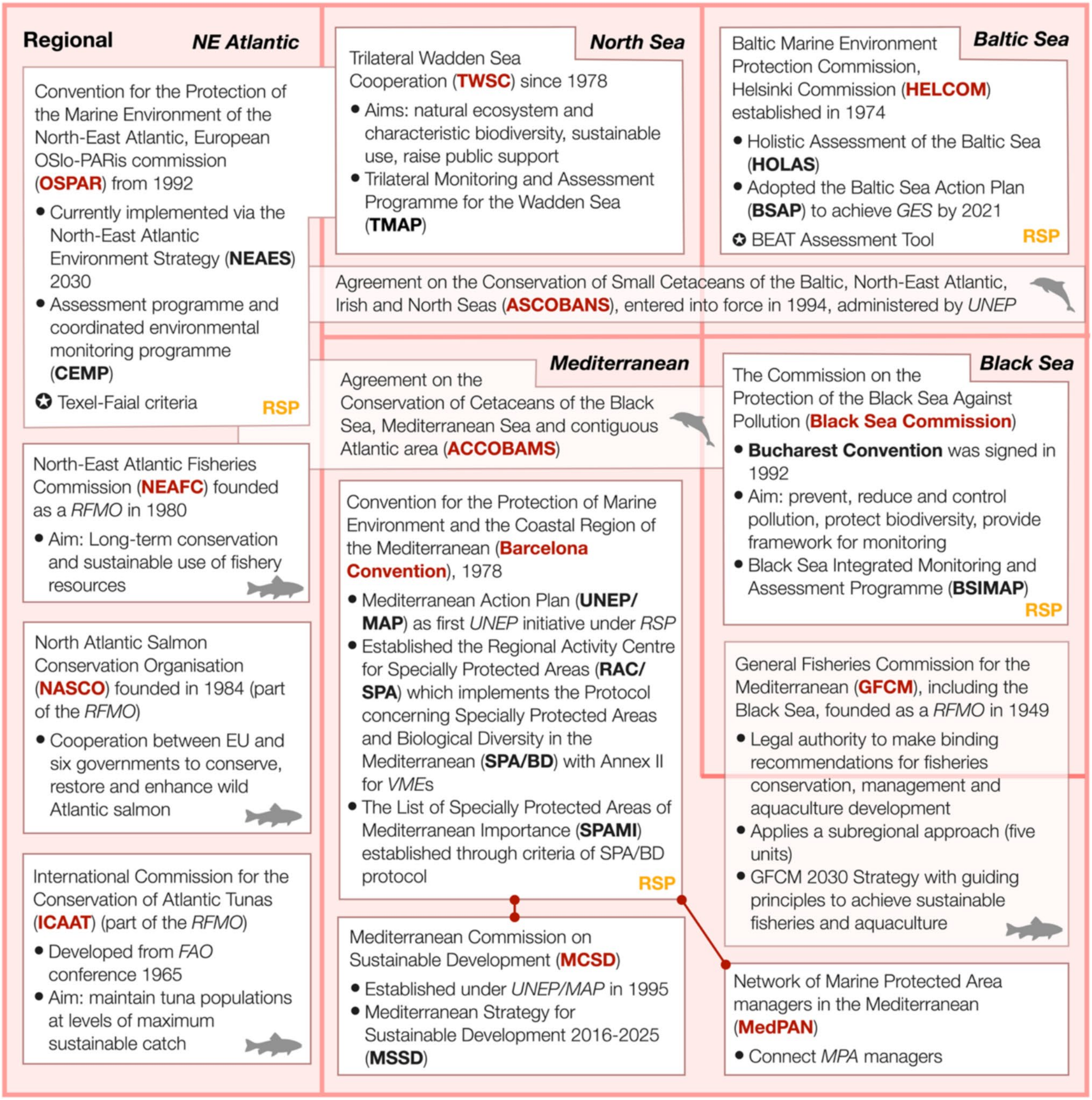
✪ 11 descriptors for good environmental status (**GES**) to be achieved by 2020 e.g., descriptor 1: „biodiversity is maintained".

EU Habitats Directive (EEC, 1992)
- Aims: maintain/restore species and habitat types to a good conservation status
- Coordinates the largest network of protected areas world-wide (**Natura 2000**), includes Special Areas of Conservation (**SAC**)
- Implements the *Bern Convention* in the EU

EU Birds Directive (adopted in 1979 as 79/409/EEC, and amended in 2009 as 2009/147/EC)
- Legislative framework on EU biodiversity, specifically to protect all wild bird species and protect and restore their habitats
- Member States must classify Special Protection Areas (**SPA**), part of *Natura 2000* network

Convention on the Conservation of European Wildlife and Natural Habitats (**Bern Convention**), 1982
- Aim: ensure conservation of wild flora and fauna species and their habitats (as included measures in planning and development policies of and pollution control), promote environmental education

Fig. 3 Structured overview of the biodiversity policy framework landscape on international, European and regional level. Symbols indicate the focus of the respective framework (fish, birds, and benthos). The star marks distinct biodiversity assessment tools or comprehensive sets of criteria (see Box 2 in the "Appendix")



Convention for the Protection of the Marine Environment of the North-East Atlantic, European OSlo-PARis commission (**OSPAR**) from 1992

- Currently implemented via the North-East Atlantic Environment Strategy (**NEAES**) 2030
- Assessment programme and coordinated environmental monitoring programme (**CEMP**)

✪ Texel-Faial criteria **RSP**

North-East Atlantic Fisheries Commission (**NEAFC**) founded as a *RFMO* in 1980

- Aim: Long-term conservation and sustainable use of fishery resources

North Atlantic Salmon Conservation Organisation (**NASCO**) founded in 1984 (part of the *RFMO*)

- Cooperation between EU and six governments to conserve, restore and enhance wild Atlantic salmon

International Commission for the Conservation of Atlantic Tunas (**ICAAT**) (part of the *RFMO*)

- Developed from *FAO* conference 1965
- Aim: maintain tuna populations at levels of maximum sustainable catch

Trilateral Wadden Sea Cooperation (**TWSC**) since 1978

- Aims: natural ecosystem and characteristic biodiversity, sustainable use, raise public support
- Trilateral Monitoring and Assessment Programme for the Wadden Sea (**TMAP**)

Baltic Marine Environment Protection Commission, Helsinki Commission (**HELCOM**) established in 1974

- Holistic Assessment of the Baltic Sea (**HOLAS**)
- Adopted the Baltic Sea Action Plan (**BSAP**) to achieve *GES* by 2021

✪ BEAT Assessment Tool **RSP**

Agreement on the Conservation of Small Cetaceans of the Baltic, North-East Atlantic, Irish and North Seas (**ASCOBANS**), entered into force in 1994, administered by *UNEP*

Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and contiguous Atlantic area (**ACCOBAMS**)

Convention for the Protection of Marine Environment and the Coastal Region of the Mediterranean (**Barcelona Convention**), 1978

- Mediterranean Action Plan (**UNEP/MAP**) as first *UNEP* initiative under *RSP*
- Established the Regional Activity Centre for Specially Protected Areas (**RAC/SPA**) which implements the Protocol concerning Specially Protected Areas and Biological Diversity in the Mediterranean (**SPA/BD**) with Annex II for *VMEs*
- The List of Specially Protected Areas of Mediterranean Importance (**SPAMI**) established through criteria of SPA/BD protocol

RSP

Mediterranean Commission on Sustainable Development (**MCSD**)

- Established under *UNEP/MAP* in 1995
- Mediterranean Strategy for Sustainable Development 2016-2025 (**MSSD**)

The Commission on the Protection of the Black Sea Against Pollution (**Black Sea Commission**)

- **Bucharest Convention** was signed in 1992
- Aim: prevent, reduce and control pollution, protect biodiversity, provide framework for monitoring
- Black Sea Integrated Monitoring and Assessment Programme (**BSIMAP**)

RSP

General Fisheries Commission for the Mediterranean (**GFCM**), including the Black Sea, founded as a *RFMO* in 1949

- Legal authority to make binding recommendations for fisheries conservation, management and aquaculture development
- Applies a subregional approach (five units)
- GFCM 2030 Strategy with guiding principles to achieve sustainable fisheries and aquaculture

Network of Marine Protected Area managers in the Mediterranean (**MedPAN**)

- Connect *MPA* managers

Fig. 3 (continued)

historical time series (section “The molecular revolution to the rescue—soon”). In the following, we discuss three major gaps that must be addressed to improve biodiversity governance: (i) aligning action across scales, (ii) embracing the dynamic, non-equilibrium nature of ecosystems, and (iii) incorporating the emotional dimension into assessment and management.

Scale is a fundamental, yet often underappreciated, dimension of biodiversity policy and management (Du Toit 2010). While biodiversity goals are typically formulated at national or international levels, their implementation inevitably takes place at local or regional scales (Agardy 2005). This misalignment can result in unintended trade-offs, where measures that enhance biodiversity locally may inadvertently undermine broader conservation objectives, and vice versa. Habitat fragmentation can increase species richness and evenness at small scales by creating a mosaic of diverse microhabitats. However, such localized gains pose a threat to biodiversity at larger scales, as species that rely on extensive, connected habitats are excluded (Olff and Ritchie 2002). Likewise, globally threatened species may be locally abundant, leading to potential conflicts over management priorities (Boyd et al. 2008). Effective biodiversity policy must therefore be grounded in a clear understanding of the scale at which biodiversity is to be managed (Paloniemi et al. 2012). This includes evaluating how management actions at one scale affect biodiversity targets at another. Specifically, policies aimed at preserving or enhancing local biodiversity should be aligned with global biodiversity goals to avoid counterproductive outcomes (Norton and Ulanowicz 1996). Integrating management strategies across scales is thus essential to ensure coherence between local implementation and overarching strategic objectives.

Crucially, cross-scale biodiversity policies must recognize the inherently dynamic and non-equilibrium nature of ecosystems by allowing for permeable, flexible boundaries that permit habitat connectivity, species migration, and ecosystem succession. Yet, most biodiversity policies and targets exist in a static mindset, best reflected by the "Good Environmental Status" mandate for European seas (EU 2017). Likewise, measures to restore nature often are designed according to "engineering" principles, assuming a reversibility of biodiversity (dis-)assembly. This is not how biodiversity works! For details, we point to the literature on the historical contingency of community assembly (Fukami 2015) and the asynchrony between colonization and extinction events in time series (Jackson and Sax 2010; Kuczynski et al. 2023). The take-away is that biodiversity is constantly in transient dynamics, spatially and temporally, which makes its ability to absorb or respond to environmental change the functionally most important aspect of biodiversity. Independent of whether this disequilibrium is captured as response diversity, resilience, or ecological stability (Elmqvist et al. 2003; Donohue et al. 2016; Ross et al. 2023), harnessing the biological adaptation and response potential must be the foremost biodiversity management task. Rather than managing ecosystems toward a predefined "Good Environmental Status," we need to manage good environmental dynamics: ongoing relational processes that recognize the plural capacities of ecological systems to adapt, co-evolve, and transform.

The final—and perhaps most central—gap in our current approach to biodiversity change and management is humans. Despite widespread recognition of anthropogenic drivers of biodiversity loss, most assessment frameworks remain largely biophysical in nature. Our current assessment frameworks are devoid of indicators of how people perceive, value, and emotionally relate to nature (Bennett et al. 2017, section "Recognizing and integrating emotions in biodiversity conservation"). This omission limits our ability to understand why biodiversity pressures persist or intensify, despite decades of scientific warnings and policy interventions (Bennett and Roth 2019). Without integrating human perceptions into our assessments, we miss critical levers for both behavioral change and effective policy design. The success of biodiversity mainstreaming thus hinges on how people understand their place in nature and the relevance of biodiversity to their everyday lives. Positive emotional connections to nature foster pro-environmental behavior, support conservation efforts, and enhance resilience in social-ecological systems. Ignoring these dimensions risks rendering biodiversity policy abstract and disconnected from lived realities. To move forward, biodiversity governance must embrace a more integrative approach that considers not only how biodiversity changes in response to us, but also how we change in response to it. This includes developing indicator frameworks that integrate metrics of human–nature relationships and metrics of biodiversity change ("Measuring biodiversity change" and "Recognizing and integrating emotions in biodiversity conservation" sections). Only by recognizing and making space for local and Indigenous knowledge systems, and creating spaces for dialogue, empathy, and collective agency, can we build the societal and political will, imagination, and responsibility needed to turn the tide of biodiversity loss.

Conclusions

Biodiversity change is both one of the most urgent and one of the most elusive dimensions of global change. Unlike climate change, which is captured in a few physical variables and felt directly through heatwaves, droughts, or floods, biodiversity loss is often hidden from view—especially in the marine environment. This intangibility poses a major challenge for scientists, who must not only detect and quantify biodiversity change but also communicate its significance in a way that resonates with society. In this regard, effective visualization tools (Fig. 1) and reflexive scientific approaches can help transform abstract numbers into intuitive, emotionally tangible representations that make the scale and consequences of biodiversity change understandable to policymakers and the public.

While changes in biodiversity are a fundamental component of evolution, they have become more noticeable during the Holocene. What is unprecedented today is the pace and global synchrony of these changes, driven by climate warming, habitat alteration, seascape use, and pollution. Rather than becoming lost in ever finer refinements of biodiversity metrics, the scientific community may need to embrace bold, integrative measures that are not only scientifically robust but also emotionally compelling. Only by making biodiversity change both visible and meaningful can we mobilize the societal and political will needed to bend the curve of biodiversity loss and ensure that the ocean's living systems retain their diversity, resilience, and capacity to inspire. It is time to recognize that biodiversity is not only a biological hyperobject, but also a lived human experience.

Appendix

Box 1: What is biodiversity?

The literature on how to measure biodiversity is vast and shall not (cannot) be reiterated here (see Magurran and McGill 2011; Pereira et al. 2013; Chao et al. 2014 for in-depth discussions). Instead, we focus on a few general facets of how compositional diversity can be measured, introducing some essential terms discussed in the main text.

α-diversity

The smallest grain of diversity measurement is the local or sample diversity, also known as α-diversity (Whittaker 1960). Measuring biodiversity is affected by sample size (e.g., area, volume, number of individuals), taxonomic accuracy, representativeness (e.g., temporal or spatial coverage), and effort, which can include the time spent sampling, the area or volume sampled, the absolute number of individuals analyzed, or the proportion of the sample classified. Thus, in contrast to physicochemical measurements, any given biodiversity number is only an approximation of the true diversity (Appendix Fig. 4A). Even when sample size is kept constant, minor changes in the dominance of species, their distribution in space, or in the total abundance will influence most α-metrics (Chase and Knight 2013).

At its core, α-diversity comprises at least two potentially independent axes of biodiversity: the number of items (genets, species, groups, e.g., species richness, S) and evenness, as the absence of dominance, which is highest when all species share the same proportion of the total abundance (Appendix Fig. 4B). Several diversity indices have been proposed that combine these dimensions with different emphasis on dominance (Shannon 1948; Simpson 1949) (Appendix Fig. 4A). Using the Hill-number approach (Hill 1973), these metrics can be aligned along a systematic series of increasingly reflecting dominance, and they can be transformed into a richness equivalent, the effective number of species (ENS). ENS is the minimum number of species required to reach a certain diversity and thus can be easily compared to richness S (which equals Hill-number 0, i.e., all species are weighed equally, independent of their abundance).

γ-diversity

The regional diversity or γ-diversity (Whittaker 1960) is the entire biodiversity derived from all α-scale sampling and increases in a decelerating power function with the cumulative number of samples in space or time (Appendix Fig. 4C). The former leads to the well-known species-area curve (Arrhenius 1921), and the latter to the species-time curve (Preston 1960)—both mutually influence each other (Adler et al. 2005). In principle, also γ-diversity can be described among the same axes and with the same Hill-number approach as α-diversity, but when γ-diversity is derived from distribution maps and range estimates rather than ground-truth sampling, the information on dominance is missing, and γ-richness is the only estimate at the regional scale.

Biodiversity change

The focus of this paper is temporal biodiversity change (see main text), and there are in essence three major approaches. First, we can analyze linear or non-linear trends in S or ENS as a measure of net turnover, as it measures the balance between increases (immigrating species, lowered dominance) and losses (extirpated species, increased dominance). Second, we can measure horizontal turnover by comparing two (adjacent) samples in time and measure their dissimilarity on a scale between 0 (complete overlap, no turnover) and 1 (no overlap, complete exchange). This horizontal turnover can make use of the same range of Hill-numbers using different dissimilarity metrics. Since these metrics cannot exceed the maximum of 1, they are not suitable for analyses across different time scales, as turnover can only decelerate with adding more samples. Instead, the initial slope of the relationship between dissimilarity and distance (in time, space or along environmental gradients) can be measured. Using similarity rather than dissimilarity leads to a negative relationship that is known as distance-decay (Graco-Roza et al. 2022). Alternatively, the ratio of γ/α-diversity gives β-diversity (Whittaker 1960) or hierarchical turnover. Classically done for species richness, where it gives the factor of how many more species there are in a region compared to the average community, it can be extended to higher Hill-numbers. Then the ratio gives the effective number of communities.

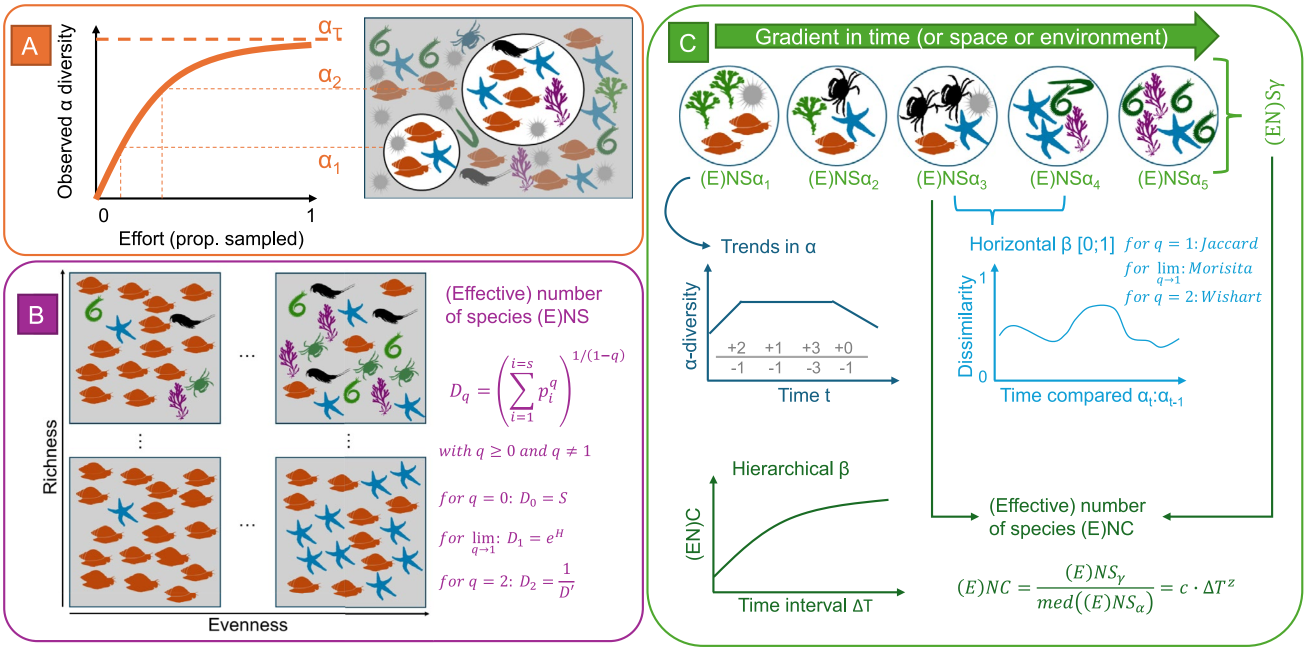


Fig. Box 1 Different approaches to measuring biodiversity. Species are represented by PhyloPic vignettes and color. **A** True diversity of a community (α_T) approached by small (α_1) or large (α_2) sampling effort; **B** dimensions of α-diversity and their unification using Hillnumbers q to weigh the importance of species proportions (p); and **C** γ-diversity over time (but analogously also over spatial or environmental gradients) and the three aspects of biodiversity change: trends in α-diversity, horizontal β-diversity (dissimilarity), and hierarchical β-diversity ([effective] number of communities)

Any of these turnover approaches is affected by the same caveats on effort-dependency. Undersampling the local community scale (too small effort) underestimates α-diversity, but overestimates β-diversity as part of the turnover is species that were present in both samplings, but were missed by one of them. Defining the region too narrowly results in an underestimation of γ and thereby an underestimation of hierarchical turnover. Using multiple Hill-numbers is highly recommended, as more dominant species are more reliably sampled and thus undersampling issues are alleviated.

Box 2: status of biodiversity assessment using the EU as an example

We report on the comparison of marine conservation frameworks and the availability of indicators on biodiversity change. For the available literature on indicators from European waters, we used a systematic review of scientific literature (O'Dea et al. 2021); for details on the approach, see Supplementary Online Material (SOM 1). Overall, we see quite an overlap between science availability and policy demand, with the number of groups addressed and metrics derived matching the multidimensionality of biodiversity change (see "Measuring 179 biodiversity change" section and Fig. Box 1). From a scientific viewpoint (Fig. Box 2), there is very little information on bacteria, benthic microalgae, meiofauna, and other functionally important groups and an absence of molecular indicators. At the same time, these aspects are also not in demand by the assessment frameworks. Otherwise, there is quite a broad coverage of organism groups from a range of biodiversity aspects (Fig. Box 2).

Table 1 provides a structured overview of the international, European, and regional frameworks in place in European waters and their requested indicators. We extracted these biodiversity indicators from the latest assessment or quality status reports for four European conventions covering the North-East Atlantic, the Baltic Sea, the Wadden Sea, and the Mediterranean Sea (Kloepper et al. 2017; Kloepper et al. 2022; OSPAR 2017, 2023; HELCOM 2023; UNEP 2025). OSPAR and HELCOM represent the most comprehensive strategies as they include several organism groups and multiple indicator types within each ecosystem component. The Trilateral Wadden Sea and Assessment Programme (TMAP), acting in the Wadden Sea, in turn, proposed the calculation of temporal trends, species richness, and dominance structure of phytoplankton in their handbook (CWSS 2008), which, despite its usefulness, is in practice not used for quality status assessments.

Across all assessment strategies, we propose to better intercalibrate the available metrics and to map these indicators onto the major dimensions of biodiversity change outlined in the "Measuring biodiversity change" section and add genetic information ("The molecular revolution to the rescue—soon" section). We also strongly suggest bringing in the human dimension of assessment by addressing the emotional relation as part of the assessment ("Recognizing and integrating emotions in biodiversity conservation" section).

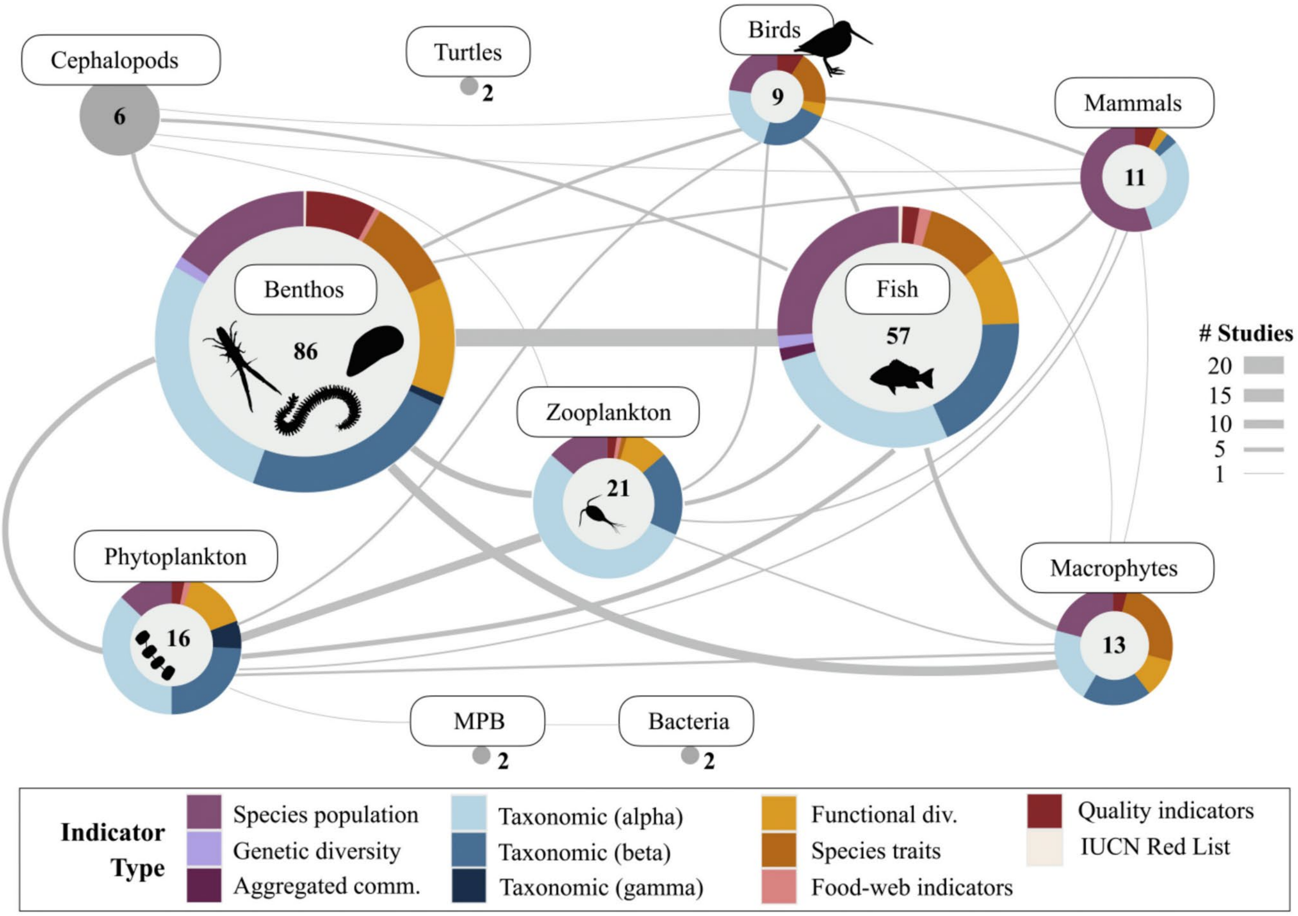


Fig. Box 2 Number of European studies per marine taxonomic group (number and size of the circle) and the links between groups in scientific studies. The thickness of the lines represents the number of studies that simultaneously studied the connected groups. The colored rings represent the relative share of indicator types used to assess biodiversity in scientific studies. Zooplankton includes gelatinous zooplankton; microphytobenthos (MPB)

Table 1 Overview of indicator use in four different regional assessment frameworks

	Pelagic habitats	**Benthic habitats**	**Fish**	**Waterbirds**	**Mammals**	**Habitat**
OSPAR	Plankton index for each life-form pair, time series anomalies in phytoplankton biomass and zooplankton abundance, species richness, species dominance, LCBD, temporal trends in phytoplankton productivity of key species or trophic groups	Multi-metric index of coastal habitats (based on e.g., species richness, proportion of sensitive, tolerant and opportunistic species)	Population size of sensitive species, time series of Large Fish Index, time series of Mean Maximum Length (MML) of each species, size composition (using the Typical Length indicator), Mean Trophic Level (MLT) indicator	Abundance trends in non-breeding and breeding species, population condition (as frequency of widespread breeding failure)	Trends in seal abundance and distribution, population size and distribution of killer whales, coastal bottlenose dolphins and other cetaceans, grey seal pup production, harbor porpoise by-catch	Margalef diversity of subtidal habitats, physical damage of seafloor habitats; pilot assessment of habitat loss in the extent and proportion of benthic habitat types
HELCOM	Diatom/dinoflagellate index, zooplankton mean size and total stock, seasonal succession of dominating phytoplankton groups, cyanobacteria bloom index, chlorophyll-a	State of the soft-bottom macrofauna community	Abundance of coastal fish key functional groups and key species, size structure of coastal fish, abundance of salmon spawners, smolt, sea trout spawners and parr	Wintering abundance, breeding abundance, breeding success, number of drowned waterbirds in fishing gear	Trends/abundance and distribution for grey seals, harbor seals, ringed seals and harbor porpoise; nutritional and reproductive status of seals, number of drowned mammals in fishing gear	
Wadden Sea	Chlorophyll-a concentration (as an indicator for eutrophication)	Total biomass and trends in macrozoobenthos, biomass to polychaete mass-ratio, suspension-to-deposit-feeder mass ratio, temporal density trends in 20 selected macrozoobenthos species, distribution of blue mussels and pacific oysters, seagrasses and sub-tidal habitats	Abundances and trends in fish species, marine juvenile species and estuarine residents	Number, distribution and trends of 35 species, breeding success and survival of 10 selected species, proportions of increasing, stable or decreasing flyway population trends	Number of harbor seals and grey seals, grey seal pups and moult per region, tracks of juvenile grey seals, harbor porpoise density, calf sightings, seasonal and inter annual distribution of harbor porpoises, number of strandings	Extent and distribution of dune vegetation types, salt marshes and vegetation zones, seagrass beds and of sub-tidal habitats, blue mussels and oyster beds
Mediterranean	Chlorophyll-a concentration in the water column (as an indicator of eutrophication)	Coralligenous habitat, Maerl and rhodoliths habitat, *Posidonia oceanica* meadows (as part of the habitat indicators)	Spawning stock biomass, total landings, fishing mortality, fishing effort, catch per unit effort, landing per unit effort, by-catch of vulnerable and non-target species	Species distributional ranges, breeding distribution range, population abundance of 11 selected species, population demographic characteristics (e.g., size, age structure, sex ratio, fecundity rate, survival)	Species distributional ranges, population abundance of selected species, population demographic characteristics (e.g., size, age structure, sex ratio, fecundity rate, survival)	Habitat distributional range to consider habitat extent; condition of the habitat's typical species and communities

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Declarations

Conflict of interest The authors declare no competing interests.

Data availability The study contains no new data.

Author contribution HH designed the study and initiated the writing process, which was jointly done by all co-authors. AH (with support from SN) conducted the systematic literature review and analysis leading to 2.

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